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# detritus

Multidisciplinary Journal for Waste Resources & Residues

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RAFFAELLO COSSU

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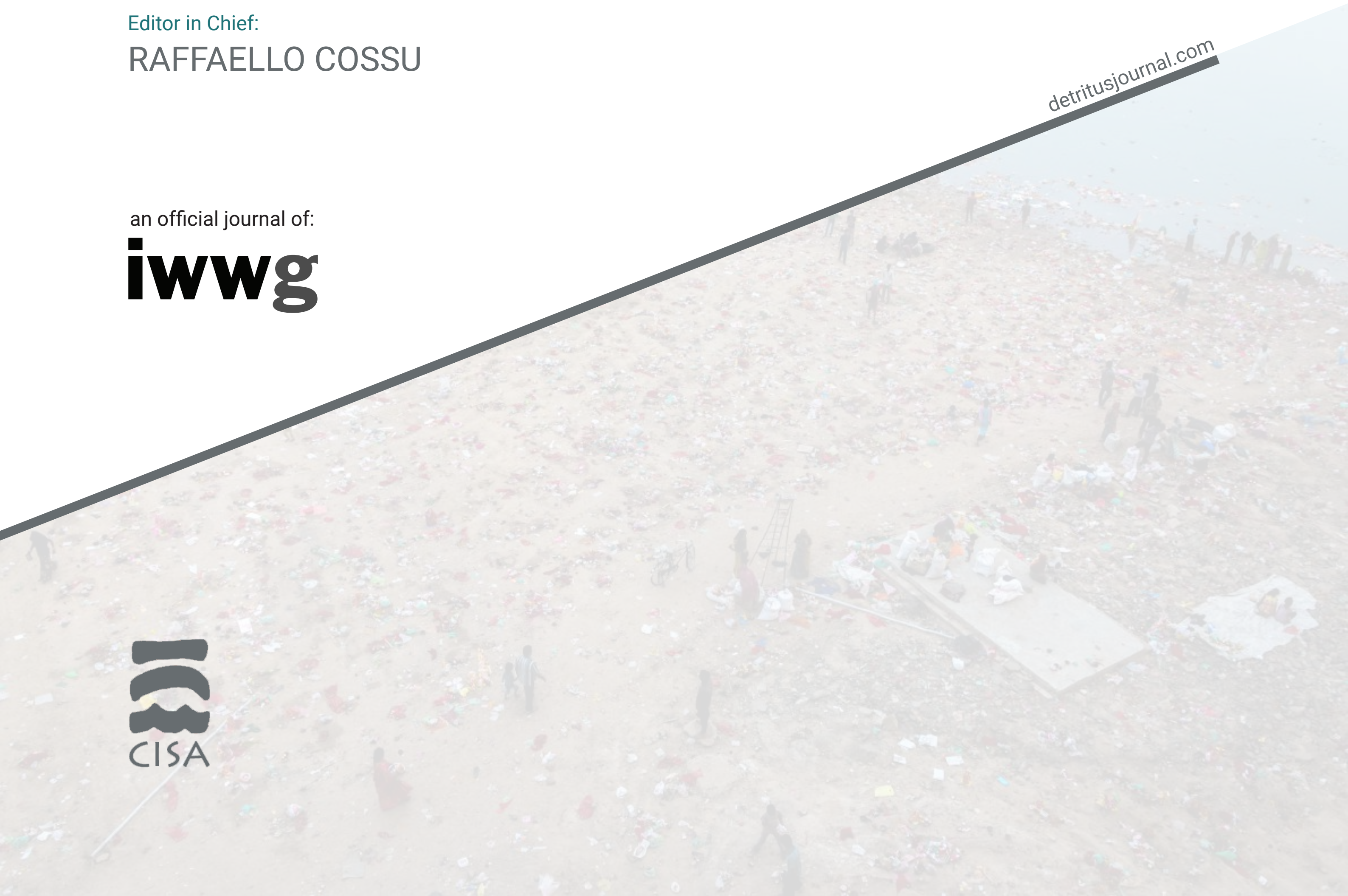

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Detritus – Multidisciplinary Journal for Waste Resources and Residues – is aimed at extending the "waste" concept by opening up the field to other waste-related disciplines (e.g. earth science, applied microbiology, environmental science, architecture, 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

# WHY THIS JOURNAL? WHY THIS NAME?

Science is the main factor involved in promoting and guiding human progress. It implies hard work, and is based on innovation, validation and diffusion.

Journals play a fundamental role in the validation, dissemination and recording of scientific results. Particularly in recent years, the publication of papers in referenced journals represents the main tool in legitimating and certifying scientific and academic careers. However, whilst an accepted publication should be a natural consequence of a virtuous scientific activity, for many young researchers, publication per se is increasingly becoming the main interest. Seen not as a means of advancing scientific knowledge but rather as a tool to increase bibliometric parameters (Cossu, 2013). Of course this tendency is provoked by the prevailing ranking systems where scientific production is measured by numbers and not in term of benefits from scientific findings. The interest to publish as much as possible is currently reflected in an increasing number of journals developed for the specific purpose of commercially exploiting this interest, with accepted Authors being requested to pay a publishing fee (Brunner, Cossu, 2015).

In addition, in the submission of manuscripts, and at times in their evaluation, formal aspects frequently prevail over other important quality parameters such as innovation, relevance, applicability and related social, economic and cultural impacts.

Particularly in the field of waste management, science should indeed be part of a triangular network. One of the vertices of the triangle is represented by Science, with the other two indicating Society and Industry. Society (in which culture, policy, administration, welfare, economy, psychology, education are included) drives and stimulates scientific activities and receives benefits from the same. Industry, and entrepreneurship in general, is committed to supporting and applying scientific knowledge, bringing it into the real world, in term of technologies, products or services, etc.

The aim of this new Journal is to address the above triangle, focusing on integrating all aspects of waste resource management and engineering, and emphasising the sustainability and circularity of material loops for residues.

The journal is aimed at extending the “waste” concept by opening up the field to other waste-related disciplines (e.g. earth science, applied microbiology, environmental sciences, sociology, political science, anthropology, architecture, art, law, etc.) and other scientific sources not limited to research laboratories (full scale plants, best practices, operational activities, discussions within working groups and associations, unsuccessful experiences, case studies from developing countries, administration, etc.).

As a consequence, various types of manuscripts will be welcome including research, review, strategic and opinion papers, position and prospective reports, case studies, country reports, development of new technologies and technical reports. Papers should target a broad audience including scientists, practitioners, managers, public authorities, educators, and students.

The submitted manuscripts will subsequently be evaluated during the review process not in absolute terms but rather in relation to the typology of articles. In an opinion paper, for instance, the absence of experimental data should not be viewed negatively. The presentation of new ideas may be beneficial to the scientific community. As the journal is intended as a multidisciplinary publication open to the needs of society, reviewers should carefully assess the diverse singularities of a specific discipline and should be receptive to potentially different social and economic contexts.

Another aspect in which this journal aims to stand out is the publishing model. In traditional journals, following the peer review process, publishing is frequently free for authors while readers pay by subscribing or buying individual articles. This model however limits the diffusion of publications. In open access journals authors are normally required to pay a publishing fee, and readers are given free access to the articles. A negative aspect of this model is the concern over the consistency and correctness of the acceptance procedure. Both models impact negatively on readers or authors from countries with economic constraints. Detritus intends to synergize the positive aspects of both models: free publication and unrestricted diffusion.

This approach has attracted the interest of highly qualified experts and scientists who have agreed to join the Editorial Board and to manage the peer review process. The process will be carried out using a traditional single blind procedure but in parallel a temporary on line publication of the submitted manuscripts will be offered to Authors, in order to allow the scientific and technical community to express publicly their own views.

Hopefully the above explanation will have provided an answer to the first question: why this journal? Now the second question: why the name Detritus?

The name was inspired by an original proposal put forward by Louis Diaz in 2002 for the naming of the first IWWG journal. It has been selected for the following reasons:

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- it is truly international, deriving from Latin, which was the first international language used in scientific communication;
- it reflects the multidisciplinary character of the journal, as detritus represents a debris or discarded matter of any kind which is encountered throughout our daily lives, in Biology, in Geology, in Psychology, in Culture, in Art, in Architecture, etc.;
- technical terms such as Waste, Recycling, Closing the loop, Sustainable Management, Renewable resources, Circular economy, etc. have already been widely used and abused;
- a detritus does not represent an end but rather a step towards the future. It opens up to hope and perspectives affording new opportunities. Leaves that fall from the trees will fertilise the soil and support new lives. The products of erosion or fracturing of rocks, such as gravel and sand, will help to build new complex structures, and so on;
- our mental detritus in conceiving traditional waste management strategies could indeed represent a starting point for new ideas. This is manifested, for instance, in a negative approach to landfilling, in line with the growing opinion that landfills should be abandoned as an unavoidable source of pollutants and nuisances. However, the deposition of waste on the ground enhances the possibility of controlling the material loop, acting as a sink for substances and elements that, alternatively, would spread and increase the already diffuse contamination.

Detritus, jointly with Waste Management, published by Elsevier, is an official Journal of the IWWG-International Waste Working Group and fully reflects the character of the association as "an intellectual platform to encourage and support integrated and sustainable waste management and promote practical scientific development in the field, … communicating effectively within the professional community".

To stick with Latin … *ad maiora*!

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# SPATIAL AND NON-SPATIAL ANALYSIS OF SOCIO-DEMOGRAPHIC ASPECTS INFLUENCING MUNICIPAL SOLID WASTE GENERATION IN THE CZECH REPUBLIC

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**ABSTRACT**

Municipal solid waste generation has been analyzed in broad range of studies but most of the studies neglect the spatial aspect of analyzed datasets. This paper's aim is to explore spatial dependency in relations between municipal solid waste generation and socio-demographic aspects. The results obtained using geographically weighted regression are compared with results of widely used ordinary least square regression. Even though both methods found the same significant socio-demographic aspects, we were able to explain much higher share of intermunicipal variability using the geographically weighted regression because this method is able to consider changing strength and even direction of relation in different spatial units. Geographically weighted regression can therefore better mirror the local situation and could be successfully utilized to plan waste management activities at local scale.

## 1. INTRODUCTION

Based on a broad range of studies (e.g. Aphale et al., 2015; or Khan et al., 2016) and using various perspectives and methods, after decades of scientific discussions municipal solid waste (MSW) generation and its variability remains in the spotlight. Regarding these discussions, significant variability in MSW generation can be explained by wide spectrum of factors. Selection of these factors is influenced by local conditions, data availability, units used in the analysis as well as by consideration of a researcher. In general, factors explaining MSW generation are divided into two broad categories - individual and situational characteristics (Schultz et al., 1995). Beigl (2004), Hage and Soderholm (2008), Sterner and Bartelings (1999), or Khan et al. (2016) analysed the impact of individual characteristics such as socioeconomic and demographic factors, Bach et al. (2004), Gellynck et al. (2011), Kipperberg (2007), or Starr and Nicolson (2015) focused on waste management organization and charging policy as situational characteristics. Another approach is represented by Sjöström and Östblom (2010), or Arbulú et al. (2015), who studied the relation between solid waste generation and economic growth.

However, most of these studies neglect the possible spatially dependency in MSW generation. The impact of particular characteristics may differ at various geographical locations on the country, region or even municipality level. For example, education may be positively correlated with MSW generation in one country or municipality and negatively in another one. In the second case, the spatial non-stationarity exists in the data. The number of studies dealing with spatial variation in MSW data has been rather limited so far (e.g. Ismaila et al., 2015 or Keser et al., 2012).

The aim of this paper is to assess if global statistical methods give relevant picture about waste management practice and if they are suitable for understanding of waste generation patterns. In our research, we focus on the three socio-demographic factors household size, sex ratio and tertiary education and how they are able to explain MSW generation. To assess the influence of socio-demographic factors on MSW generation we used two different approaches – ordinary least square regression (OLS) and geographically weighted regression (GWR). While OLS gives one statement about relation between analysed variables for the whole area and can therefore represent a global statistical method, GWR as a representative of local statistical methods reflects relations between variables varying in space.

## 2. MATERIAL AND METHOD

### 2.1 Data collection

We used MSW generation per capita and year in kilo-

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grams as dependent variable for non-spatial as well as spatial data analysis. MSW includes mixed municipal waste and separately collected fractions of MSW belonging to the group 20 of the List of Waste. The data for dependent variable were obtained from the state database on waste management "Waste Management Information System" (ISOH). To the system, every waste producer who produces more than 100 kg of hazardous or 100 tons of non-hazardous waste yearly must report his production (SMOCR, 2011). Regarding MSW generation, municipalities are seen as waste producers and they are bound to report into the system if they surpass the aforementioned limit. In 2011 about 4% of all Czech municipalities did not report its waste production because of their low MSW generation. However, all these municipalities are rather small and only 1% of state population lives there.

Based on previous study by Rybová and Slavík (2016), we selected three socio-demographic characteristics as independent variables - average household size (HHS; number of persons per household), sex ratio (IMA; computed as a number of men per 100 women) and proportion of people with tertiary education (TER; computed from the population aged 15 and more years, in %). In Czech conditions, these three variables are significantly correlated with MSW generation. Because all three indicators on municipal level could not be obtained from routine yearly statistics collected by Czech Statistical Office or any other institution, we used the values from the Population and Housing Census. In the Czech Republic, the last Census was organized by Czech Statistical Office in 2011 (Czech Statistical Office, 2013). That is the reason why we could analyse the relationship between dependent and independent variables for this year only.

The initial sample consisted of all Czech municipalities (6,251 in 2011), but for the analysis it was reduced to approximately 5,500 municipalities. First, we removed all municipalities reporting the absence of (or a zero value for) municipal waste (resulting in a new total of 5,820) from the state database on waste management ISOH. Then we sorted the sample based on municipal waste per capita level and removed extreme values from the top and bottom of the list (trimming top and bottom municipalities from the list, which differed by more than three standard deviations from the average). Lastly, we removed 13 cities, mostly regional capitals, because the data from ISOH and Census were available for different administrative units that complicated the comparability and they are also too heterogenous to be sufficiently described by a single value of each variable.

The resulting sample consists of 5,445 municipalities, which still represents 87% of all Czech municipalities. Table 1 depicts the basic information about the sample and characteristics of used variables. The average MSW generation in 2011 was 276 kg per capita.

From Table 1 it is obvious that even after removal of outliers there is considerable variability in MSW generation among Czech municipalities. The local distribution is presented in Figure 1. Even from this picture we can assume that there is some kind of spatial non-stationarity. Spatial clusters of municipalities with higher MSW production are located especially in Bohemian (western) part of the Czech Republic. In the eastern part of the country the variability as well as the average MSW production is lower.

## 2.2 Non-spatial data analysis

In the non-spatial part of the analysis, we applied OLS to get global regression coefficient estimates. The spatial dependency was omitted in this case. The estimated model has a form:

$$Y_i = \beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_3 X_{3i} + \cdots + \beta_k X_{ki} + \varepsilon_i \qquad (1)$$

where $Y_i$ are observations of dependent variable, $X_{1i}$, $X_{2i}$, ..., $X_{ki}$ are observations of independent variables, $\beta_0$, $\beta_1$, ..., $\beta_k$ are the underlying regression coefficients and $\varepsilon_i$ are random errors.

Stability of error variability was tested via Glejser test and its normality via Kolmogorov-Smirnov test. Linearity was examined using a scatterplot between the standardized predicted values and the standardized residuals (Lebersorger and Beigl, 2011). The computations were made using IBM SPSS Statistics 20.

## 2.3 Spatial data analysis

To test the spatial stationarity of the data, we used the Koenker's studentized Breusch-Pagan statistic. Significant result of this test indicates statistically significant hetero-

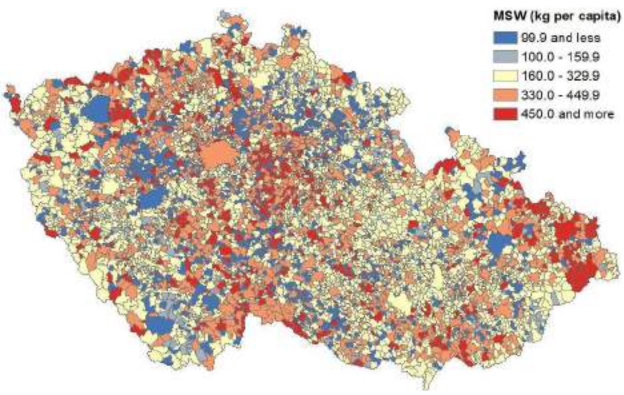


**FIGURE 1:** MSW generation in Czech municipalities, 2011.

**TABLE 1:** Basic statistical characteristics of MSW generation and selected socio-demographic indicators, Czech municipalities, 2011.

| Variable | Minimum | Lower quartile | Mean | Median | Upper quartile | Maximum | Std. deviation |
|---|---|---|---|---|---|---|---|
| MSW (kg/cap.) | 10.0 | 197.3 | 275.8 | 273.3 | 351.6 | 770.6 | 131.4 |
| HHS (person) | 1.3 | 2.4 | 2.6 | 2.6 | 2.7 | 3.7 | 0.2 |
| TER (%) | 0 | 6.0 | 8.8 | 8.0 | 11.0 | 35.0 | 4.4 |
| IMA (no) | 48.4 | 95.2 | 101.7 | 100.0 | 105.8 | 369.8 | 11.2 |

scedasticity and/or nonstationarity. Based on the results of Koenker's statistics, we proceeded to computation of GWR. If we assume that the relation between dependent and independent variables varies spatially, we can use GWR to explore local relation by moving spatial kernel through study area. Kernel functions are used to calculate weights that represent spatial dependence between observations. So, unlike OLS, GWR gives us so called local coefficients $\beta$ that are specific to each areal unit (Keser, 2012). For each model calibration location, $i$ = 1…, n, the GWR model is

$$Y_i = \beta_{i0} + \sum_{k=1}^{p-1} \beta_{ik} X_{ik} + \varepsilon_i \qquad (2)$$

where $Y_i$ is the dependent variable value at location $i$, $X_{ik}$ is the value of the $k$th covariate at location $i$, $\beta_{i0}$ is the intercept, $\beta_{ik}$ is the regression coefficient for the $k$th covariate, $p$ is the number of regression terms, and $\varepsilon_i$ is the random error at location $i$ (Wheeler and Paez, 2010).

The GWR model was computed using ArcGIS.

## 3. RESULTS AND DISCUSSION

### 3.1 Non-spatial data analysis

The results of OLS are statistically significant but the model explains only 2.9% of the variation of MSW between municipalities (Table 2), which is relatively low compared to other studies. The results indicate that the household size has the highest relative impact, followed by sex ratio and the share of tertiary educated people. Household size and sex ratio have negative impact on MSW generation, while education has positive impact. That means that bigger households produce in average per capita less MSW than smaller households and men produce in average less MSW than women. In the case of education, in municipalities with higher share of tertiary educated people is higher MSW generation.

### 3.2 Spatial data analysis

In second step, in order to analyse spatial stability of the influence of the three selected socio-demographic variables we computed the Koenker's studentized Breusch-Pagan statistic. The results of this test were statistically significant, which indicates heteroscedasticity and/or nonstationarity. Therefore, the application of GWR is in this case justified.

The GWR model explains 73% of intermunicipal variation in MSW generation. The local $R^2$ distribution for analysed municipalities is visualized in Figure 2, the values

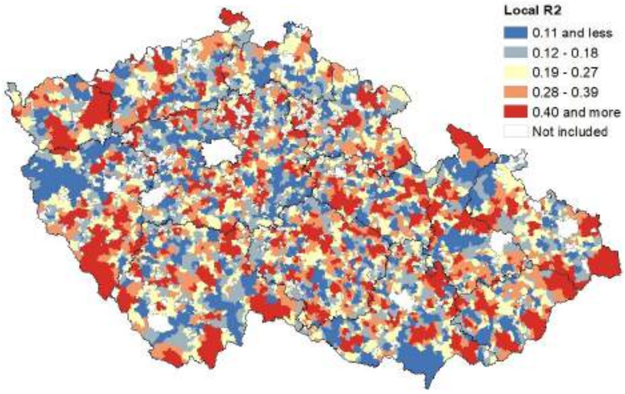


**FIGURE 2:** The local $R^2$ distribution for GWR model, CR, 2011.

range from almost zero to 85%. We detected significant spatial non-stationarity regarding influence of independent variables on MSW, the relation to MSW changes spatially, but we can also detect regions with similar patterns of MSW generation. It is interesting that the local $R^2$ does not correlate with population size of municipalities, demographic variables therefore do not explain situation in smaller cities better than in bigger citiesand vice versa.

The local regression coefficients show variation over the study area and moreover, all three variables incur negative as well as positive coefficients in different municipalities as shown in Figure 3. This result indicates that effect of independent variables does not only vary, but can have even opposite influence in different municipalities.

Regarding household size, it is generally assumed that there is a negative relation (e.g. Beigl et al., 2004; Beigl et al.; 2008). That means that the average generation of MSW per person decreases depending on the growing number of household members. Some products are still bought by households regardless of their size. We confirmed this relation on the global level using OLS, but on the local level there are also municipalities with opposite relation. On the local level, 60% of analysed municipalities show negative coefficient estimates, on the contrary, in 27% of municipalities the coefficient is positive.

Situation regarding sex ratio and its influence on MSW generation is similar. On the global level, there is also negative effect. This supports the conclusion of Talalaj and Walery (2015) that men produce less MSW than women. On the local level, we found the same direction of relation in 51% of municipalities, in 35% of units the relation is op-

**TABLE 2:** OLS model for MSW generation, CR, 2011 (Source: Authors).

| | Coefficients | Standardized Coefficients | t Value | Significance |
|---|---|---|---|---|
| Constant | 541.01 | | 23.89 | 0.000 |
| HHS | -92.21 | -0.16 | -11.56 | 0.000 |
| TER | 1.15 | 0.04 | 2.86 | 0.000 |
| IMA | -0.38 | -0.07 | -5.21 | 0.000 |
| Number of observations | | | | 5445 |
| $R^2$ | | | | 0.029 |
| Adjusted $R^2$ | | | | 0.029 |

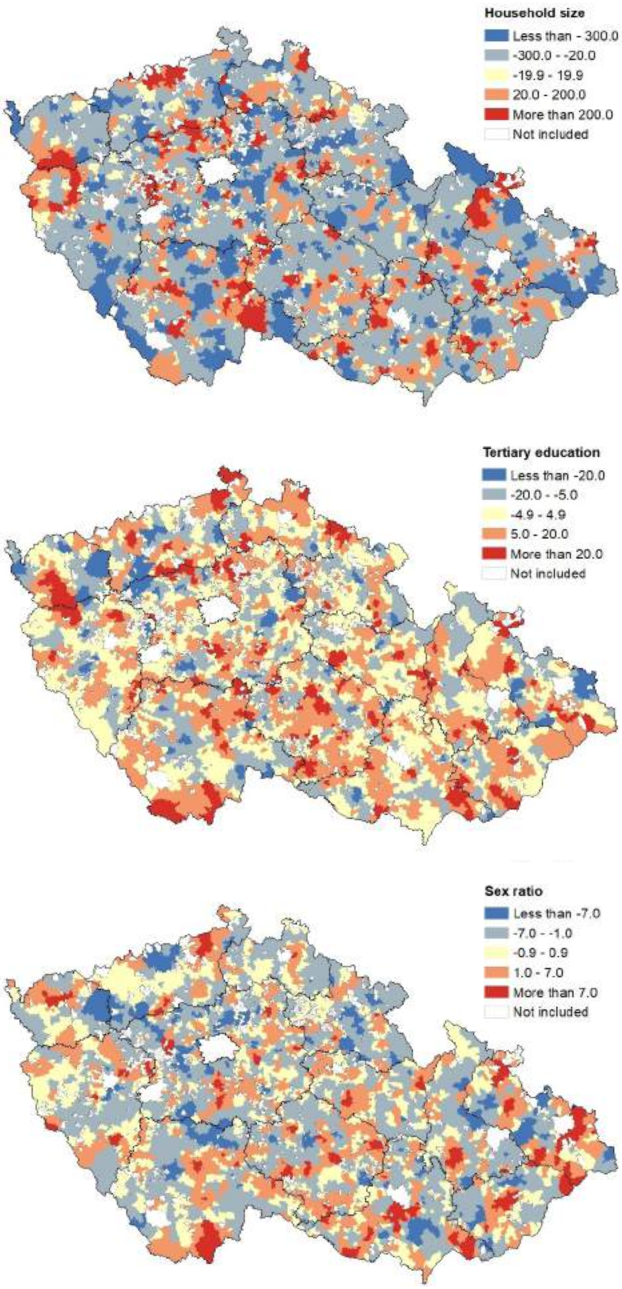


**FIGURE 3:** GWR coefficient estimates, CR, 2011.

posite, and the rest shows no impact of sex ratio on MSW generation.

As in the global model, share of population with tertiary education is the only predominantly positively correlated independent variable. The coefficient estimates are positive in 50% of municipalities and negative in 37% of units. Education as independent variable was also used in the study of Keser et al. (2012). On the global level this variable was not statistically significant, but on the local level the authors discovered in some provinces positive relation. It is interesting that they did not find any provinces with negative relation as is the case in our paper.

Because the spatial analysis is not a widely used method in explaining MSW generation so far and based on the used data it is hard to explain the differences in particular factors. There are probably other variables (such as environmental values, situational characteristics incl. socio-demographics, and psychological factors incl. social norms, Barr, 2007) that are influencing the behaviour of the inhabitants in the regions. An important role could play also the location of the municipality regarding metropolises or periphery regions (including the so called inner periphery that is in the Czech conditions influencing many social aspect, Musil and Müller, 2008). Further studies should consider more spatial information such as population density, characteristics of housing or average income of the households.

## 4. CONCLUSIONS

The results of GWR show that to understand MSW generation on municipal level, we cannot use only standard global statistical methods (such as OLS). As concluded, spatial effects matter. Based on the GWR method, the socio-demographic characteristics have significant influence on MSW generation, but this influence varies spatially and has even opposite signs in different municipalities. This situation can diminish the detected variability explained by OLS and can lead to neglecting of socio-demographic aspect in decision making. Our conclusion is important for waste management planning as well, because it supports application of the subsidiarity principle in the practice. Even though objectives of the waste management policy are given at national level, many decisions are made at the local level (Lazarevic et al., 2012)

Further important benefit of the GWR method is a fact that it allows to define areas with a similar character of the examined relation, which may serve as the basis for follow-up analyses carried out directly in the chosen regions.

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# A COMPARATIVE STUDY OF ISSUES, CHALLENGES AND STRATEGIES OF BIO-WASTE MANAGEMENT IN INDIA AND ITALY

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Italy

## ABSTRACT

The aim of the present study is to compare the current level of implementation of bio-waste management in Italy and India. Italy generates about 1.33 kg/per capita/day of municipal solid waste (MSW). Bio-waste makes up about 30% of the whole MSW generated and about 60% is recycled. The main process is by aerobic composting, whereas anaerobic digestion is used to a limited extent. In India waste production ranges from about 0.20 kg/per capita/day to about 0.60 kg/per capita/day. The amount of recycling of this waste is very poor and there is a lack of treatment facilities. Anaerobic digestion for the Indian scenario could be a suitable solution for co-treatment of bio-waste with other biodegradable materials in order to supply energy and fuel in rural areas. The main differences between the two countries concerning waste and in particular bio-waste management are mainly the recycling strategies, end of waste criteria and energy recovery perspective.

## 1. INTRODUCTION

Even with different perspectives, waste management is one of the key issues to be addressed both by developed and developing countries in order to achieve a sustainable implementation of the different human activities worldwide.

According to Marshall and Farahbakhsh (2013), progress in this activity was historically driven by five key factors: public health, the environment, resource scarcity and the value of the waste, climate change, and public awareness. In particular health care and the environmental aspect are affected directly, as for example, by the emissions generated from incorrect collection and disposal of waste (Couth and Trois, 2011, 2012; Tian et al., 2013, Aich and Ghosh, 2016), and indirectly as a consequence of raw materials consumption and transformations (Di Maria and Micale, 2014-2015).

Nowadays the most effective approach for waste management recognized worldwide is based on the 3-R concept: Reuse, Recycle and Recovery. This was extrapolated from the broader concept of the waste management hierarchy introduced in the EU in 1977 by the European Commission (CEC, 1977), stating the main activities and goals to be pursued with strict hierarchical order in waste management: Prevention, Reuse, Recycle, Recovery, Disposal.

The concept of hierarchy was definitively introduced in the EU legislation in 1991 by the first Directive 91/156/EEC on waste (Council Directive, 1991), becoming a fundamental component of the integrated waste management approach. This was in force up to 2008 when the latest Waste Framework Directive (WFD, 2008) introduced another important goal to be achieved within 2020 by member states, stating that at least 50% of waste generated has to be reused or recycled. Recycling also includes the organic fraction via biological treatments, able to generate organic fertilizer exploitable in agriculture according to legislation of the member states.

Furthermore, putting the waste management hierarchy into practice was also indicated by the EC as a key activity in communication n.614 (COM 2015) concerning the EU Action Plan on circular economy. A key factor for maximizing recycling and reuse is proper waste collection based on efficient source segregation able to return high quality recyclables directly exploitable in the recycling industry. Municipalities are the authorities charged with providing municipal solid waste (MSW) collection directly or by private/public companies. Presently in the EU15 and in Italy, collection coverage is practically 100%.

Continuous effort for the full implementation of these concepts and goals led to the following main figures concerning MSW management at the EU15 level (ISPRA, 2016):

- Global MSW generated: 207,862,000 Mg (i.e. about 1.4

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kg/day per capita);
- Fraction of MSW recycled: 29.5%;
- Fraction of MSW composted and/or processed by anaerobic digestion (AD) for recycling: 17.4%;
- Fraction of MSW incinerated: 29.9%
- Fraction disposed in sanitary landfill: 23.1%.

Considering that the percentage of bio-waste in MSW at the EU15 level is about 30%, the above figures indicate that more than 50% is currently recycled by composting or processed with AD and post-composting (Di Maria et al. 2016; Smidt et al., 2011). The remaining amount could be considered quite equally shared between incineration and landfilling. All landfills currently operating are sanitary landfills that have all of the necessary equipment required by current legislation for emissions control and also very often with landfill gas energetic recovery.

Currently the EC is still focusing particular attention on bio-waste management. In fact, even if collected with high source segregation efficiency, bio-waste, unlike other waste materials (e.g. plastics, paper, metals), as returned cannot be directly exploited as raw material by the recycling industry. Furthermore its disposal in landfill is a serious environmental threat due to the generation of gaseous and liquid (i.e. leachate) emissions having high polluting potential. According to the European Environmental Agency (EEA, 2011), landfill gas emissions contribute up to 3% of the whole anthropogenic greenhouse gas (GHG) emissions in the EU due to the high amount of methane (i.e. about 50% v/v) and $N_2O$ with a GHG potential 23 and 300 times higher than $CO_2$, respectively (Beyolt et al., 2013; De Gioannis et al., 2009; Desideri et al., 2003; Di Maria et al., 2013a). On the other hand, mainly as a consequence of rainwater infiltration, landfills generate leachate that can be considered a triphasic system with the characteristics of a highly polluted wastewater with high concentrations of: organic and inorganic contaminants; pathogens; humic acids; ammonia nitrogen; heavy metals; xenobiotic and inorganic salts (Di Maria et al., 2018). The content of these substances is influenced by many factors (e.g. waste composition, climatic conditions, extent of waste degradation/decomposition) and must be removed in accordance with EU water standard legislation (Di Maria and Sisani, 2017, Landfill Directive 1999/31/EC; Schiopu and Graviliescu, 2010; Slack et al., 2005; Spagni et al., 2008; Wisizniowski et al., 2006).

On the other hand, waste management in the majority of developing countries is still primarily based on uncontrolled dumping and/or littering (Henry et al., 2006; Sharholy, et al., 2008) together with domestic burning (Guerrero et al., 2013), causing serious health and environmental problems (Al-Khatib et al., 2010). As reported by Kumar et al. (2009), 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% (Henry et al., 2006; Zhang et al., 2010). Couth et al. (2001) reported that in Africa GHG emissions from waste management are 3 times higher than those in developed countries and similar results were also reported by Tian et al. (2013) concerning the Chinese scenario. Zhang et al. (2010) reported per capita production in China ranging from 0.4 kg/day to 1.0 kg/day (with the peak achieved in given areas also up to 1.7 kg/day (Manaf et al., 2009). The organic fraction ranged from 45% up to more than 80% of the whole waste generated (Al-Khatib et al., 2010; Henry et al., 2006; Zhang et al., 2010), leading to serious health and environmental concerns. In general the main goal in waste management consists in its transport outside of cities (Marshall et al., 2013). Furthermore the rapid and unplanned growth of cities has resulted in a number of extreme land use planning and infrastructural challenges that have crippled the capacity of government and local authorities to increase MSW service to the degree they are demanded (Marshall et al., 2013). Collection services are also inadequate due to lack of funding and technical expertise (Al-Khatib et al., 2008; Henry et al., 2006). Similarly, Guerrerio et al. (2013) reported that failure of waste management in cities of developing countries is due to inadequate technical, environmental, financial, socio-cultural, institutional and legal aspects. A primary role for recycling is played by informal waste scavenging and picking, often done in unsafe conditions directly on dumpsites or on collection trucks, scattering waste all around along the route (Manaf et al., 2009). In general recycling figures are very poor, less than 10%, (Kumar et al., 2009). Also there is a major lack of facilities for the treatment of the largest and most threatening waste component, the organic fraction.

As indicated by Henry et al. (2006), composting is a sustainable way to manage organic waste in these countries, leading to environmental protection as well as to generating revenue from selling fertilizer. Similar recommendations were also reported by Sharholy et al. (2008) for improving the rural economy in India. Couth and Trois (2011, 2012) indicated composting as one of the main activities to pursue for sustainable waste management in Africa, able to reduce GHG from about 900 $kgCO_2eq/Mg$ to about 300 $kgCO_2eq/Mg$ compared to disposal in landfill. Kumar et al. (2009) reported that centralized composting facilities produced poor quality fertilizer due to the absence of source segregation of the organic fraction, also causing decrease in interest by potential investors. Promoting decentralized facilities for community composting has been indicated as an alternative solution, able to overcome the low quality of fertilizer of centralized plants (Henry et al., 2006; Sharholy et al, 2008). As an alternative to decentralized composting, some authors have indicated decentralized AD as another suitable way for processing organic waste in developing countries. AD is a widespread method for disposal of various types of waste and returns a biogas with a methane content from 50% v/v to 70% v/v (Bond and Templeton, 2011). Various appliances such as stoves, electrical generators, and lighting and cooking devices can be fuelled with biogas, offering an appropriate application for its use in developing countries. Furthermore, due to the absence of transmission and distribution of energy generated from fossil fuels in rural areas, particularly in remote locations, decentralized renewable energy generation could be an important contribution to improve the quality of life in such areas (Demirbas and Demirbas, 2007).

AD is able to ensure sanitation of biodegradable compounds by reducing the pathogen content in substrates (Bond and Templeton, 2011) and by reducing in-door emissions. In fact the biogas can substitute the use of woody/solid fuels for fuelling stoves, reducing in-door particulate emissions (Demirbas and Demirbas, 2007; Houng et al., 2014) and hence reducing health risks. There has been rapid improvement in public health in China due to use of biogas, with a reduction of schistosomiasis and tapeworm by 90% and 13%, respectively (ISAT/GTZ, 1999; Remais et al., 2009). Furthermore the digestate returned from AD also showed rather good fertilizer properties, with the potential of improving soil fertility (Smidt et al., 2014; Di Maria et al., 2013b). Decentralized and micro-AD facilities have already been developed and adopted in many areas of developing countries such as the Chinese fixed dome, the Indian floating dome and the PVC digester tube (Bond and Templeton, 2011; Ferrer et al., 2011; Mungwe et al., 2016).

The aim of this paper is to compare current figures and challenges in the Italian and Indian scenarios concerning bio-waste management. Possible exchanges of experiences and good practices are also analysed and discussed.

## 2. MATERIALS AND METHODS

Assessment of the current management schemes for bio-waste in Italy and India was performed using official documents obtained from local and central authorities, literature surveys and from direct observations in given areas and facilities.

The analysis included methodologies, technologies, legislation, and social and economic aspects associated with the different areas analysed. In particular the two areas were compared using the following three main indicators:

- Recycling strategies;
- Presence of end of waste criteria;
- Energetic considerations.

### 2.1 Italian scenario

Like other sectors, waste management legislation for all the EU member states including Italy is based on Directives of the European Commission, the European Parliament and the Council.

According to the EC Environment, bio-waste is defined as "biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. It does not include forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles, paper or processed wood. It also excludes those by-products of food production that never become waste". Relevant Directives concerning the management of bio-waste are mainly the Waste Framework Directive (WFD, 2008), the Landfill Directive (LFD, 1999), and the Integrated Prevention Pollution Control Directive (IPPC, 2008).

The WFD indicates the recycling goals and the need to activate dedicated collection services for bio-waste. LFD mandates Member States to reduce the amount of biodegradable municipal waste that they landfill to 35% of 1995 level by 2016 (for some countries by 2020). Finally the IPPC Directive, soon to be substituted by the Industrial Emission Directive, also indicates measures to prevent and/or reduce the emissions generated by waste management and treatment.

Other important legislation concerning bio-waste is by the Italian D.M. (1998) and by the D.Lgs. (2010). They provide current End of Waste (EoW) criteria for bio-waste, while waiting for the adoption of European criteria. The former indicates aerobic composting, and associated performance (Table 1), as a suitable process for bio-waste recycling together with the waste type. The European directive defines the chemical and physical quality of the final compost for consideration as organic fertilizer and then exploitable/recyclable in agriculture in compliance with the legislation of different member states. The Italian standards for quality for organic fertilizer are reported in the D.Lgs. (2010). Another process suitable for bio-waste recycling is by AD, but presently the absence of specific EoW criteria at both the EU and Italian level pose some criticism for the full implementation of this process.

Aerobic composting of bio-waste separated at source, alone or combined with AD, is the main and most commonly used recycling option. Other possibilities for processing biodegradable waste not separated at source before final disposal are by mechanical and biological treatment (MBT) and/or incineration. MBT can be carried out by bio-stabilization or bio-drying. The aim of biostabilization (Adani et al., 2004; Di Maria, 2012; Zach et al., 2000) is to reduce the biological reactivity of the biodegradable components, prior to mechanical sorting, before disposal in landfill in compliance with LFD. The aim of bio-drying (Adani et al., 2002; Wiemerand Kern, 1995) is to produce a Solid Recovered Fuel (SRF) (UNI CEN/TS, 2006) for fossil fuel replacement, also including the carbon-rich bio-waste content by removing excess humidity via aerobic treatment.

*2.1.1 Recycling*

The MSW generated in Italy in 2015 (ISPRA, 2016) was 29,524,263 Mg, representing 14.2% of the whole MSW generated in the EU15. On average, the fraction of waste collected separately was 47.5% with a maximum value of 58.6% in the northern regions. For central and southern regions these figures were 43.8% and 33.6%, respectively. More than 20% of the waste collected separately was bio-waste (i.e. about 6,000,000 Mg), indicating a specific source separation level > 60%.

Practically all the bio-waste collected separately was processed for being recycled as organic fertilizer in the

**TABLE 1:** Waste type and main performances for aerobic composting according to Italian End of Waste legislation.

| Parameter | Value | u.m. |
|---|---|---|
| Waste type | Bio-waste from separated collection | - |
| Days of treatment | 90 (min.) | Day |
| Process temperature | 55 (at least for 3 days) | °C |

263 biological treatment facilities (Di Maria 2012; Di Maria et al., 2014) operating at the national level, 26 of which are equipped with integrated anaerobic digestion and post composting (Di Maria et al., 2016: Smidt et al. 2011). There are 162, 43 and 58 facilities for Northern, Central and Southern regions, respectively. Bio-waste processed in the integrated AD and post-composting facilities was about 1,600,000 Mg, equal to the whole amount of organic fertilizer (recycled) generated in 2015. About 220,000 Mg of bio-waste were co-digested exclusively with other substrates in 20 AD facilities, 18 of which were located in Northern and 2 in Southern Italy. According to the current legislation AD alone was not able to comply with the End of Waste criteria necessary for agronomic utilization (D.M., 1998; D.Lgs, 2010). From the technical point of view, the first limitation was due to the temperature levels needed for achieving the sanitization requirements (Table 1), making only thermophilic processes able to warrant > 3 days of treatment at not less than 55°C. The second limitation was the lack of complying with the characteristics required by the organic fertilizer legislation (Table 2). One of the main criticisms was the low level of the Germination Index, indicating significant residual phytotoxicity of the digestate. Di Maria et al. (2013b, 2014) and Massaccesi et al. (2014) reported a Germination index of digestate ranging from 35.7% to 53%, resulting lower than the threshold value of 60% imposed by Italian legislation (Table 5). These problems can be solved by a successive post composting treatment. In any case even if EoW criteria were not fully met, use on land can be performed by specific authorizations released by the competent authorities. On the other hand AD leads to the production of renewable energy even if investment, operating and maintenance costs are higher. For this reason there has been a large diffusion of AD facilities since 2013 when economic incentives became available for the production of electricity from biogas. In general, these incentives, up to 0.28 €/kWh, enabled the viability of the whole investment for plant sizes not less than about 1,000 kW (i.e. about 7,000-8,000 MWh/year). At the end of 2013 these incentives were substantially eliminated and the new frontier for incentives for AD was by bio-methane production. Bio-methane can be obtained from biogas by upgrading the process to remove $CO_2$ and other pollutants, returning a gas in compliance with technical requirements (UNI/TR, 2014) for its injection in natural gas grids or exploitable as fuel for transport. In any case a preliminary economic analysis shows that bio-methane plants are viable for thermal power associated with the gas generated > 3,000 MW. The organic fertilizer generated was, in general, quite low, usually < 12 €/Mg, but very often < 5 €/Mg, whereas the inlet fee for such recycling facilities ranges from about 60 €/Mg to about 100 €/Mg.

#### *2.1.2 MBT, incineration and landfill*

In 2015 the amount of waste processed in the 118 MBT facilities, 36 in Northern, 32 in central and 50 in Southern regions, was 10,532,209 Mg, 89.7% of which was from residual MSW coming from separated collection and 7.5% from waste generated from other waste treatments. The remaining fraction was from other wastes generated from civil and industrial sectors. Of the 8,804,068 Mg output generated by MBT, about 64% was landfilled or used for landfill management (e.g. covering layers), 1.3% was recyclables (i.e. metals), 1% was SRF and about 28% was incinerated or co-incinerated. In practice MBT is an alternative solution to incineration quite used due to the lower capital, operating and maintenance costs. Investment costs range from 200 €/Mg/year to 300 €/Mg/Year, depending on plant size and on the technological solutions adopted. The gate fee was from 60 €/Mg to 100 €/Mg. MBT energy consumption ranged from about 30 kWh/Mg to about 50 kWh/Mg of electricity necessary for both process and emission controls. On the contrary MBT is not able to handle the same mass, volume and biological reactivity reduction as incineration, entailing more landfill.

The 41 incineration plants operating in 2015, 26 in Northern, 8 in Central and 7 in Southern regions, processed 5,582,052 Mg of waste, about 18.9% of the whole MSW generated, producing 4,430 GWh of electricity and 2,754 GWh of thermal energy. Even if some studies highlight the environmental performances of alternative processes (Di Maria and Fantozzi, 2004), mass burning is the only technology adopted at the industrial scale. The technologies adopted were: 87% grid, 10% fludised bed, and 3% rotary kiln. Similar results were also reported at the EU level. Of the 15 plants operating in combined heat and power mode, the average electrical and thermal energy recovery per Mg of waste processed was 0.65 kWh and 1.04 kWh, respectively. For the plants recovering electrical energy exclusively, the figure was of 0.77 kWh/Mg. Investment costs for incineration facilities ranged from about 300k€/Mg/day to 500k€/Mg/day, whereas the inlet fees are usually more than 100€/Mg. Finally the amount of waste landfilled was 7,818,796 Mg (26% of the whole generated). More than 80% of this waste was disposed of in landfill after previous treatment as MBT. The landfill inlet fee ranged from about 70€/Mg up to 120 €/Mg, depending on local conditions and on the features of the waste disposed. The high landfill fee is often a consequence of environmental taxes and penalties (landfill levy) aimed at discouraging the use of such facility in waste management, making other solutions and pre-treatment, such as incineration, economically more attractive.

### 2.2 The Indian scenario

In India almost all waste management legislations were revised or introduced in 2016. According to the 2016 solid waste management (SWM) Rules, solid waste is defined as solid or semi-solid domestic waste, sanitary waste, commercial waste, institutional waste, catering and market waste and other non-residential wastes, street sweepings, silt removed or collected from surface drains, horticulture waste, agriculture and dairy waste, treated bio-medical waste excluding industrial waste, bio-medical waste and e-waste, battery waste, radio-active waste generated in the area under the local authorities. Hence the bio-waste in India includes biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. Similar to the EC directives, in India it does not include forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles and processed wood. India is very rich in three distinct sources of biomass energy, namely energy plantations, agricultural crop residue and municipal and industrial wastes. Being an agricultural country, there are high numbers of cattle and livestock in India. Thus, Indian villages have always had a wealth of bio-resources which can easily be converted to energy.

The practice of source separation of municipal waste is limited or absent in most of cities and villages in India. This results in a mixed composition of waste, leading to several constraints in introducing and implementing technologies for treatment in a sustainable way. The 2016 SWM Rules specifically state that waste generators will store source segregated bio-degradable, non-biodegradable and domestic hazardous wastes and they must be collected throughout the country by designated authorities. The bio-degradable waste shall be processed, treated and disposed of by composting or by bio-methanation in the premises as far as possible for gated communities or by the local urban bodies in the waste treatment facility in identified areas including other energy recovery options. The local urban body (LUB), being the lowest body within the three-tier Indian governmental system, has to prepare a solid waste management plan according to state policy and strategy on solid waste management with a target time and submit a copy to the respective departments of the State Government or Union territory Administration or agency authorised by the State Government or Union territory Administration.

The rules specify composting as one of the technologies for processing biodegradable waste in the waste treatment facility and recommend standards for the compost for compliance in order to prevent pollution from the composting plant. The incoming organic waste at the site must be stored properly prior to further processing. To the extent possible, the waste storage area should be covered. If, such storage is done in an open area, it must have an impermeable base with the means for collecting leachate and surface water run-off into lined drains leading to a leachate treatment and disposal facility. Pre-process and post-process rejected material shall be removed from the processing facility on a regular basis and shall not be allowed to pile up at the site. The windrow area must have an impermeable base. Such base shall be made of concrete or compacted clay 50 cm thick and have a permeability coefficient less than 10–7 cm/sec. The base must have a 1 to 2 per cent slope and be surrounded by lined drains for collecting leachate or surface run-off. The leachate must be re-circulated in the compost plant for moisture maintenance. The end product compost is required to meet the standards specified under the Fertilizer Control Order notified from time to time. In order to ensure safe use of the compost, the specifications for compost quality must be met (SWM Rules 2016, India). The Indian rules define the chemical and physical quality of the final compost for being considered as organic fertilizer and then exploitable/recyclable in agriculture. The comparison will be discussed latter in this paper.

The overall flow of 90% of the collected solid waste, nearly 1,270,531 tonnes per day out of 1,410,046 tonnes per day of the solid waste generated (data based on 2013-14, Source: CPCB Bulletin Vol.- I, July 2016, Government of India) is distributed in processing and treatment activities like, Recycling, Composting, biomethanation, waste-to-energy and land filling (Figure 1).

The general supply chain of the solid waste in India has both formal and informal intervention (Figure 2). The collection system in most of the cities is formally carried out by the local urban bodies. Segregation is done by a group of people called rag pickers or Kabadiwala, who are involved informally on their own for separating valuable materials from the waste collected to sell to recyclers for their livelihood, whereas in many cities the LUB are involved in the waste segregation or source segregation collection system.

*2.2.1 MRF and recycling*

In India the various main treatments of the organic fraction of municipal solid waste involve composting, and biomethanation, while the non-biodegradable part of MSW involves material recovery, recycling, RDF, waste-to-energy and the remainder goes to landfill (Figure 1). Waste collection is done in a source-segregated manner as well as mixed types of waste. The source-segregated waste undergoes further segregation for recycling in the Material Recovery Facility (MRF) manually or with the aid of machines. Valuable recyclable waste is separated informally from the landfill or dumpsites by people on their own. Nearly 27% of the MSW processed amounts to 34,752 tonnes per day. As per the CPCB Bulletin (Vol.- I, July 2016), the Government of India follows some of the data of the MSW treatment and disposal status in India as in 2013-14. Table 2 shows the number of composting/vermi-composting plants in the States of India as of 2013. Of course in the next five years, the number will increase at a faster rate because of the new solid waste management rules introduced in 2016. The Government of India supports the use of compost. Co-marketing of compost at 3 to 4: 6 to 7 bags by fertilizer companies is now being promoted. House–to-house collection ranges from 40-90% in 18 States, whereas waste segregation ranges from 20-80% in 5 States. The rest of the Sates are in the process on introducing the process. The recorded number of operational compost / vermi-composting facilities are in 553 Urban Local Bodies (ULBs) and compost/vermi-composting facilities are under construction in 173 ULBs. Pipe composting is very popular in the State of Kerala where more than 7000 units are in operation. There are more than 0.4 million bio gas plants oper-

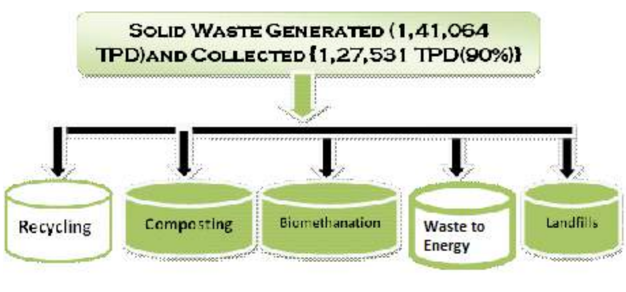


**FIGURE 1:** Flow of the solid waste collected in India.

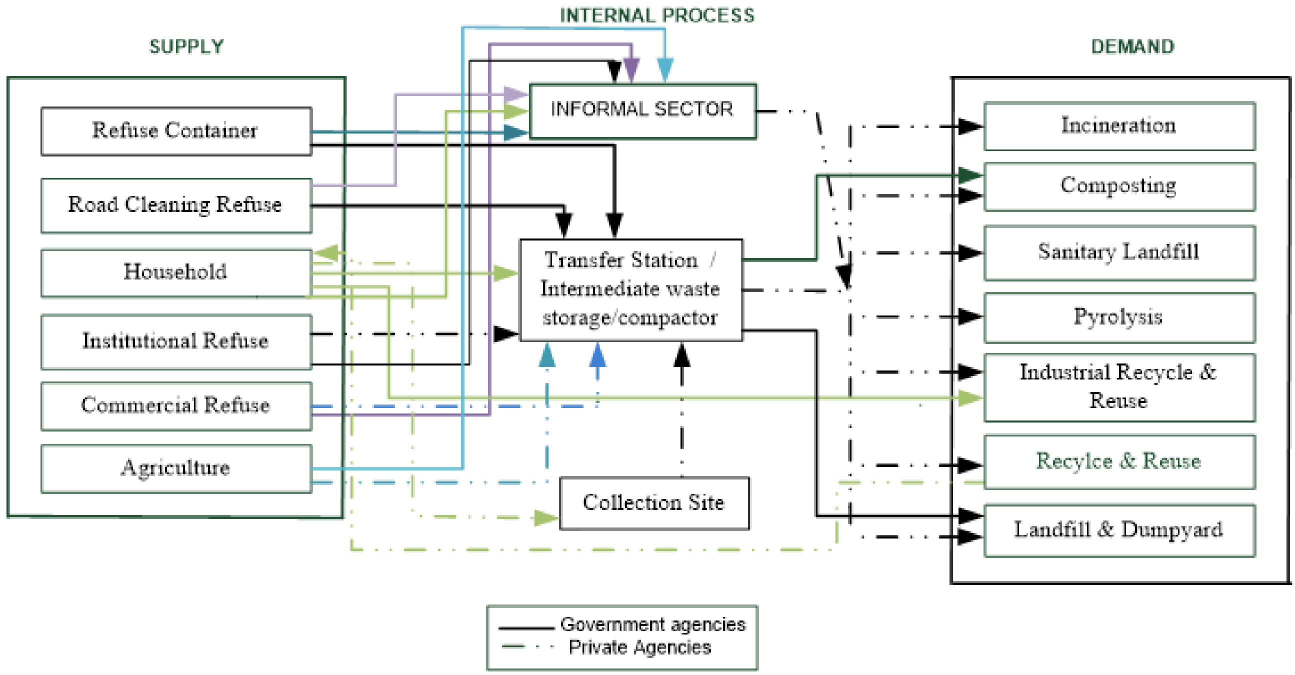


**FIGURE 2:** MSW Supply Chain Framework in India (Source: Ghosh, et al, 2016).

ating in India at the small-scale household level. There are nearly 645 recorded biogas plants carried out by ULBs and industries (600 in Kerala).

*2.2.2 MBT, incineration and landfilling*

More than 70% of the solid waste in India is destined to go to more than 1285 landfill sites and 95 landfill sites are under construction in different parts of the country. The number of sanitary landfill sites is very low. At present nearly 10,000 Tonnes Per Day is landfilled with respect to 4,515 TPD in 2013-14. The 2016 SWM Rules mandate the minimum amount of waste to go to landfill. There are very few waste to energy plants, but there is a new initiative to install WtE plants in many of the States (Tables 3 and 4). Nearly 12 RDF/Pellet manufacturing plants are in operation in India, whereas there are six energy generation plants. The calorific value of waste in India is low (Nixon et al, 2013a,b).

The use of biomass and bio-waste resources at the country level (using all residues including rice husk) and cogeneration (using Bagasse) plants are the major sources of power generation. Nearly 4.9 GW are produced in India from the use of bio resources, out of which 56.28% is generated from bagasse-cogeneration, 27.43% is generated from biomass power gasification, 11.27% is generated from non-bagasse based cogeneration, 3.07% is from biomass gasification in rural areas, 2.15% from waste–to-energy and only 0.36 from gasification in rural areas (Figure 3).

## 3. COMPARATIVE ANALYSIS AND DISCUSSION

In general one of the main drivers detected during this study for improving management of waste and in particular bio-waste is by the combination of efficient and effective political, economic and legal programs. This was implemented the EU and in particular in Italy since the be-

**TABLE 2:** Number of composting / vermi-composting plants in various States in India.

| State | Number of plants (composting/ vermi-composting) | State | Number of plants (composting/ vermi-composting) |
|---|---|---|---|
| Andhra Pradesh | 32 | Madhya | 4 |
| Chhattisgarh | 15 | Maharashtra | 125 |
| Delhi | 3 | Meghalaya | 2 |
| Goa | 5 | Orissa | 3 |
| Haryana | 2 | Punjab | 2 |
| Gujarat | 86 | Rajasthan | 2 |
| Himachal | 13 | Tripura | 13 |
| Karnataka | 5 | Uttarakhand | 3 |
| Kerala | 29 | West Bengal | 9 |

*Source: CPCB (2013)*

**TABLE 3:** Recent initiatives of Waste to Energy Plants in India (as of 2017).

| State | WTE Plant Location/capacity | Status |
|---|---|---|
| Delhi | Okhla (2000 TPD) | Okhla plant operating, one commissioned, one under construction |
| Madhya Pradesh | 600 TPD at Jabbalpur, Indore, Bhopal | Jabbalpur generating power to grid, others are in contract stage |
| Gujarat | Surat, Vadodara, Mahar (3000TPD) | Contract stage |
| West Bengal | Kolkata, and Howrah | Tendering Stage |
| Andhra PradeshTelengana | Four Locations Karimnagar | Tendering Stage - In Operation |
| Maharashtra | Pune two | One failed, one in Tendering Stage |
| Bihar | One Plant | Tendering Stage |

**TABLE 4:** Number of energy recovery plants in some states.

| State | No. of RDF plants/Waste to Energy Plant (PP)/Biogas (BG) | State | No. of RDF plants/Waste to Energy Plant/Biogas (BG) |
|---|---|---|---|
| Andhra Pradesh | 3-RDF, 4 PP | Delhi (UT) | 1-RDF, 1PP |
| Chandigarh (UT) | 1-RDF | Gujarat | 2-RDF |
| Chhattisgarh | 1-RDF | Kerala | 2-BG |
| Maharashtra | 19-BG | Madhya Pradesh | 1--RDF |

*Source: CPCB (2013)*

ginning of the 1990s, and has led to a quite high efficiency and effectiveness of the waste management system both in terms of efficient use of resources and of environmental protection. Citizen awareness concerning waste management has stimulated policy and public administrators to pursue continuous efforts to improve this activity by increasing reuse and recycling, decreasing disposal and pursuing an efficient economic program. Considerable attention in Italy is mainly focused on material reuse, recycling and the environment, including health care protection. Energetic recovery is mainly oriented at improving the efficiency of the whole management system rather than as a priority to be pursued in waste management. Furthermore due to the high level of electrification throughout the country there is a low need for decentralized plants for satisfying the electrical needs of isolated communities. The interest in decentralized energy generation was promoted mainly as a consequence of the introduction of economic support programs for the production of renewable energy as a consequence of the broader European policy on renewable energy production. Furthermore, economic and technical aspects typical of the waste sector limited the wide diffusion of power plants in this sector compared to others such as that of biomass. In particular, for a total of > 1,550 AD facilities operating in Italy, only 46 process bio-waste. This clearly indicates the existence of technical and economic barriers.

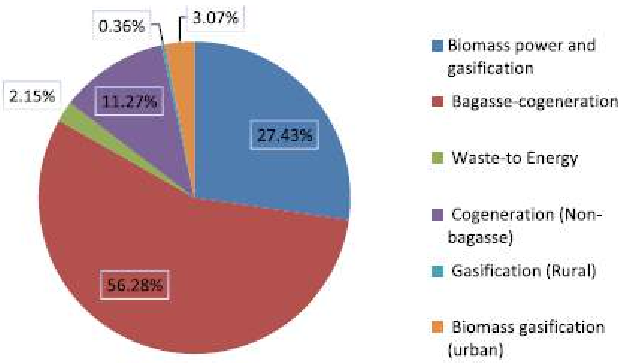


**FIGURE 3:** Source-wise break-up (%) of bioelectricity production (4.9 GW) as of 2016.

The Indian scenario is characterized by different aspects. The political and legal program for waste management is faced with important social, economic and also health care issues. The informal sector, which is mainly the extraction of recyclables from waste, gains revenues from this activity and is an important income for a large part of the population. On the other hand they operate in critical health and working conditions. Furthermore the large use of dumpsites is also another relevant risk for both the environment and public health. Implementation of a legal program and economic support by public authorities for the activity performed by the informal sector could be a first step for improving both waste management and environmental and social aspects. As largely demonstrated at the EU level for increasing both the amount of recyclables and their economic value, an efficient separation at the source is mandatory. Another important aspect concerning the Indian scenario is the need for decentralized energy production, mainly in rural areas were small-scale AD facilities also fed with bio-waste could lead to an improvement in the quality of life. Due to the full implementation of a national electrical grid, this option is of relevant interest and is also an opportunity for creating new jobs. The large presence of bio-waste in MSW generated together with economic and technical aspects appears to be a key limiting factor for the widespread implementation of waste-to-energy plants, making construction of new sanitary landfills a more attractive solution. The low quality and economic value of the resulting compost is one of the limiting factors for an efficient and larger implementation of bio-waste recycling. This is mainly a consequence of the low quality of the bio-waste that is generally collected, commingled with other waste and successively hand separated. The most recent Indian legislation (Table 5) imposes quality standards for both chemical and physical aspects to be achieved for producing organic fertilizer from bio-waste. Currently two main standards are in operation (FCO, 2009; 2013). Comparing these with the current Italian legislation (Table 5) (D. Lgs., 2010), physical features imposed by the different legislations appear quite similar with respect to the moisture content and pH values. Italian legislation imposes higher values for the total organic carbon content together with an indication of the fraction of organic nitrogen with respect to the total nitrogen. There were other main differences for the heavy metals concentration, which were, on average, significantly higher for the Indian legislation. These differences in limits can be a consequence of the features of agricultural soils between the two areas, but also of the different quality of the bio-waste processed. There were some relevant differences for the nitrogen (N) concentrations between the two Indian standards, whereas there was no indication as to the concentration of other nutrients such as P and K as reported in the Italian legislation. Finally no limits were imposed on the level of maturity achieved by the fertilizer by the Indian legislation, whereas this aspect is of particular concern in Italian legislation.

## 4. CONCLUSIONS

The present study highlighted the main differences between Italian and Indian bio-waste management. The

**TABLE 5:** Mean chemical and physical features required by Italian and Indian, legislation for classification of compost from bio-waste as organic fertilizer.

| Parameter | Italian | | Indian | | | |
|---|---|---|---|---|---|---|
| | D.Lgs. 75/2010 | | Organic Compost (FCO 2009) | | Phosphate Rich Organic Manure (FCO 2013) | |
| | **Value** | **u. m.** | **Value** | **u. m.** | **Value** | **u. m.** |
| Moisture Content | <50 | % w/w | 15.0-25.0 | %w/w max | 25 | %w/w max |
| Bulk density | | | < 1 | mg/cm3 | Bulk density 1.6 | g/cm3 |
| pH | 6.0-8.5 | - | 6.5-7.5 | | 1:5 solution) maximum6.7 | |
| TOC | >20 | % on TS | >12.0 | % w/w | >7.9 | % w/w |
| TKN | - | % on TS | % on TS | % on TS | | |
| N organic | >80% of TKN | % on TS | % on TS | % on TS | | |
| C/N | <25 | - | <20 | - | < 20:1 | - |
| Cu | <150 | ppm on TS | <1000.00 | mg/kOon TS | <1000.00 | mg/kg |
| Zn | <500 | ppm on TS | <300.00 | mg/kg | <300.00 | mg/kg |
| Pb | <140 | ppm on TS | <100.00 | mg/kg | <100.00 | mg/kg |
| Arsenic | - | - | <10.00 | mg/kg | <10.00 | mg/kg |
| Cadmium | <1.50 | mg/kgTS | <5.00 | mg/kg | <5.00 | mg/kg |
| Chromium | <0.5 | mg/kgTS | <50.00 | mg/kg | <50.00 | mg/kg |
| Mercury | <1.5 | mg/kgTS | <0.15 | mg/kg | <0.15 | mg/kg |
| Nickel | <100 | mg/kgTS | <50.00 | mg/kg | <50.00 | mg/kg |
| Total Nitrogen (as N) | - | - | >0.8 | % w/w | >0.4 | % w/w |
| Total Phosphate (as P205) | >0.5 | %TS | >0.04 | % w/w | >10.4 | % w/w |
| Total Potassium (as K20) | - | - | >0.04 | % w/w | - | % w/w |
| Colour | - | - | Dark brown to black | | Dark brown to black | |
| Odour | - | - | Absence of foul Odor | | Absence of foul Odor | |
| Particle size | - | - | Minimum 90% material should pass through 4.0 mm IS sieve | | Minimum 90% material should pass through 4.0 mm IS sieve | |
| Conductivity (as dsm-1), not more than | - | - | 4.0 | | 8.2 | |
| Germination Index | >60 | % | - | - | - | - |

*Legend: TKN = Total Kjeldal Nitrogen - TOC = Total Organic Carbon* differences can be grouped into three main categories: recycling strategies, end of waste criteria, energetic policy. Legal, social and economic aspects are also involved.

Recycling in Italy is based on the implementation of a reliable economic, legal and political supporting structure with a sustainable funding program. In India recycling is mainly performed by the informal sector with no legal and economic scheme.

Concerning end of waste criteria, Italy focuses attention on the whole process starting from the source of the bio-waste (separated collection), the performance of the processes (temperatures, length of treatment) until receiving the media and fixing quality standards for the final product. Indian legislation cares more about the receiving media by fixing quality standards for the final product.

In Italy the anaerobic digestion of bio-waste is also coherent with the renewable energy production legislation and with the EU 2020 and 2030 goals. Current legislation and economic supporting schemes has led to the construction of centralized plants. For India anaerobic digestion is a suitable solution for supplying energy and fuel, particularly in rural areas, hence promoting the adoption of decentralized facilities in which bio-waste can be co-processed with other biodegradable substrates.

On the basis of the above-reported results the following recommendations can be made.

For India: activation of an economic and legal plan for turning the informal sector into a formal organization able to ensure the same level of income for workers; implementation of a collection scheme able to increase the separation at the source of bio-waste for improving the final quality of the compost.

For Italy: implementation of legislation on the end of waste criteria for anaerobic digestion, focusing on the final quality of the product and on the receiving media; greater promotion of the organic fertilizer market for increasing the economic value of the final product.

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# FROM PILOT TO FULL SCALE OPERATION OF A WASTE-TO-PROTEIN TREATMENT FACILITY

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**ABSTRACT**
Recycling of municipal organic waste material (biowaste) still remains fairly limited especially in low and middle-income settings although this is by far the largest fraction of the municipal waste generated. A fairly novel approach is rapidly establishing itself and concerns biowaste conversion by larvae of the Black Soldier Fly (BSF), *Hermetia illucens*. The popularity of this approach links to the promising opportunities of using the harvested insect larvae as a protein source for animal feed thus providing a valuable alternative to conventional animal feed that requires a large amount of increasingly scarce resources (water and land for soya meal, and fish catch for fishmeal). The research project presented here has developed relevant and easily applicable guidance for practitioners who aim to establish a commercial facility, learning from pilot scale activities in Indonesia and then using modelling and up-scaling assumptions to highlight challenges and opportunities of larger scale implementation.

## 1. INTRODUCTION

In low- and middle-income urban settings, the organic biodegradable matter (=biowaste) constitutes the highest overall waste fraction with 50-80 percent of the total municipal waste amount generated in weight (Wilson et al. 2012). If not managed in a proper way – as is typical for many low- and middle-income settings - biowaste affects the environment and public health in a negative way. Rotting waste creates olfactory nuisance, attracts vermin and other disease transmitting agents. Furthermore, uncontained and untreated leachate from biowaste contaminates surface and groundwater supplies (Reddy and Nandini 2011). In uncontrolled disposal sites decomposing biowaste generates methane contributing to greenhouse gas emissions (Bogner et al. 2008). Management of biowaste is still fairly limited, especially in low- and middle-income settings, although a paradigm shift towards a circular economy focused on 'closing loops' through recovery, is a promising concept. Biowaste in a circular economy tackles nutrient resource scarcity or the need for renewable energy. The shift to biowaste recycling in municipal solid waste management is however limited by the potential net value typically obtained, as the costs of obtaining good quality waste are high and the revenue stream from sales of the waste derived product is either limited or complicated to ensure on a regular basis. From a research perspective however, biowaste treatment has already attracted considerable interest and many potential solutions exist (Lohri et al., 2017).

This paper focuses on a fairly novel approach of biowaste treatment based on waste conversion by insect larvae – namely using the Black Soldier Fly (BSF), *Hermetia illucens*. This approach comprises a transformation of biowaste into insect protein and insect oil. The process of converting biowaste into insect protein has already been studied by many researchers. The research has focused predominately on the biological mechanisms in the process and at lab or bench scale, for instance around the topics of mating behaviour or survival rates in the different life stages (Lohri et al., 2017). Although important, such research has limited use for practitioners. Implementation of a BSF-treatment approach for low and middle-income setting relies on practical evidence at either neighbourhood or industrial scale. It is this knowledge of operational steps, monitoring and control protocols, as well as equipment reliability and appropriateness, that helps stakeholders in waste management implement, operate and sustain a treatment facility successfully. There is some evidence that private enterprises are already investing into this technology, but they are interested in keeping a competitive edge and are quite secretive on all practical aspects of operating such a facility in a cost-effective way. This hinders an academic debate and open exchange but more importantly is a major barrier for widespread dissemination and replication. Filling this gap in open-source practical

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knowledge is the objective of a recent publication that is available for download online (www.sandec.ch) as: Black Soldier Fly Biowaste Processing - A Step by Step Guide (Dortmans et al. 2017). The paper at hand presents some of the results contained in that publication but goes a step further by also providing additional information on financial considerations and cost-revenue streams for a small scale treatment unit and then discussing implications for scaling up to a larger industrial scale.

## 2. MATERIALS AND METHODS

The particular interest of the study presented in this paper focuses on two main and strongly interlinked management aspects of a BSF waste treatment facility: 1) the operational activities and their sequencing to ensure that waste can be treated reliably and consistently every day; and 2) the associated costs involved in establishing, operating, and maintaining a BSF treatment facility of a certain scale in a given local context.

To develop the list and sequence of activities, tasks and related equipment needs, the project used an experimental approach at pilot level. Two ongoing applied research projects provided the experimental backbone resulting in the data presented here. FORWARD is a 4-year-long applied research project with a focus on integrated strategies and technologies for the management of municipal organic solid waste in medium-sized cities of Indonesia. Among other activities the project designed, implemented and operated a pilot-scale BSF waste treatment facility at a local wholesale market. It operated with an incoming vegetable and fruit waste amount varying between 0.5-1 ton per day. This BSF pilot facility allowed to derive "Standard Operating Procedures" for further dissemination (Dortmans et al, 2017) and currently also acts as a showcase and training site. The FORWARD project is funded by SECO, the Swiss State Secretariat for Economic Affairs, under a framework agreement with the Indonesian Ministry of Public Works & Housing (PU-PeRa). A second research project - SPROUT - is a three-year project with a focus on hygienic aspects, design and operation of BSF treatment units, quality of products (feed and fertilizer), post-harvest processing regarding feed quality and product safety, business models for BSF waste processing, and evaluation of the environmental impact of BSF waste processing compared to other biological treatment options. SPROUT is a multi-national project with SLU (Swedish University of Agricultural Sciences) and Eawag (Swiss Federal Institute of Aquatic Science and Technology) as main research partners and Pacovis AG from Switzerland as the partner from industry and it is funded via the EU-program ECO-INNOVERA, the Swedish Research Council Formas, the Swiss Federal Office for the Environment FOEN, and Pacovis AG.

For assessing and estimating cost-revenue aspects, we used the work measurement technique of time-motion studies applied to the facility in Indonesia (Dortmans, 2015). As labour costs are the most dominant variable costs, it was crucial to understand how much work load was utilized for which task in the treatment facility. Furthermore, the accounting method of activity-based costing (Weygandt et al., 2012) was used to structure cost types in each operational step of the facility. Activity-based costing (ABC) is a method that looks at the activities that a firm performs to finalize a product and then assigns multiple cost types and cost drivers to each defined distinctive activity unit. In the BSF case these distinctive functional activity units are those that together contribute to "treating waste and producing larvae and residue". These units were established in an experimental setting and are used to structure the standard operational procedures. Activity-based costing can help analyse the relationship between costs, activities and products. It can assign capital costs and variable costs to such units to give better insight on how these units compare to each other, and where optimization potential is highest for instance by introducing a certain automation and machinery. In ABC indirect costs, linked to management and office staff for instance, are sometimes difficult to attribute to a unit. That is why this method is particularly relevant in the product manufacturing sector. In the BSF case the products being "manufactured" are the larvae and the residue. Depending on how costs respond to a change in a business activity, they can be classified as fixed, variable, or mixed costs (Weygandt et al., 2012). Fixed Costs, as the name implies, do not change regardless of the activity level. They are independent and remain the same even if there is an increase or decrease in production, but only as long as the production does not require additional machinery. Examples of fixed costs are capital costs such as construction and land costs or periodic fixed costs like depreciation, rent, insurance, and cost of capital. Variable costs are linked to the activity level and thus change directly and proportionally with the goods or services produced. Typically, direct material, direct labour, and water or electricity can be categorized under such costs. Mixed Costs consist of both fixed and variable costs. These costs will change with the activity level but not proportionally with the goods or services produced. Mixed costs are not relevant for production facilities of this type and were neglected in our study. A framework of modelling the costs was developed based on defined units and using excel spreadsheet software. Besides the primary data sources from the experimental sites, secondary data sources were collected from literature. For specific equipment, also quotes and tenders from the industry were obtained and included. Two scenarios were then analysed for the geographical context of Indonesia, a 1 ton and a 60 ton per day incoming waste facility.

## 3. RESULTS AND DISCUSSION

### 3.1 Operational aspects of Black Soldier Fly waste treatment

The Black Soldier Fly (BSF), *Hermetia illucens* is of the dipteran family Stratiomyidae and can be encountered worldwide between the latitudes of 40°S and 45°N. The female fly lays eggs close to decomposing organic matter and into small, dry, sheltered cavities. On average, the eggs hatch after 4 days and the emerged larvae, which are barely a few millimetres in size, will search for food and start feeding on the organic waste nearby. The larvae then feed on decomposing organic matter to grow from a few millimetres size to around 2.5 cm length and 0.5 cm width. Under optimal conditions and ideal food quality and quantity, the growth of larvae will require a minimum period of 15-17 days. BSF larvae however have the ability to extend their life cycle if conditions are unfavourable making it a very resilient organism. When reaching the final larval stage, as prepupa, it moves out and away from the food source and to a dry, shaded and protected environment for pupation. Pupation takes around 2-3 weeks and ends when the fly emerges from its pupa shell. After emerging, the fly lives for about one week. As a fly, BSF do not feed, but will only search for a partner to copulate and then lay eggs. In this life stage, natural light and a warm temperature (25-32°C) is required.

For a BSF processing facility the goal is to optimize this natural life cycle using engineering principles. Based on this concept, we suggest to differentiate distinct processing units as described below and shown in Figure 1.

The BSF rearing unit ensures that a reliable, consistent and sufficient amount of small larvae is continuously available to inoculate the daily amount of biowaste received for processing. One fraction of the larvae is kept in the rearing unit and bred in a controlled manner to ensure a stable population. A well-engineered BSF nursery monitors the survival rates at every step in this cycle and keeps track of the colony's overall performance. With this information, it controls the number of prepupae that are allowed to pupate which allows control of the number of flies that emerge, as well as egg packages deposited, and number of larvae hatchings. In the rearing unit two type of cages can be distinguished: the "dark cage" and the "love cage". Pupation containers with moist soil-like substrate (matured compost) are placed inside a "dark cage". The dark environment, provides the pupae with sufficient protection from the changing outside environmental conditions (moisture, temperature, movement of air, etc.). Due to the darkness inside the cage, the emerged flies will remain motionless. Emerged flies are collected from the dark cage by connecting this dark cage to another cage, which is not darkened, through a tunnel. A light source set at the end of the tunnel will attract the flies to fly from the dark cage into this undarkened "love cage". The love cages each contain a box with a smelly substrate attracting the flies to induce egg laying and a media for laying eggs into, as well as a wet cloth to allow the flies to hydrate. The love cages are then exposed to (indirect) sunlight to stimulate mating. Eggs deposited are harvested and deriving hatchlings are fed for five days in a controlled environment before they can be put into the BSF waste treatment unit.

In the waste pre-processing unit the quality of the incoming waste is controlled to avoid hazardous or inorganic substances. Then a reduction of the waste particle size follows. Particles of smaller than 1-2 cm in diameter are ideal as BSF larvae do not have appropriate mouthparts to break apart larger waste pieces. This helps to speed-up the BSF processing time. In the case of high water content in waste (moisture of 70-80% is best for the larvae), dewatering may be necessary whereby blending different types of biowaste with complementing (lower) moisture content is also an option.

In the BSF waste treatment unit a part of the waste is transferred to empty containers (so-called larveros) where small larvae from the rearing unit are added. The other part of the waste is added to existing larvero containers with already larger larvae. As a rule of thumb, we work with 10,000-12,000 "five-day-old-larvae" (5-DOL) added into one larvero container with a size of 40x60x17 cm. While the larvae feed and grow, more waste is added to the same larvero container on day 5 and again on day 8, adding up to a total of 15 kg of fresh waste per larvero. The larvae are harvested after 12 days. The individual larvero containers can be stacked on top of each other to maximize the use of the space available, however it is important to leave space between the containers to allow sufficient air ventilation (Figure 2).

Finally, in the processing and refining unit, larvae and residue can be further processed depending on local market demand. They may be sold alive to customers (e.g. reptile farms or bird market) or processed to feed pellets to establish a blend, which meets the nutritive requirements of the targeted animal (broiler chicken, layer hen, different

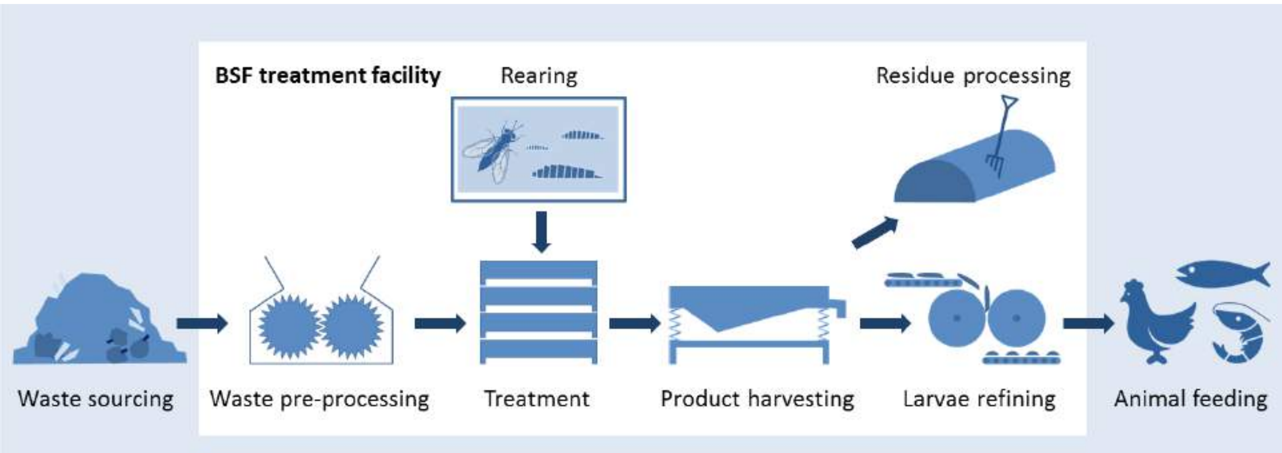


**FIGURE 1:** The different units of a BSF treatment system (Dortmans et al, 2017).

**FIGURE 2:** Stack of larvero containers with ventilation frames in-between levels (Dortmans et al, 2017).

fish species). In most cases, larvae will need some form of post-processing to ensure they can be sanitized, stored and transported easily to the respective customers. Sanitising involves killing off any bacteria, which might be adhering to the larvae skin and ensuring that the larvae empty their gut which contains only partly digested residue. We recommend dipping the larvae into boiling water as this kills them instantly and sanitizes the product. A storable product requires a water and fat content of below 10%. Sun drying is suitable to reduce water content and the fat content can be reduced using an oil press. The waste residue has similar characteristics as immature compost and therefore requires post-processing. Composting the residue is the simplest approach, but an alternative is vermicomposting or adding it into an anaerobic digester.

### 3.2 Overall siting considerations

The following points must be taken into consideration when selecting an appropriate site for a BSF processing facility:

- Water and electricity supply and wastewater management options should be available.
- Environmental buffers that separate the facility from the surroundings should be ensured (e.g., open areas, trees, fences) as visual barrier as well as to minimize impact of the slight odour emission
- Sufficient, regular and predictable amounts of fresh biowaste (source segregated) should be available at lowest cost possible. Although this is not discussed in further detail in the paper, this aspect is crucial. For the city waste manager that faces the challenge of large amounts of household waste and is in search for a treatment solution, household segregation is a key determinant to obtain good quality organic waste for BSF treatment. This is not a simple endeavour. On the other hand, an easier way to start with BSF waste treatment is to target business and industrial enterprises that generate large amounts of already quite "pure" organic waste for instance slaughterhouses, breweries, agroindustry, supermarkets, vegetable and fruit markets or restaurants.
- A closed and ventilated room is required for the rearing unit but also sunlight is required to ensure mating of flies. A sheltered area without direct sunlight is needed for the treatment containers as well as office, lab space, toilet and hygiene facilities.

### 3.3 Costing considerations

A BSF-treatment facility can produce 200 kg (wet weight) of grown larvae per ton of incoming waste processed. Results of the financial analysis show that total annual costs, assuming one ton of processed waste per day and an average equipment depreciation of 3 years, amounts to around 16,700 €. The land costs and building construction was however not included in this calculation, as for the case of Indonesia the land and building were existing and provided. All activities related to the rearing unit amount to up 31% of these costs. Labour costs amount to 45% of the total costs whereby other variable costs such as electricity, water and chicken feed for the nursery only amount to about 12% of the total costs (Table 1). This scenario reflects a situation where all machinery is kept to the minimum requirement and where all processes rely as much as possible on manual labour (1 person in the rearing unit and 2 persons for all the other units). One aspect to consider is the maximum operating capacity of the equipment. In other words, much of the equipment is designed for larger capacity and at these small scales of one ton of waste per day would be utilized under capacity therefore increasing the production costs. Another aspect to consider is that the labour requirements at the rearing unit are least dependant on scale of production. With a similar setup at the rearing unit, the capacity can easily be increased to around 5 ton/day without much change in cost.

With an increase in scale to a 60 ton/day facility, both labour and equipment cost will obviously increase. Here some economies of scale come into effect. For instance when analysing labour requirement per ton of waste, a significant reduction from 6 staff per ton to 0.58 staff per ton can be shown. Given the low unskilled labour costs in In-

**TABLE 1:** Unit costs for the rearing unit and other units based on a 1 ton/day BSF facility.

| **Activity Unit** | **Euro/Year** 1 t/day | **% of total** |
|---|---|---|
| Rearing units | | 31% |
| • Labour | 2,483 | |
| • Consumables | 1,095 | |
| • Annual equipment costs | 1,526 | |
| Treatment units | | 56% |
| • Labour | 4,966 | |
| • Consumables | 881 | |
| • Annual equipment costs | 3,545 | |
| Indirect costs | 2,174 | 13% |
| **Total** | **16,670** | **100%** |
| Labour | | 45% |
| Consumables | | 12% |
| Equipment | | 30% |

donesia however, the impact of this on total costs is lower than in a high labour cost situation such as Switzerland. Furthermore, technology and equipment cost, which will be unavoidable at this larger scale, do not vary much between Indonesia and a high-income country. The ratio of equipment to labour cost will thus increase in the large-scale facility and this will increase dependency on skilled operators, skilled maintenance and availability of spare parts. All these aspects are well known as typical barriers that reduce the technical feasibility of a facility in low and middle-income countries.

Regarding revenue sources, reliable data is not yet available as markets are still novel and price not well established. Furthermore, the small amounts of larvae currently produced limits the potential of market exploration and at this stage only caters to specialized niche markets. Based on current sales revenues of 0.35 € per kg of dry BSF larvae could be expected.

## 4. CONCLUSIONS

The structure of activity units in a BSF waste treatment facility is helpful not only for defining standard operating procedures and material lists but also to better assess the costs associate with different tasks. The cost analysis of the BSF production gives important insights concerning the economic feasibility of this technology. Closely assessing the different components has made it obvious that certain production steps include sizing issues when being brought to a larger scale. These steps are namely the rearing unit and the treatment unit. Insights from professional process analysts could be of additional value as upscaling opportunities are not yet fully exploited and different sets, sizes and arrangements of the machinery could significantly decrease costs. In general, cost behaviour is always on the side of the economies of scale of bigger facilities. The conducted calculations emphasize most opportunities and drawbacks and lay open where and when costs do not behave proportionally. Besides showing upscaling benefits in most areas, the legal uncertainties and inflated capital costs connected to a 60 tonnes BSF facility might still shy away public and private stakeholders from investing in such a project. The changes in legislations and common perception of insect meal as a source of protein are important determinants of the future success of the BSF technology. Although the cost difference may not justify the administrative and logistic pitfalls included in producing in Indonesia and other countries in the region might be even more attractive financially, the investment seems equitable, regardless, if the financial incentive of producing BSF larvae meal is secondary and the purpose of the facility focuses primarily on adequate waste management.


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# INNOVATIVE TECHNOLOGIES FOR PHOSPHORUS RECOVERY FROM SEWAGE SLUDGE ASH

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**ABSTRACT**

The scarcity in Europe of phosphate ore along with the constantly growing demand for phosphorus-based products make it essential to find new sources and innovative recovery techniques for phosphorus in all of its forms. In order to avoid phosphate rock reserves exhaustion, its recovery from incineration Sewage Sludge Ash (SSA) might be a solution. Phosphorus concentration in municipal SSA is 9% (Krüger et al., 2014), which is within the range of the currently mined phosphate rock. However, the high amount of metallic elements (especially iron and aluminium) leads to a higher consumption of concentrated sulphuric acid, as it is used for the phosphate mineral treatment. The aim of this preliminary survey is to assess the acid demand and the efficiency of different acids towards the dissolution of the phosphate minerals in ash. Elemental and mineral composition, leachability and further tests were performed using four different SSA samples originated from three different sewage sludge incinerators located in Germany. First results show that the extraction yields with organic acids are higher compared to the ones achieved with mineral acids. Especially for oxalic acid, for which dissolution occurs both due to protonation and reduction, extraction rates close to 100% were achieved using lower amounts of acid.

## 1. INTRODUCTION

The phosphate rock processing trend shows a steady increase in the last years (Figure 1). In addition, the forecasts concerning the fertilizers demand published by the Food and Agriculture Organization of the United Nations emphasises that the global demand is constantly increasing (Table 1). It is important to associate the fertilizer utilisation with the phosphate rock mining industry, since it is estimated that 82% of the produced phosphorus is used for the production of fertilizers (IFCD, 2010).

Due to growth in phosphate rock extraction and demand, it is legitimate to question when this trend will ultimately be sustainable, since phosphate rock is a non-renewable resource and it cannot be substituted in agriculture (IFCD, 2010). Some authors have previously reported alarming data concerning phosphate rock depletion in the near future:

- 50 - 100 years (Atienza et al., 2014)
- A few decades (Biswas et al., 2014)
- 60 - 125 years (Franz, 2007)
- 370 years (Cooper et al., 2011)

However, it should be noted that these data are only related to the reserves of phosphates, and not to resources.

According to the United Stated Geological Survey (USGS), classification for phosphate rock, reserves are a subset of resources that meet specified minimum physical and chemical criteria related to current mining and production practices, including those for grade, quality, thickness, and depth. In contrast, resources are classed as the concentrations of materials in a form and quantity that can be extracted currently. Extraction itself might be performed after a detailed evaluation of the deposit with the aim to prove that it satisfies the minimum requirement for its exploitation.

According to the USGS (USGS; 2016), the identified reserves (69 million Gg), which are at the current extraction rate of 224 thousand Gg/a, are expected to be depleted in approximately 300 years. However, considering the resources, which amount to more than 300 million Gg, the depletion time is shifted to more than 1,300 years from now.

Therefore, theoretically the amount of phosphate rock supply is not a global issue in the near future. For Europe, the situation could be different. Evidence of this is the inclusion first of phosphate rock in 2013, and later of phosphorus in 2017 within the Critical Raw Materials (CRMs) list. The issue concerning the choice of the European Com-

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**TABLE 1:** World demand forecasts for fertilizer nutrients, 2014-2018 (FAO, 2015).

| Year | 2014 | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|---|
| $P_2O_5$ demand (in thousand Mg) | 42,706 | 43,803 | 44,740 | 45,718 | 46,648 |

munity to include phosphate rock in the CMRs list resides in the fact that Europe imports this resource almost completely. The main exporters worldwide are three countries (China, United States and Morocco) and actually, there is not a relevant phosphorus recycling process (European Commission, 2014).

Hence, the basis of research should not be the scarcity of raw materials, but rather the opportunity to reduce phosphorus waste whenever it is possible.

## 2. PHOSPHATE ORE AND SEWAGE SLUDGE ASH

Table 2 shows the average content of the extracted phosphate rock worldwide. The elemental phosphorus content in ore is calculated using the following equation (European Parliament, 2003):

$$Phosphorus\ (P) = Phosphorus\ Pentoxide\ (P_2O_5) * 0.436$$

The first four producer countries (China, United States, Morocco and Russia) share 78% of the global phosphate ore production. Considering only the main producers, the average content is 31.8% $P_2O_5$, ranging between 28.5-36.8% (average data for the years 2009 – 2013) (USGS, 2016).

Similar values are also reported from the International Fertilizer Development Center (IFDC), where commercial phosphate rock shows a P2O5 content from less than 25% to over 37%, (10.9-16.13% P) (IFCD, 2010).

As a comparison, German municipal sewage sludge ash has a lower phosphorus content (9%, see table 3) than the one of extracted phosphate rock, but still within the world

**TABLE 3:** Elemental mass fractions in German SSA (Krüger et al., 2014).

| Element | Min | Max | Mean | Median | Mass flow [Mg/a] |
|---|---|---|---|---|---|
| P | 1.5% | 13.1% | 7.3% | 7.9% | 18,812 |
| P (Municipal) | 3.6% | 13.1% | 9.0% | 9.1% | 10,939 |
| P (Mun. / Ind.) | 2.8% | 7.5% | 4.9% | 4.8% | 7,319 |
| P (Industrial) | 1.5% | 3.8% | 2.3% | 2.3% | 554 |
| Ca | 6.1% | 37.8% | 13.8% | 10.5% | 42,669 |
| Si | 2.4% | 23.7% | 12.1% | 12.1% | 38,637 |
| Fe | 1.8% | 20.3% | 9.9% | 9.5% | 29,049 |
| Al | 0.7% | 20.2% | 5.2% | 4.8% | 14,999 |
| S | 0.3% | 6.9% | 1.5% | 1.0% | 6,028 |
| Mg | 0.3% | 3.9% | 1.4% | 1.3% | 4,061 |
| Na | 0.2% | 2.6% | 0.7% | 0.6% | 2,416 |
| K | 0.0% | 1.7% | 0.9% | 0.9% | 2,227 |
| Ti | 0.1% | 1.5% | 0.4% | 0.4% | 1,264 |

**TABLE 2:** Phosphorus pentoxide and elemental phosphorus content in phosphate ore (USGS, 2013).

| Compound | Min | Average | Max |
|---|---|---|---|
| $P_2O_5$ | 11.00% | 30.41% | 46.00% |
| P (elemental) | 4.80% | 13.26% | 20.06% |

range for extracted ore (USGS, 2013).

According to the IFDC report, phosphate ore has to meet specific metal content requirements in order to be processed. For this, two ratios are defined (IFCD, 2010):

$$M_2O_3=(Fe_2O_3+Al_2O_3) \div P_2O_5 < 0.10 \qquad (1)$$

$$M_2O_3 + (MgO \div P_2O_5) < 0.12 \qquad (2)$$

Referring to the values highlighted in the survey performed by Krüger et al. (2014) for Germany, it can be seen that the two ratios for SSA vary in comparison to the indications of the IFDC study, resulting of 1.43 and 1.57 for (1) and (2) respectively.

Another recycling route could be the direct application of SSA in soil as fertilizer. In this case, issues come from the fact that the European Directive 87/278/CEE establishes limit values for the concentration and annual load for specific elements, which often are exceeded in SSA. Moreover, national legislations have stricter limits that hinder the reuse of ash without pretreatment. Table 4 uses the situation in Germany as an example.

Furthermore, it has to be considered that phosphorus contained in SSA is not plant available (BMEL, 2011; Petzet et al., 2012; Ottosen et al., 2013) due to its strong bonds in ash minerals.

## 3. MATERIALS AND METHODS

### 3.1 General

The aim of this work is the evaluation of the phosphorus extraction yields from sewage sludge ash via wet processes. In this study, the extraction of phosphorus is performed dissolving ashes from sewage sludge incineration.

The dissolution of the mineral phases present in sewage sludge ash were investigated using different inorganic and organic acids, such as nitric, sulphuric oxalic, citric and lactic acid. Tests were performed on 4 different sewage sludge ash samples obtained by 3 different incineration plants located in Germany.

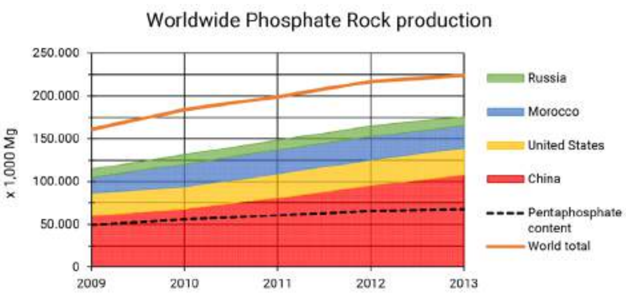


**FIGURE 1:** Leading producers of phosphate, and the total rate of worldwide production (USGS, 2013).

**TABLE 4:** Heavy metals concentrations in German SSA (Krüger et al., 2014) and comparison with limits from different German ordinances (BMEL, 2001).

| Element | Min [mg/kg] | Max mg/kg] | Mean mg/kg] | Median [mg/kg] | Mass flow [mg/kg] | Limit [mg/kg] | Ordinance |
|---|---|---|---|---|---|---|---|
| Pb | 3.5 | 1,112 | 151 | 117 | 62 | 150 | AbfKlärV |
| Cd | 0.1 | 80,3 | 3.3 | 2.7 | 1.4 | 1,5 | BioAbfV |
| Cr | 58 | 1,502 | 267 | 160 | 107 | 2 | DümV |
| Cu | 162 | 3,467 | 916 | 785 | 395 | 100 | BioAbfV |
| Ni | 8.2 | 501 | 105.8 | 74.8 | 58 | 50 | BioAbfV |
| Hg | 0.1 | 3.6 | 0.8 | 0.5 | 0.3 | 1 | BioAbfV |
| Zn | 552 | 5,515 | 2,535 | 2,534 | 763 | 400 | BioAbfV |
| As | 4.2 | 124 | 17.5 | 13.6 | 6.7 | 40 | DümV |

First, elemental and mineral composition of the ash were investigated in order to evaluate the extraction yields achieved in the following leaching tests. These tests include aqua regia dissolution of the sample and characterisation via X-rays diffraction technique.

Once the elemental composition of each sample was deciphered, a standardised leaching test with nitric acid was performed at 16 different pH values. The aim of this was to obtain a baseline standard for the acid requirements and to observe indications concerning the operational pH values to perform additional experiments. In particular, further leaching tests with sulphuric, oxalic citric and lactic acid were conducted at values of liquid-to-solid ratios and pH values suggested by the previous analyses.

The overall result obtained pursuing this approach is a preliminary understanding of the parameters affecting the phosphorus extraction yield which will be used as a basis for further research on the topic.

The following paragraphs present the methodology applied in order to evaluate the physical-chemical characterisation of samples, pH-dependency of the sewage sludge ash dissolution and the phosphorus extraction carried out with different inorganic and organic acids.

### 3.2 Elemental and mineral composition

In order to evaluate the elemental composition of the ash, water content and organic dry matter; samples were oven dried at 105 °C according to DIN EN 13346 and DIN EN 15935 respectively.

Afterwards, 1 g of each sample was treated for 10 min at 175 °C in triplicates via microwave assisted aqua regia digestion (CEM Corporation MARS 6 microwave digestion system) according to DIN EN 16174.

At the end of the digestion, samples were vacuum filtered at 0.45 µm and properly diluted prior to analysis via ICP-OES. (Agilent 5100 ICP-OES) according to DIN EN ISO 11885. Furthermore, in order to recognise the mineral species, X-rays diffraction analysis was performed for each sample to detect the main crystalline phases contained in the ashes. For this purpose, 1 g of milled sample was prepared and analysed with a Siemens D500 XRD system.

### 3.3 Sequential extraction

The leaching behaviour of sample SSA-1 was tested in order to evaluate the acid requirements for the successive extractions according to DIN EN 14429 norm. For this, 16 individual experiments were carried out to cover a pH range between 0 and 7.56. Glass bottles of 500 ml volume were used to perform the extraction.

Extracting solutions were prepared by adding correct amounts of pure nitric acid in demineralised water for a total volume of 300 ml. For each bottle, the total amount of extracting solution was added to 30 g of dried sample, in three steps within the first 2 hours of the experiment, for a final L/S ratio of 10 ml/g. The experiment duration was 48 h in order to reach the chemical equilibrium. The pH values were monitored according to the norm identified above.

### 3.4 Leaching tests

Phosphorus extraction tests were performed for all samples with 0.4 M solutions of sulphuric, oxalic, citric and lactic acid. While sulphuric acid application is justified by its strength and low cost, organic acids were selected for their reduction properties (especially oxalic acid) and for their environment-friendly production (citric and lactic acid).

In order to simulate different conditions, the amount of sample for each test was varied, to keep the molarity of the solutions constant. This translates into a change in the specific amount of acid per mass of ash (or mol H+/g ash) allowing the utilisation of limited quantities of chemicals (especially for organic acids). Therefore, the main parameters controlled during the leaching were the L/S ratio and the pH variation of the solution. Liquid-to-solid ratios of 10, 20 and 40 ml/g were obtained through mixing 100 ml solution with 10, 5 and 2.5 grams of ash respectively.

The leaching was performed in sealed, stirred bottles for 1 hour at an ambient temperature. Afterwards, samples were filtered at 0.45 µm and analysed with ICP-OES system.

## 4. RESULTS AND DISCUSSION

### 4.1 Elemental and mineral composition

*4.1.1 Elemental composition*

The water content and the organic dry matter analysis for samples n°1 and 4 is reported in table 5. Samples n°2 and 3 were received in a dry state.

Table 6 shows the results of the ICP-OES analysis. As it can be seen, SSA-1 and SSA-4 clearly arise from incinera-

**TABLE 5:** Water content and organic dry matter.

| | SSA-1 | SSA-2 | SSA-3 | SSA-4 |
|---|---|---|---|---|
| Water content | 20.95% | 0.00% | 0.00% | 19.93% |
| Organic dry matter | 0.73% | 1.33% | 1.89% | 4.93% |

**TABLE 6:** Elemental composition of sewage sludge ash.

| Element | | SSA-1 | SSA-2 | SSA-3 | SSA-4 |
|---|---|---|---|---|---|
| P | Phosphorus | 9.31% | 4.50% | 10.25% | 8.99% |
| Fe | Iron | 14.38% | 6.23% | 1.83% | 10.14% |
| Ca | Calcium | 9.27% | 5.88% | 9.00% | 10.32% |
| Al | Aluminum | 3.10% | 5.21% | 12.67% | 4.63% |
| Mg | Magnesium | 1.02% | 0.82% | 0.82% | 1.35% |
| K | Potassium | 0.91% | 1.04% | 1.28% | 0.80% |
| S | Sulphur | 0.59% | 0.67% | 0.31% | 0.62% |
| Na | Sodium | 0.37% | 0.29% | 0.33% | 0.32% |
| Main elements (Total) | | 38.95% | 24.64% | 36.49% | 37.17% |
| Balance | | 61.05% | 75.65% | 63.51% | 62.83% |

tion of sludge where iron precipitants are used, while SSA-3 arises from sludge with aluminium salts as precipitants. SSA-2 shows a comparable concentration of iron and aluminium. Furthermore, this sample also showed a total mass of primary elements of 24.7%, compared to 36-39% found in the other samples. This can be explained by less silicon being dissolved by aqua regia and clearly detected via X-ray diffraction analysis.

*4.1.2 Mineral phases*

Results show that the main crystallised minerals in the sample SSA-1 and SSA-4 are quartz ($SiO_2$), hematite ($Fe_2O_3$) and calcium phosphates with substitutions of iron ($Ca_9Fe(PO_4)_7$) and magnesium (whitlockite, $Ca_{18}Mg_2H_2(PO_4)_{14}$), in line with the findings of Petzet et al. (2012), Donatello and Cheeseman (2013). Analysis on sample SSA-2 detects a predominance of a quartzose component, followed by calcium iron phosphates and aluminium phosphates ($AlPO_4$). SSA-3 shows besides quartz, also the presence of aluminium phosphates and whitlockite.

### 4.2 Sequential extractions

The leaching characteristic is shown in Figure 2. It can be seen that calcium and magnesium are leached simultaneously at a pH close to neutral. Starting from pH below 4.5 it can be observed that Ca and P concentrations increase, whereas calcium phosphates are also dissolved. This can be demonstrated by the increase of the phosphorus concentration in solution. Additionally, it can be observed that aluminium is participating as substitution for magnesium in Ca-Al-P crystals.

The leaching of iron starts at pH values close to 1.8. Therefore, in order to avoid the dissolution of hematite and other heavy metals (released simultaneously with iron (Franz, 2008)), a pH higher than 1.8 should be used. Under this condition, it is possible to extract 70% of the total phosphorus contained in the ash, limiting the amount of impurities in the leachate. Considering the acid demand, this corresponds to a need of 0.004-0.007 mol H+/g ash for respective phosphorus extraction rates of 68.8-97.5%, which is in line with the acid demand according to Petzet et al., (2012).

### 4.3 Leaching tests

For all the samples, the higher extraction yields were achieved for oxalic acid, followed by sulphuric acid. The effectiveness of citric acid was proven for high L/S ratios, while due to the limited strength of the solutions lactic acid was least effective.

As displayed in figure 3, extraction yields over 90% were achieved for SSA-1. SSA-2 and SSA-3 using oxalic acid, which shows a higher effectiveness in phosphate solubilisation. On the other hand, SSA-4 showed the lowest extraction yield; this is most likely to be caused by its higher organic dry matter content (4.93%).

For sample SSA-1, values from the different extractions above described and from the sequential extraction performed with nitric acid were compared.

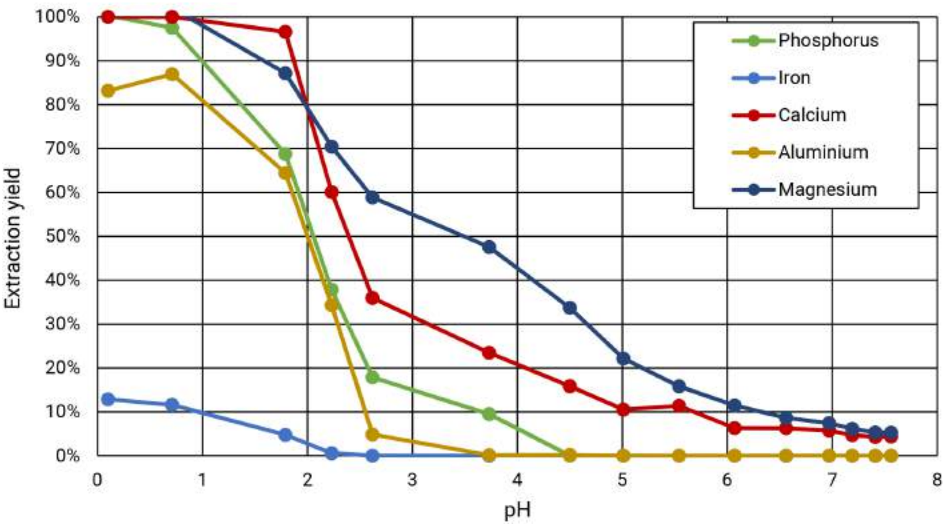


**FIGURE 2:** Leaching behaviour with nitric acid at different pH values for SSA-1.

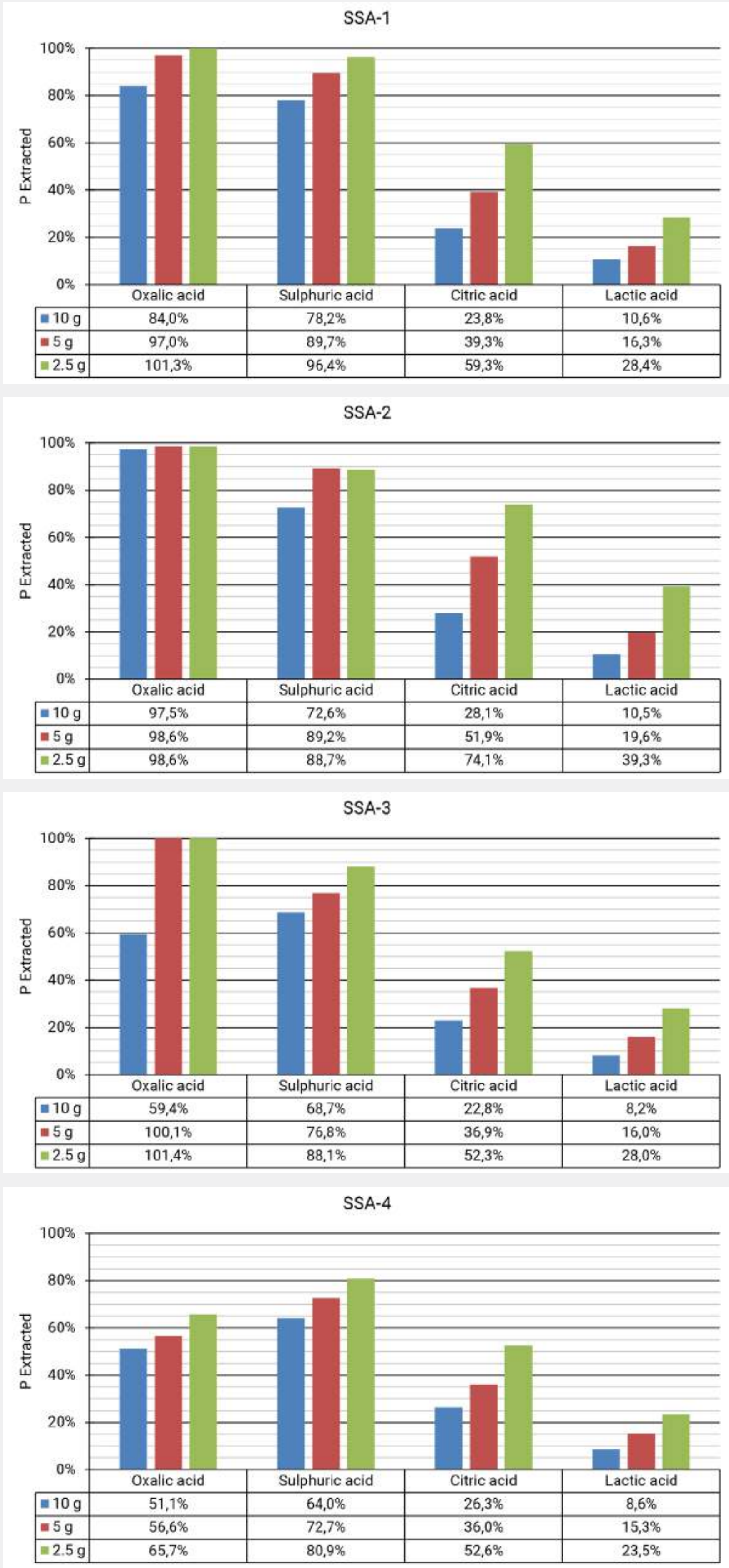


**FIGURE 3:** Leaching tests performed with different acids at different L/S ratios.

Figure 4 shows the acid demand in mmol H+/g SSA. The amount of protons required for the phosphorus solubilisation for nitric and sulphuric acid is comparable, even if performances of this last one are slightly lower, probably due to a re-precipitation of gypsum in the ash surface (from the reaction of Ca with $H_2SO_4$ (Ottosen et al., 2013)). Moreover, for oxalic, citric and lactic acid higher extraction rates are achieved in comparison with nitric acid even at low mmol H+/g SSA values. This is caused not only by the low pH of the solution, but also for reduction potential of organic acids, which promotes the dissolution of Ca-Fe-P/Ca-Mg-P minerals. This fact can clearly be seen in Ca and Fe concentrations in the lactic and citric acid solutions in Figure 5 and Figure 6.

## 5. CONCLUSIONS

In this work the wet phosphorus leaching from sewage sludge ash was investigated. The elemental composition of ash evaluated after aqua regia dissolution of samples showed a phosphorus content in the samples investigated

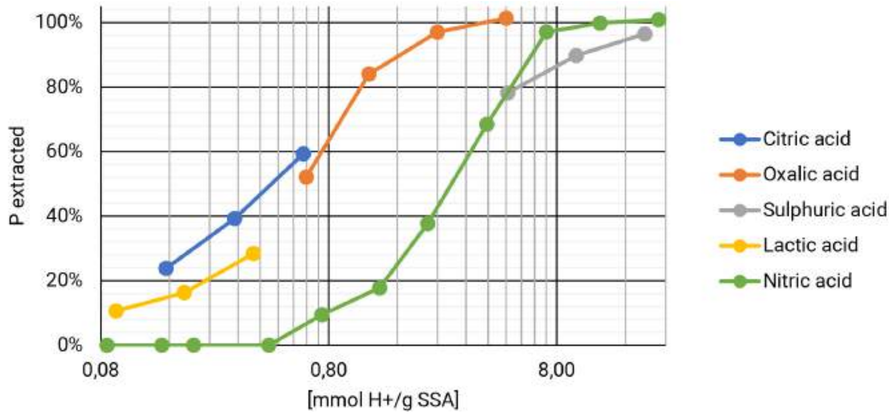


**FIGURE 4:** Phosphorus extraction yields for SSA-1.

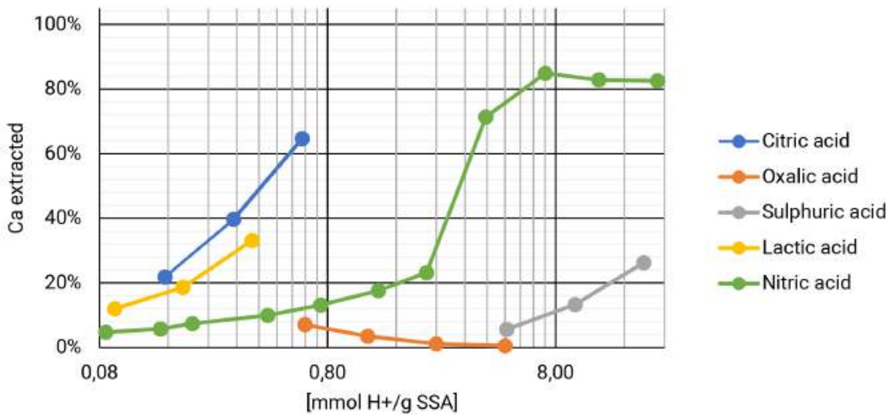


**FIGURE 5:** Calcium extraction yields for SSA-1.

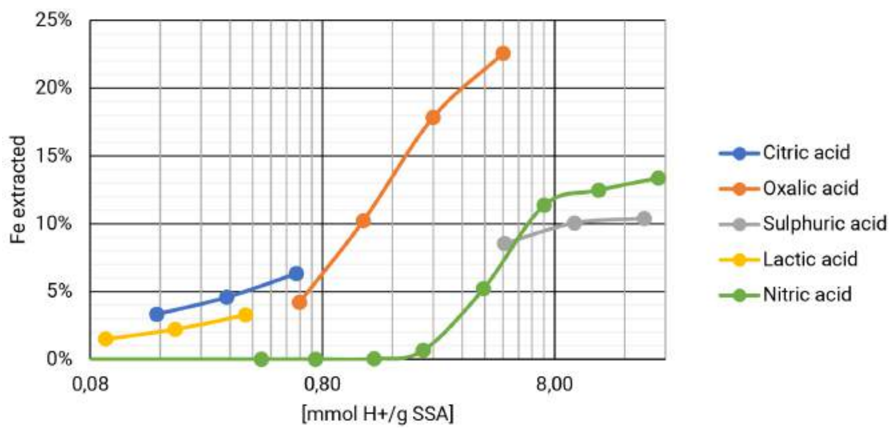


**FIGURE 6:** Iron extraction yields for SSA-1.

ranging between 4.50% and 10.25%. Other main elements of the ash were iron, calcium, and aluminium, which reflect the mineral composition of the samples. In fact, phosphorus is bonded in calcium phosphates in which iron, magnesium or other minor elements are substitutes in the mineral phases. Aluminium phosphates were found to be the main mineral phase in ash arising from wastewater treatment plants that use aluminium salts as precipitant agents.

The leaching performed under standard conditions using nitric acid showed that in order to achieve phosphorus extraction yields close to 100%, a specific amount of at least 7 mmol H+/g SSA is necessary. Furthermore, leaching with nitric acid shows that the extraction of phosphorus closely follows the solubilisation of calcium. This confirms the connection of the two elements already determined via X-rays diffraction. Therefore, the dissolution of calcium phosphates is the main process to consider while recovering phosphorus via wet acid processes.

Using these preliminary information as basis, further leaching tests were performed using different acids While using sulphuric acid for the leaching, similar phosphorus extraction yields were obtained though performances were slightly lower than if using nitric acid. This could be explained by the successive precipitation of gypsum on the ash surface, which may obstruct further dissolution of phosphorus-containing minerals.

Different conclusions can be drawn concerning the use of organic acids. Oxalic acid achieved the best extraction yields in this work. The use of oxalic acid allows extraction rates close to 100% to be achieved with 1 mmol H+/g SSA, compared to values higher than 8 mmol H+/g SSA using nitric and sulphuric acid

The main difference in the leachate obtained using oxalic acid was the lower concentration of calcium. This is probably due to its precipitation as oxalate salt. This information could be useful for the context of further research focused on the precipitation of phosphorus in other forms rather than calcium phosphates. However, an increased solubilisation of iron was also observed, which should be investigated in more detail when assessing successive recovery steps.

The effectiveness of organic acids for the dissolution of SSA was also proven for citric and lactic acid. In these cases, the overall yield was sensibly lower than the one achieved for oxalic and sulfuric acid. Nevertheless, extraction yields for citric and lactic acid were higher than the ones achieved using nitric acid in standard conditions. This was explained by the fact that the extraction with organic acids is principally given from reduction at even higher pH values, and not from the protonation of the solution. For citric acid in particular, investigations on leaching at higher temperatures might bring an increase in phosphorus extraction yields.

In conclusion, further investigations will be focused on the experimentation through the application of additional leaching conditions. In particular, higher solution molarities with lower L/S ratios may lead to similar extraction yields, but also to an increased overall concentration of phosphorus in solution due to the higher quantity of ash available for leaching.

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# CHARACTERISTICS AND ASSESSMENT OF THE SUITABILITY OF COAL-RICH MINING WASTE IN THE PRODUCTION OF CERAMIC BUILDING MATERIALS

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**ABSTRACT**

Hard coal mining involves the generation of large quantities of waste materials. For every 1000 kg of hard coal produced in Poland an average of approx. 400 kg of different kinds of waste material is generated. Environmental regulations and new approaches based on the Clean Coal Technologies require manufacturers to recover or neutralize the generated wastes. Technological progress and the development of clean coal technologies have led to the wastes being treated as products for other industries. The ability to use the waste material depends on their chemical and mineral composition. Mining wastes are characterized by different properties depending on the stage of the coal-enrichment process in which they are formed. In spite of ever more efficient coal-enrichment technologies, in addition to mineral material, there is still a significant amount of carbon in the wastes that can be recovered in various ways. After the coal recovery process, the waste material is wet (20-30%), has a thick pasty consistency and is deposited in landfills, thus representing a hazard and nuisance to the environment.

This study focuses on the characteristics of the raw waste material and the possibility of using it in the production of building materials. Mineral compositions were tested with XRD and STA methods and the characteristics of thermal properties of waste material were investigated using STA and heating microscopy. Gases released during thermal processing of the waste material were analysed by the EGA / MS method. Particle size distribution was measured and the specific surface area of the waste mud samples was calculated. Preliminary analysis was carried out to assess the feasibility of using this waste as a raw material in the production of fired ceramic building materials.

## 1. INTRODUCTION

Hard coal mining is associated with the generation of significant quantities of wastes. The mining industry is one of the highest generators of wastes in Poland, where for every 1000 kg of hard coal produced approximately 400 kg of different kinds of wastes are generated. In the years 1998-2008, the majority of mining waste in Poland was generated by the exploitation of hard coal and accounted for 47% to 52% of total produced extractive wastes. Over the period analysed the amount of coal mining waste ranged from approx. 30 million Mg to 40 ml Mg per year. According to the Polish Central Statistical Office (2009) (GUS), at the end of 2008 approx. 600 million Mg of waste from hard coal mining was deposited in Poland (accumulated in landfills, dumps, sediment ponds), as the result of several hundred years of operation of this industry in Upper and Lower Silesia. In the past, due to limited possibilities of reuse, the vast majority of mining wastes were deposited on the ground in different kinds of mining waste landfills. The situation changed in the second half of the 1990s following economic changes and new regulations that resulted in the waste quantities used economically exceeding the amount of wastes directed to landfill.

Currently, in the formal and legal sense, approximately as little as 5% of waste generated in the mining sector is deposited in landfill on the ground surface. This does not mean, however, that the amount of waste disposed on the ground has decreased proportionately, as the waste subjected to so-called recovery processes is usually located in the form of various types of earthworks, as well as used for land levelling and reclamation. Accordingly, regardless of whether the wastes are stored on landfill or used economically (to build embankments, earthworks, levelling of land), they are in fact placed in the environment on the surface

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of the earth and affect the environment. The distinction between recovery and landfilling of waste is a technical rather than a legal issue, and is related to regulations governing waste management. In one instance the placement of material in the ground (for example road foundations) will be viewed as recycling, whilst in another, placing the same material on the ground may be treated as landfilling, which is comprised in a completely different legal category. The characteristic feature of hard coal mining wastes is that they are largely not inert wastes as intended by Article 3(3) of Directive 2006/21/EC. According to this article, the term "inert waste" refers to waste that does not undergo any significant physical, chemical or biological transformations. Inert waste will not dissolve, burn or otherwise physically or chemically react, biodegrade or adversely affect other matter with which it comes into contact, likely to give rise to environmental pollution or harm human health. The total leachability and pollutant content of the waste, and ecotoxicity of the leachate, should be insignificant, and in particular not endanger the quality of surface water and/or groundwater. From a technical point of view, the management of actual mining wastes should consist in processing of the latter to render them either inert wastes that are harmless to the environment, or conversion into a product compliant with relevant standards.

Based on this definition, the majority of waste generated from the extraction of hard coal is non-inert waste, especially those containing carbon residues and sulfur compounds. According to the applicable regulations, i.e. "Ustawa z dnia 10 lipca 2008 r. o odpadach wydobywczych, Dz. U. Nr 138, poz. 865 z późn. zm." (the act on mining waste), these wastes should be disposed of in the so-called Waste Neutralization Facility. Wastes of this type may also be processed using appropriate technology in products of standard value that conforming to relevant standards. The latter represents an optimum system of waste management as it contributes towards protecting the environment and safeguarding natural raw material resources, which can be replaced by the waste material. Wastes from hard coal mining are generated in millions of tonnes and should be utilized in industries equipped to manage such large quantities of material; the construction material industry is perfect for this purpose, once the technical properties have been appropriately verified. A good example of an appropriate industry is represented in the production of building ceramics, where clay raw materials are traditionally used, having a mineral and chemical composition similar to that of wastes from hard coal mining. An average-sized factory in this industry produces hundreds of thousands of tonnes of products, using the same quantity of raw materials.

## 2. MATERIALS AND METHODS

### 2.1 Materials

Waste mud originating from the hard coal mining industry in Poland was used in this study. The basic characteristics of the material were assessed using the methods described below in terms of the possibility of using this waste as a raw material in the production of fired ceramic building materials. The material is produced in large quantities and requires the use of large-scale management methods. The potential utilisation site for this type of waste may be the ceramic industry that uses a large amount of raw materials. The material initially has a high humidity of more than 30%, with a thick pasty consistency depending on storage conditions, which may, over time, dry and harden. Hardened material may need to be crushed for further use with the use of appropriate machinery and equipment, although due to the consistency of the material, the process should not require a lot of energy.

### 2.2 Methods

Raw waste material was subjected to chemical composition analysis by means of Wavelength Dispersive X-ray Fluorescence method, using WDXRF Axios maX PANalytical analyser, with 4 kW Rh lamp.

Analysis of the mineral composition of the raw material was carried out by XRD method (Phillips PW-1040 analyser, in the 2θ measuring range 5-70°).

Thermal characteristics of the raw material were characterized by means of simultaneous thermal analysis using NETZSCH STA 449 F3 Jupiter®; analysis of gaseous products was conducted using coupled mass spectrometer QMS 403 Aëolos (heating rate 10°C/min, atmosphere: synthetic air, dynamic flow 40 ml/min).

Thermal behaviour of waste sample with recording of area changes on heating was carried out using the Hesse Instruments heating microscope (heating rate 10°C/min to 1400°C in static air atmosphere).

Particle size distribution was measured by means of the Mastersizer 2000 device by Malvern Instruments. The device measures particle size in the range of 0.2 - 2000 µm. Measurements were made in liquid dispersion (isopropanol) on a 5 gram sample.

The specific surface area of the waste mud sample was measured using the Blaine method according to PN-EN 196-6 standard.

## 3. RESULTS AND DISSCUSSION

The analysis illustrated below shows basic properties of the raw material in view of the potential of use in production of building materials obtained by means of technologies including thermal treatment of materials. Characterization comprised chemical analysis of the material, and evaluation of thermal properties, mineral composition and particle characterization.

### 3.1 T Raw materials characteristics

*3.1.1 Chemical analysis of the sample*

Chemical analysis of the sample is shown in Table 1 and relates to 11 items. The dominant component in the tested sample was silicon dioxide, accounting for more than 53% of total sample content. Aluminium oxide was also present in the sample (26% of total content), thus underlining the possibility of using this waste in the production of building materials. The third largest content was iron oxide, with a value of approx. 10%. The oxides described above represent 90% of the total sample. Chemical analysis also showed the presence of approx. 4% alkaline ox-

**TABLE 1:** Semiquantitive chemical analysis (XRF) of raw materials.

| Oxides | Content [%, wt] |
|---|---|
| $SiO_2$ | 52.4 |
| $Al_2O_3$ | 25.6 |
| $Fe_2O_3$ | 9.6 |
| $TiO_2$ | 1.3 |
| CaO | 2.4 |
| MgO | 1.8 |
| $K_2O$ | 3.6 |
| $Na_2O$ | 0.5 |
| $SO_3$ | 2.3 |
| $P_2O_5$ | 0.3 |
| Cl | 0.2 |
| Sum | 100 |

ides ($Na_2O$ + $K_2O$). Sodium and potassium oxides were not a predominant component of waste, but should not be underestimated. With regard to the process of storing these wastes, alkalis may represent an ecological hazard, while processing at high temperatures may exert a positive influence by reducing the temperature of eutectic points in the formation of desired phases. The presence of 2% calcium oxide (CaO) and also 2% $SO_2^{2-}$ suggests that oxides may be present in the form of calcium sulphate compounds as well as in the form of sulphides (pyrite).

*3.1.2 STA-EGA analysis of raw materials*

Figure 1 illustrates thermal DTA/TG/DTG curves of the waste mud sample on heating to 1000°C. The initial loss of t sample mass, just over 2% wt, visible on the TG curve with a peak at about 70°C in the temperature range 30÷200°C, is related to adsorbed water evaporation and possibility dehydration of clay minerals. This effect is observed on the DTA curve as a slight endothermic deflection. The second, much larger, mass loss observed between 200÷745°C is mainly due to oxidation of organic matter and is equal to approx. 31%. On the DTA curve this effect is marked as wide exothermic deflection between 200÷745°C with low endothermic effect peaking at about 514°C, likely due to dehydroxylation of clay minerals. No effects were registered above the temperature of 745°C on DTA and TG curves, implying that no exo and endothermal processes took place in the material during further heating.

The EGA curves of waste mud samples are shown in Figure 2 in form of ionic current curves in function of the temperature for $CO_2$ (m/z =44), $H_2O$ (m/z = 18) and $SO_2$ (m/z = 64). The EGA curves are confirmative and complementary to DTA TG results and facilitate interpretation

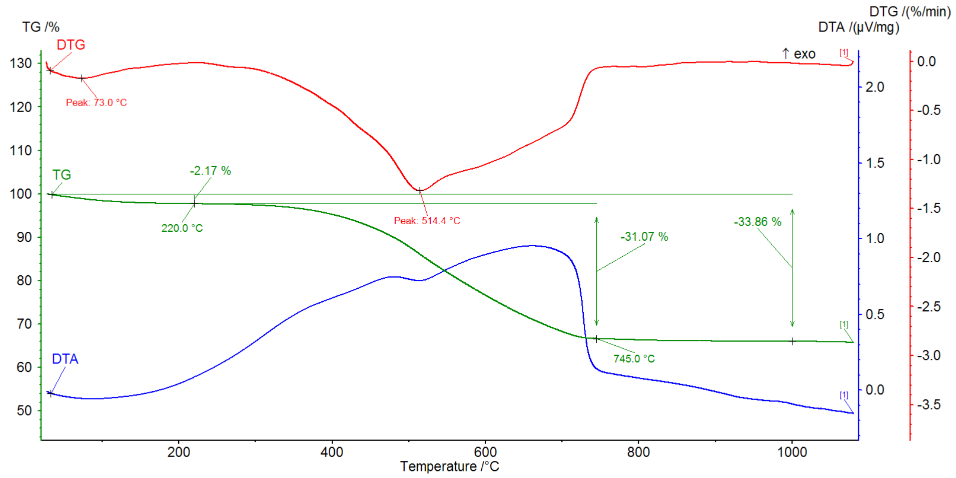


**FIGURE 1:** DTA/TG/DTG curves of waste mud.

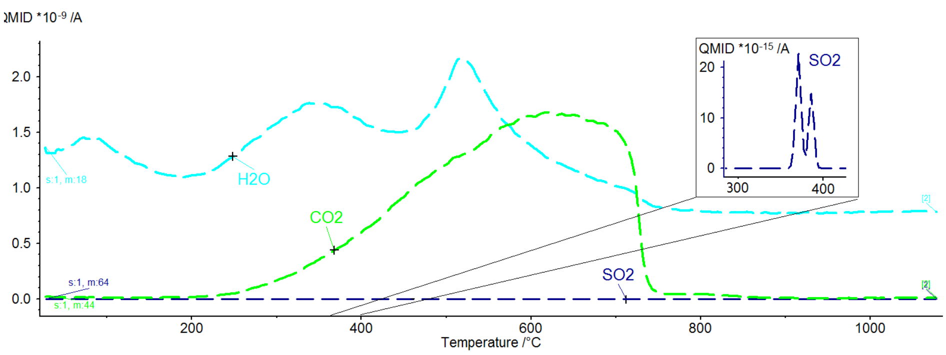


**FIGURE 2:** EGA curves of waste mud.

of the results of thermal analysis allowing for the separation of processes in temperature coincidence with others. The curves show that water is released in three steps between 20 and 745°C. First release is due to dehydration of clay minerals, second release to firing of organic matter (convergence with release of $CO_2$) and third release to dehydroxylation of clay minerals. $CO_2$ is released between 220°C and 745°C in one step visible as a large and wide convexity. This release is connected with burning of organic matter present in the sample mainly in the form of residual carbon released into the atmosphere as $CO_2$. The $CO_2$ curve illustrates gas release kinetics and the asymmetrical shape of the convexity. This demonstrates that gas release rate is variable with temperature. From a temperature of 220°C the quantity of evolved $CO_2$ increases uniformly up to about 600°C. The maximum level of gas release is reached between 600°C and 700°C and then suddenly ceases. $SO_2$ emission is very low (3 orders of magnitude lower) in comparison to other gases and is placed in the middle of the temperature range between 370°C and 400°C. The temperature range of sulphur dioxide emissions suggests the presence of sulphide (pyrite) residues in the sample. No gas emissions are detected above a temperature 7of 45°C, and no changes are observed on TG curve. This information may facilitate set up of the firing program of products the properties of which are obtained though high temperature treatment.

*3.1.3 High-temperature characteristics of raw material observed by a heating microscope*

Figure 3 shows the thermal behaviour of raw material as observed through a heating microscope. The results of measurements are shown in the form of a curve reflecting changes in the sample profile area as a function of temperature; a set of images (Figure 4) showing the changes in shape of the sample profile while heating at temperatures ranging between room temperature to 1400°C is also provided. From the start of measurements up to a temperature of 830°C, the sample displays a small, regular thermal expansion corresponding to approximately 3% in comparison with the area manifested at 22°C. At a temperature of 830°C, the sample begins to shrink as the sintering process starts and continues up to a temperature of 1260°C. At this temperature, the sample profile reaches a minimum area corresponding to 61% the initial value and is referred to as the temperature of maximum sintering of the material. On reaching maximum sintering, as a result of a further rise in temperature, the sample profile begins immediately to swell and increases to approx. 76% once measurements stop at 1400°C.

The results obtained show how the material has a desirable sintering characteristic, placing it as a potential raw material for use in the production of fired ceramic building materials. By means of this technology, the thermal properties of the material are particularly advantageous, consisting in a wide sintering interval that starts at a relatively low temperature of approx. 830°C and peaks at 1260°C. This behaviour of the raw material on heating may result in a wide range of materials, from highly porous materials fired at low temperatures to highly sintered products with low porosity fired at higher temperatures with use of fluxes. Such a wide sintering temperature interval promotes the safe firing of ceramic products without the risk of deformation caused by reaching of the softening point.

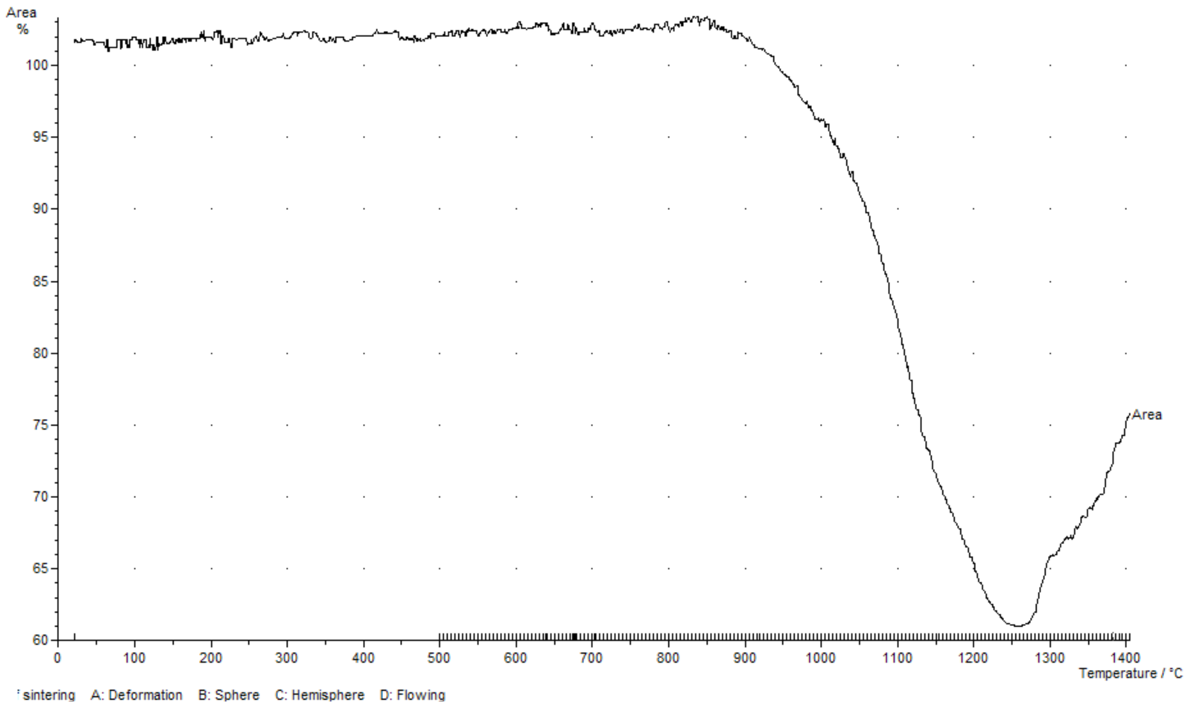


**FIGURE 3:** Curve showing the changes in sample profile area [in %] in line with measurements taken under the heating microscope.

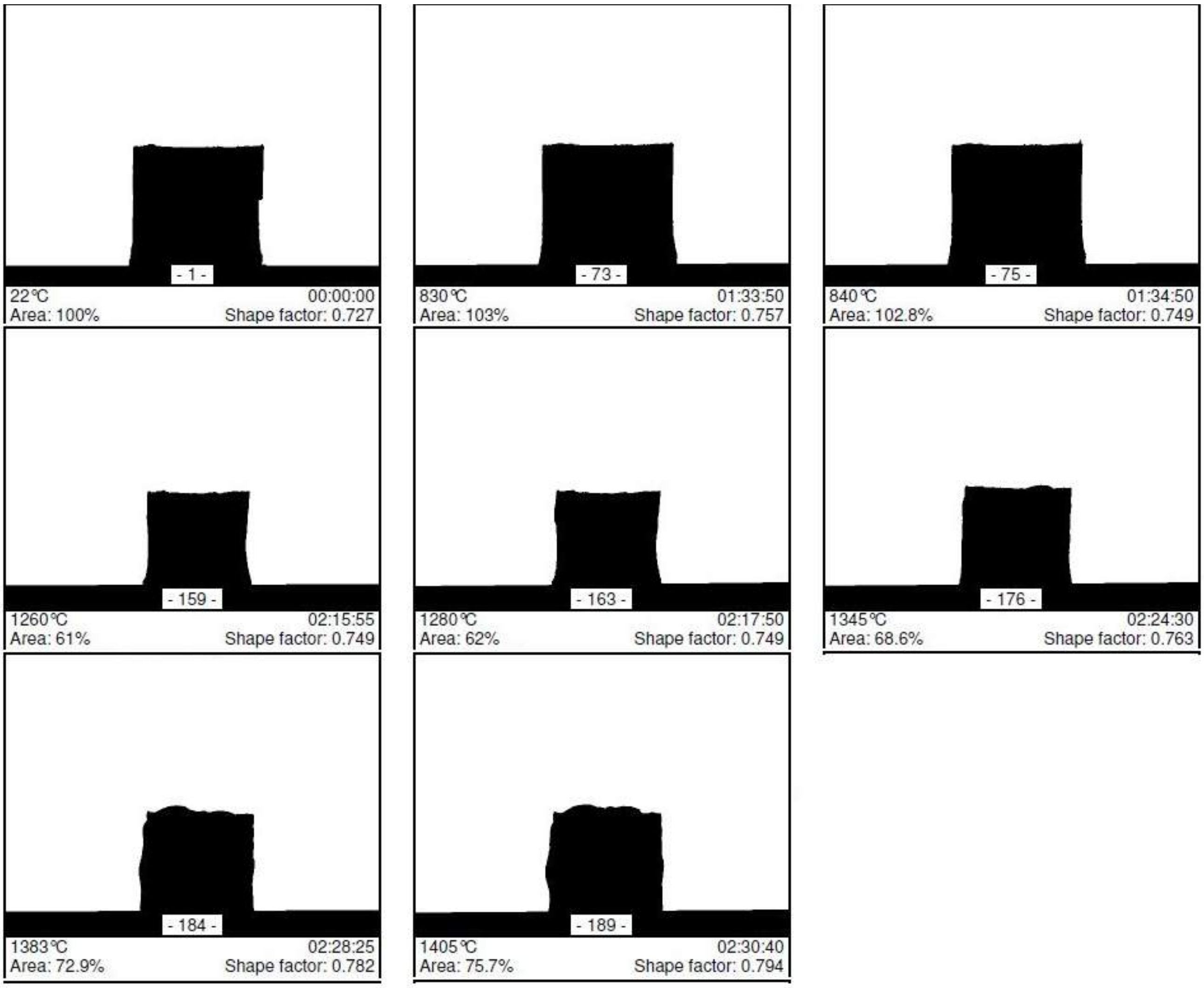


**FIGURE 4:** Images of changes in sample profile shape in line with measurements taken under the heating microscope.

*3.1.4 XRD analysis of sample material*

XRD analysis (Figure 5) revealed the presence of three crystalline phases visible in the diffraction pattern. In the range of low angles of 2 theta, i.e. the range from 0 to 10 degrees, there is a clearly visible reflect [dhkl = 10.000] coming from the crystalline phase of illite [ICDD 26-911]. Characteristic next peaks with lower intensity coefficients coming from the illite phase are assumed to correspond to the following values dhkl = 5.025; 3.344; 2.004. In the angular range of 10-15 2 theta, the most intense visible reflection comes from the kaolinite phase [ICDD 29-1488] with values of the strongest reflection corresponding to dhkl = 7.194. The marked phase of the hydrated aluminosilicate and its successive peaks assume the following values dhkl = 3.584; 2,342; 1,488.

The dominant phase in X-ray diffraction pattern is the $SiO_2$ phase with ICDD 33-1161, the highest reflection of which occurs in the angular range of 25-30 2theta (dhkl = 3.344). Further peaks occur throughout the course of the diffractogram and the values of the most intense peaks are shown in dhkl = 1.818; 1.541; 4.255.

It should be noted that the accuracy of diffractometric analysis reaches a value of crystalline phase content corresponding to approx. 5%. All crystalline phases present in the tested waste material below this value are outside the detection limit of XRD method.

*3.1.5 Analysis of particle size distribution and specific surface area of the waste mud sample*

The results of Blaine method air permeability analysis showed that the test material had a high surface area exceeding 10 000 $cm^2/g$ at a specific density of 2.19 $g/cm^3$.

The surface area development analysis was supplemented by a detailed particle size analysis, the results of which are illustrated in the Figures 6 and 7. The particle size analysis showed how the material tested was very fine. The maximum grain size did not exceed a diameter of 0.5 mm. Detailed analysis of particle size ranges showed that the dominant fraction in the sample was 2.8 to 10 μm particles with a content of 31% v / v of the total sample. Significant contents in the sample were represented by 18% content of particles in the range of 10-19 μm and 16% particles in the range 19-40 μm Grains sized from 1.2 to 2.8 μm corresponded to 14% content and 40-100 μm 10%. The

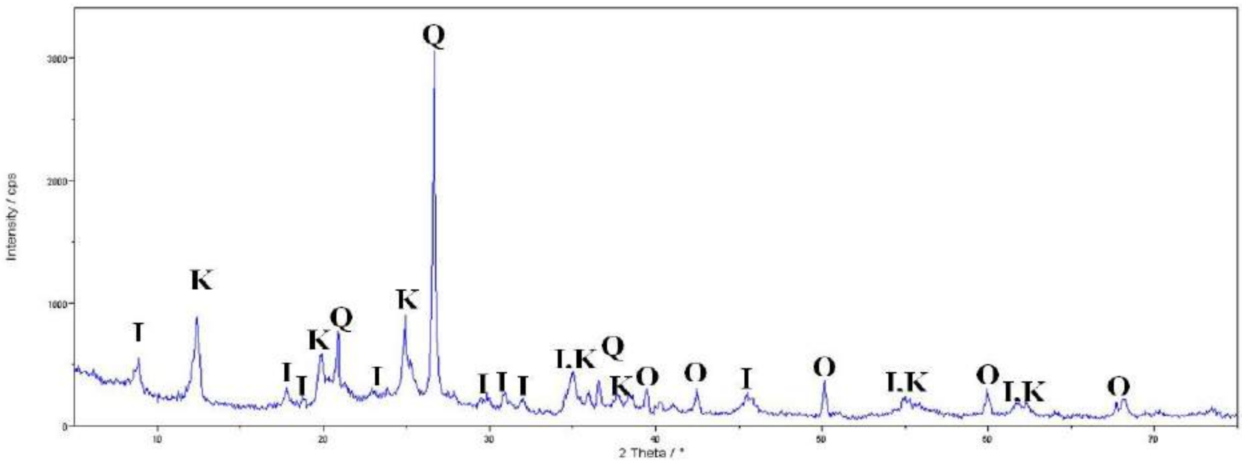


FIGURE 5: XRD curve of the waste mud sample; I - illite, K - kaolinite, Q - quartz.

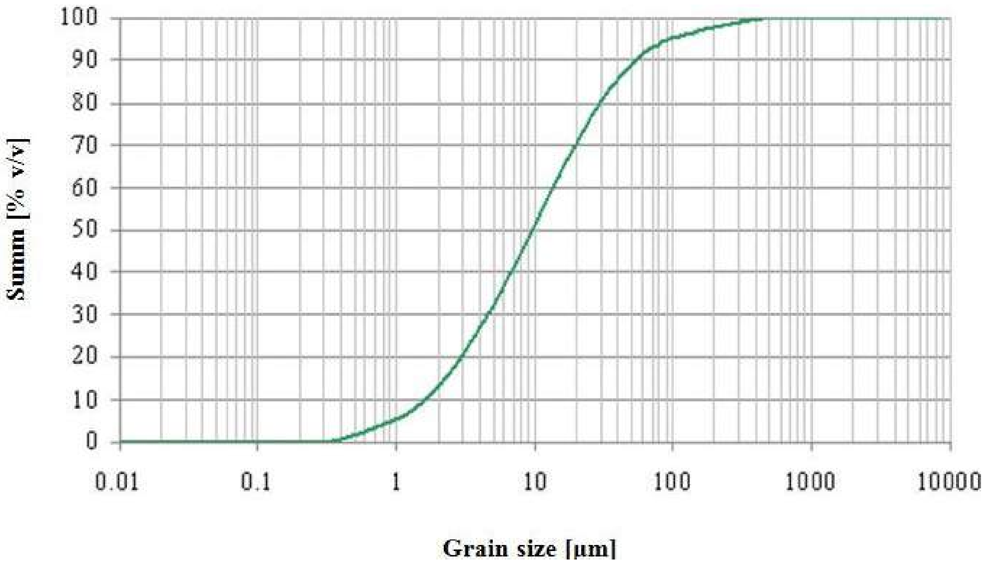


FIGURE 6: Cumulative curve of particle size distribution of the waste sample.

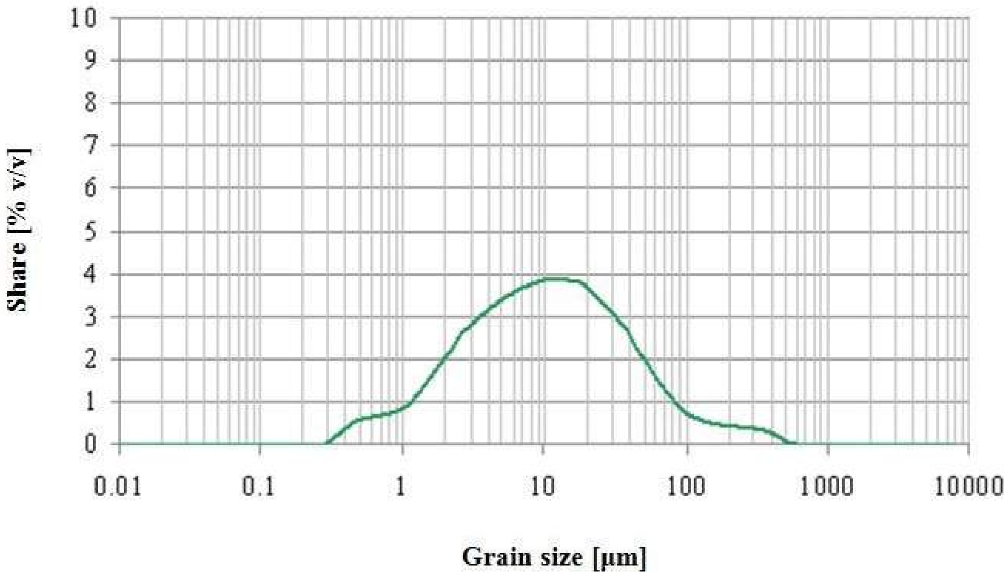


FIGURE 7: Histogram curve of particle size distribution of the waste sample.

sample was also characterized by particles sized between 0.5-1.2 and 100-300 µm in the range of 4% and 0.3-0.5 and in the range of 300-600 µm in the amount of about 1.5%.

## 4. CONCLUSIONS

The present study was focused on the preliminary characterization of waste from coal mines in the context of evaluation of its usefulness in the production of ceramic building materials. The results obtained in terms of thermal properties, mineral composition and granulometric analysis of the examined waste showed that these wastes could potentially constitute a component of ceramic masses in these technologies. In agreement with other studies, XRD analysis showed the presence of three main crystalline phases at varying contents in the material studied. The most prevalent component was $SiO_2$ quartz, as confirmed by chemical analysis, with approx. 54% of silica content. The second and third crystalline phases present in the sample were illite and kaolinite, both hydrated layer aluminosilicates. The content of these crystalline phases was confirmed by chemical analysis, which revealed an approx. 25% alumina content and approx. 3.5% potassium oxide. It should be noted that small amounts of other cations, such as $Fe^{3+}$ or $Na^{2+}$, may be incorporated into the structure of clay minerals. Due to their crystallographic structure, with these silicates (mostly illite), the possibility of isomorphic substitution in the elemental cell and the ionic radii of the cations listed may serve as a matrix in which these cations can be incorporated. The suggested phenomenon may be explained by a small drift of peaks in the kaolinite diffractometric image in which $Al^{3+}$ cations may incorporate $Fe^{3+}$ cations. The same mechanism applies to illite mineral and sodium cations. It should also be noted that in the XRD image no reflections from hydrated calcium sulphates are visible, likewise failing to show STA thermal analysis, although the findings of chemical analysis suggest that this compound may be present. It is likely that content of the latter is below the detection threshold of these methods. The low sulfur content in form of sulphates and sulphides is advantageous for building ceramics, as it reduces the risk of corrosion of burnt products. The results of thermal analysis showed a significant amount of organic matter (carbon residue) in the waste to be treated, which may be problematic during firing, although at times may also prove advantageous. The organics present in the ceramic mass (in this case carbon) reduces consumption of the main fuel during firing due to an exothermic combustion effect, but only in the presence of a suitable firing process that ensures oxidation conditions in the kiln.

The results of thermal behaviour characterisation demonstrates that the material has a desirable sintering characteristic, placing it as a potential raw material for use in the production of fired ceramic building materials. In using this technology, similar thermal properties for the material, consisting in a wide sintering interval starting at a relatively low temperature (840°C) and reaching approx. 1260°C is an advantage. These properties of the raw material, when used during firing in the production of ceramics, facilitates the obtaining of a wide range of materials, from highly porous products fired at low temperatures to highly sintered with low porosity fired at higher temperatures with use of special fluxes. Such a wide sintering temperature interval likewise promotes the safe firing of ceramic products, without the risk of deformation caused by reaching of the softening point

The preliminary results obtained, confirming the presence of clay minerals, fine graining and thermal properties of the waste, suggest potential possibilities of using these wastes as a raw material to correct the rheological properties of the ceramic mass and porosity of the fired material in the technology of building ceramics.

The use of the waste material as a ceramic mass component in combination with a specific basic clay raw material should always be confirmed by initial laboratory tests. Subsequent tests should be performed on an industrial scale to determine the appropriate content of the additive and appropriate parameters for the manufacturing process.

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# INTEGRATED MICRO X-RAY FLUORESCENCE AND CHEMOMETRIC ANALYSIS FOR PRINTED CIRCUIT BOARDS RECYCLING

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**ABSTRACT**

A novel approach, based on micro X-ray fluorescence (µXRF), was developed to define an efficient and fast automatic recognition procedure finalized to detect and topologically assess the presence of the different elements in waste electrical and electronic equipment (WEEE). More specifically, selected end-of-life (EOL) iPhone printed circuit boards (PCB) were investigated, whose technological improvement during time, can dramatically influence the recycling strategies (i.e. presence of different electronic components, in terms of size, shape, disposition and related elemental content). The implemented µXRF-based techniques allow to preliminary set up simple and fast quality control strategies based on the full recognition and characterization of precious and rare earth elements as detected inside the electronic boards. Furthermore, the proposed approach allows to identify the presence and the physical-chemical attributes of the other materials (i.e. mainly polymers), influencing the further physical-mechanical processing steps addressed to realize a pre-concentration of the valuable elements inside the PCB milled fractions, before the final chemical recovery.

## 1. INTRODUCTION

PCB are an essential part of all technological devices commonly utilized by consumers. The development of technology and society, as well as the higher performance of the electronic devices, and the corresponding miniaturization, produce, as a consequence, an increase of the waste electrical and electronic equipment (WEEE) to be disposed of (Zhou et al., 2010). Precious and rare earth elements are around us, not only in nature but in our everyday lives (i.e. car, computer, smartphone, energy-efficient fluorescent lamp, and color TV, as well as in lasers, lenses, ceramics and in many other products). We need rare earths for so many applications, but their supply is restricted to few mining districts in the world, this fact led these elements to create a critical-metals agenda. (Chakhmouradian et al., 2012). To face this crucial situation, the European Commission in 2008, through the "European Raw Materials Initiative" (European Commission, 2008) suggested a combined strategy based on enforcing deeper links and co-operation contracts with producer countries (by improving foreign investment agreements), encouraging and promoting internal mining potential and developing more efficient recycling policies (Massari et al., 2013; Tiess et al., 2010). An ad-hoc working group of the European Union has determined a set of critical resources as: Be, Co, Ga, Ge, In, Mg, Nb, Ta and W, the platinum group metals (PGM): Pt, Pd, Rh, Ru, Os, Ir) and rare earth elements (REE) (European Commission, 2010). Starting from these premises the possibility to utilize specific EOL products as secondary raw materials sources, the recovery of precious and rare earth elements, practically "became a must". The concept of "urban mining", referred to the different EOL materials and/or manufactured products of human origin, was thus introduced: precious and rare earth elements in dismissed electronic units of large use as mobile phones, "tablets" and personal computers, thus representing an important secondary raw materials source (Hagelüken et al., 2010; Palmieri et al., 2014).

The most commonly applied techniques for determination of rare earth elements are inductively coupled plasma–optical emission spectrometry (ICP-OES), inductively coupled plasma mass spectrometry (ICP-MS), X-ray fluorescence (XRF) and neutron activation analysis (NAA) (Zawisza et al., 2011). ICP-MS and ICP-OES require a preliminary strong manipulation of the samples in order to separate the rare earth element from the matrix. This approach is complex, time-consuming, and can always be a potential source of random, or even systematic, errors. The NAA technique has been increasingly utilized to de-

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tect the presence of rare earth elements in solid samples. Sometimes, NAA presents practical implementation problems mainly due to interference and the required long radiation time (Kumar et al., 2014). Recently also laser-induced breakdown spectroscopy (LIBS) techniques (Carvalho et al., 2015) have been successfully applied to determine rare earth elements. The XRF technique offers the possibility to determine rare earth elements in solid materials not requiring, as for LIBS, any specific sample preparation; it also allows simultaneous determination of both trace and main components (Zhang et al., 2007; De Vito et al., 2007; Smoliński et al., 2016). However, to obtain a correct quantification of the detected elements it is necessary to consider matrix effects (absorption and enhancement) and peaks overlapping. Theoretically, the intensity of a peak is linearly proportional to the concentration of the analyte, but practically the intensity of a peak does not depend only on the concentration of the respective elements, but it is also determined by presence and concentrations of other elements and by the interaction with matrix (Smoliński et al., 2016).

The new generation of scanning XRF analytical units, based on confocal XRF method, realizes the best acquisition conditions, both in terms of speed and analytical data set reliability. For every acquired hyper-map, a XRF spectrum is associated to each pixel. Thus, an acquisition consists in a $n \times m$ matrix of spectra, where n and m are the number of pixels in the x and y direction, respectively. Analyzing every spectrum associated to each pixel, it is possible to know the total counts corresponding to a given energy. By selecting the energy ranges depicted in the spectrum, it is possible to obtain a 2D image corresponding to the distribution of selected elements (Figueroa et al., 2014).

Aim of this work is to verify the possibility to utilize the confocal µXRF imaging based approach as an analytical technique to perform an automatic detection and mapping of the elements present in dismissed iPhone PCB and/or in the products (i.e. particles) resulting from their mechanical-physical processing before the final chemical recovery (i.e. leaching). Following this approach, it is thus not necessary the presence of an operator performing a preliminary identification/selection of the different energy ranges/peaks representative of a specific element. To reach this goal, data were analyzed by chemometric techniques (exploration and classification methods) and the results compared with the maps of the elements obtainable by the classical approach, that is the manual selection of the energy ranges associated to each pixel. This approach could be successfully applied to perform quality control actions referred to other WEEE and resulting milled/classified products for precious and rare earth elements chemical recovery.

## 2. MATERIALS AND METHODS

### 2.1 Samples and experimental set up

The reference PCB samples utilized for the analysis are constituted by 3 electronic boards belonging to 3 different iPhone models (i.e. iPhone 4, iPhone 3s and iPhone 4s) (Figure 1). Investigations have been carried out with reference to 8 elements (palladium, silver, gold, zinc, copper, tantalum, lead and iron), being among those of higher interest in terms of recovery and/or interfering actions, when XRF analyses are performed.

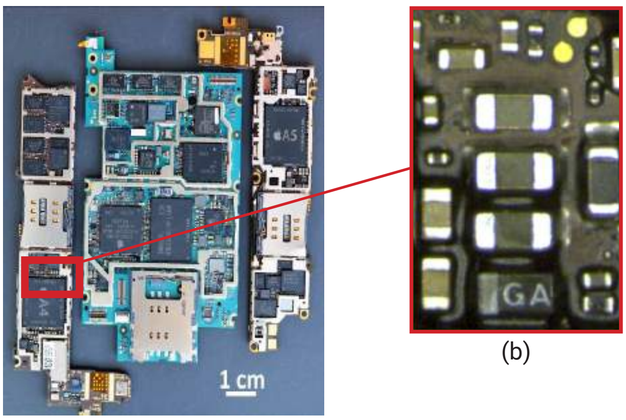
Precious and rare earth elements are in a small concentration on each electronic board, however the preliminary correct identification, and further separation, when applied on large quantities, allow the process to be economically valid (Bonifazi et al., 2017).

The µXRF based elements mapping was performed at Raw-Ma Lab (Raw materials Laboratory) of the Department of Chemical Engineering, Materials & Environment (Sapienza - University of Rome, Italy) using a benchtop spectrometer (M4 Tornado, Bruker®) equipped with a Rh X-ray tube with poly-capillary optics as the X-ray convergence technique, and XFlash® detector providing an energy resolution better than 145 eV and 5 filters (Guerra et al., 2013).

The whole spectra comprised 4096 channels with a spot size of approximately 30 µm. Spectrum energy calibration was daily performed before each analysis batch by using zirconium (Zr) metal (Bruker® calibration standard). The sensitivity of µXRF is determined by the excitation probability of the sample and the peak to background ratio. The background intensities were directly computed by the equipment (ESPRIT Bruker® software). The sample chamber can be evacuated to 20 mbar and, therefore, light elements such as sodium can be measured (Nikonow et al., 2016). Constant exciting energies of 50 kV and 500 µA, were adopted for acquisition. The set up mapping acquisition parameters were a pixel size of 80 µm and an acquisition time, for each pixel, of 6 milliseconds. Spectral data (i.e. hyper-maps) analysis was carried out adopting chemometric methods, using the PLS_Toolbox (version 8.2.1, Eigenvector Research, Inc.) running inside MATLAB (version 9.1.0, The Mathworks, Inc.).

### 2.2 µXRF: acquisition and data handling

The experimental procedure was defined and implemented in two steps: the 1st one finalized to the acquisi-




(a) (b)

**FIGURE 1:** Digital image representing the electronic 3 acquired boards (a) and an example of one of the region of interest (ROI) selected to set up the best filtering/acquisition conditions (b).

tion of the hyper-maps and the further XRF peaks deconvolution and the 2nd one addressed to the acquisition of the calibration standards and to the classification of the different elements detected in the electronic boards.

*2.2.1 Step 1: hyper-maps acquisition and XRF peak deconvolution*

Smartphone electronics boards (Figure 1a) were acquired by µXRF in order to build hyper-maps of all the elements. Small regions of interest (ROI) (Figure 1b) of the electronics boards were selected and acquired utilizing different filters in order to set up the best conditions to reduce the signal of light elements (i.e. silicon) maximizing, at the same time, the signal of precious and rare earth elements (Gallardo et al., 2016). One of the main strategies to apply in order to improve measurement conditions for elements of interest is, in fact, the utilization of primary beam filters, aluminum made, that are placed between the X-ray source and the sample. In the Bruker M4 Tornado µXRF device, five internal filters are available (Al 12.5 µm, Al 100 µm, Al 630 µm, Al/Ti 100/25 µm and Al/Ti/Cu 100/50/25 µm).

*2.2.2 Step 2: acquisition of calibration standards and identification of elements by Partial Least Squared Discriminant Analysis (PLS-DA)*

Calibration standard were acquired by µXRF to build a classification model able to recognize the different elements without any human based investigation finalized to optimal mapping set up to enhance the presence of precious metals and rare earths. A set of 8 elements clearly identified in the dataset as palladium, silver, gold, zinc, copper, tantalum, lead and iron was used as training dataset to build the classification model (Figure 2). The classification model was then validated utilizing the electronic boards dataset generated by the experimental approach described in Step 1.

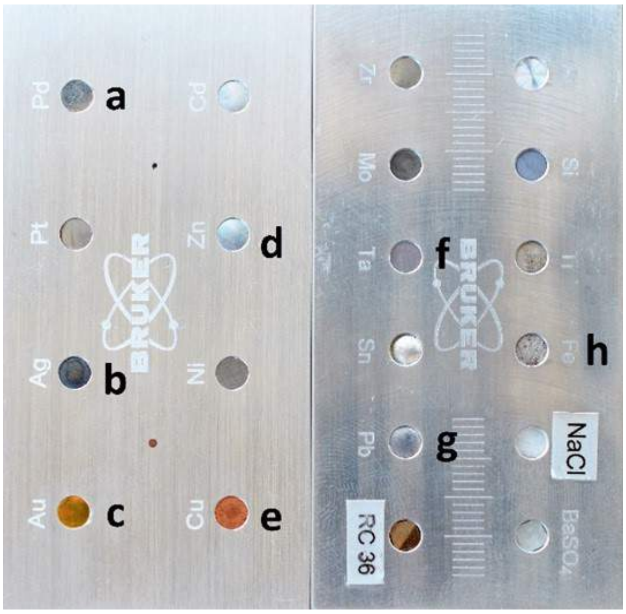



**FIGURE 2:** Digital image representing the acquired standard calibration elements: a: palladium, b: silver, c: gold, d: zinc, e: copper, f: tantalum, g: lead and h: iron.

Spectral data analysis was preliminary addressed to explore and to evaluate the quality of the acquired information to be utilized for the further classification model definition, design, implementation and set up. To reach these goals, a preliminary ***P**rincipal **C**omponent **A**nalysis* (PCA) and a further ***P**artial **L**east **S**quares **D**iscriminant **A**nalysis* (PLS-DA) were carried out.

PCA is the most utilized multivariate data analysis method for exploratory data handling, outlier detection, rank (dimensionality) reduction, graphical clustering, classification, regression, etc. (Bro et al., 2014). It was used to decompose the "pre-processed" spectral data into several principal components (PCs) (linear combinations of the original spectral data) embedding the spectral variations of each collected spectral data set. According to this approach, a reduced set of factors is produced and used for discrimination since it provides an accurate description of the entire dataset. The first few PCs, resulting from PCA, are generally utilized to analyze the common features among samples and their grouping: samples characterized by similar spectral signatures tend to aggregate in the score plot of the first two or three components. Spectra could be thus characterized either by the reflectance at each wavenumber in the wavenumber space, or by their score on each PC in the PC space (Bro et al., 2014). Samples characterized by similar spectra, belonging to the same class of products, are grouped in the same region of the score plot related to the first two or three PCs, whereas samples characterized by different spectral features will be clustered in other parts of this space.

PLS-DA was used to find a model able to perform an optimal discrimination among classes of samples and to predict new images. PLS-DA is a supervised classification technique, requiring a prior knowledge of the data (Ballabio et al., 2013). PLS-DA is used to classify samples into predefined groups by forming discriminant functions from input variables (KeV) to yield a new set of transformed values providing a more accurate discrimination than any single variable. A discriminant function is then built using samples with known groups to be employed later to classify samples with unknown group set. Therefore, once the model is obtained, it can be applied to an entire hypercube and for the classification of new hypercubes. The result of PLS-DA, applied to the hyperspectral images, is a "prediction map," where the class of each pixel can be identified using color mapping.

## 3. RESULTS AND DISCUSSION

Results and discussion are reported in the following, presenting and comparing the classical human based µXRF mapping approach and the proposed one based on PLS-DA classification.

### 3.1 Step 1: acquisition of hypermaps and deconvolution of XRF peaks

The "preliminary" hyper-maps acquisition of the selected ROIs was carried out adopting two different aluminum filters (Al100 and Al630), whose aim is mainly to reduce the signal due to Si and Ba presence, thus allowing better heavy metals display, and the results were compared with those obtained with the acquisition without filter. An example is reported in Figure 3. The map of the elements, referred to the ROI, clearly shows as the detection of gold and of the other heavy elements is negatively affected by the presence of silicon and barium, present on electronic board surface.

The same ROI acquired with the Al100 filter shows a better "visualization" of gold and other heavy elements. The "noise effect" related to silicon and barium is reduced. Finally, the acquisition with Al630 filter shows the same gold distribution as that obtained using the Al100 filter, producing a stronger reduction of the signal associated to all the lighter elements (i.e. lower atomic number than barium). It was thus chosen to perform the acquisition of the hyper-maps, for all the boards, utilizing the Al100 filter. Following this strategy it was thus possible to obtain a better detection of precious and rare earth elements, reducing, at the same time, the negative effects of the signal of all those elements not of interest for this study, as silicon (Figure 4).

### 3.2 Step 2: Acquisition of calibration standards and identification of elements by PLS-DA

Calibration standards were acquired adopting the same experimental conditions utilized to perform the electronic boards hyper-mapping (Figure 5). Starting from the acquired raw data of the different calibration elements, currently utilized as standards (i.e. palladium, silver, gold, zinc, copper, tantalum, lead and iron), reference energy spectra have been selected.

The reference energy spectra associated to each element show different signatures (Figure 6a). Each energy spectra is characterized by several peaks according to the emission of a photon quantum (fluorescence radiation), related to the energy difference between the inner and outer shell. To emphasize the spectral characteristics of all the elements, "only" the mean spectra between 0 and 20 KeV, have been considered, processed and mean centered (Figure 6b), before the application of PCA. The PCA score plot allows identifying eight distinct groups according their spectral signature.

The 3D score plot (PC1-PC4-PC6) reported in Figure 7a shows a good separation (i.e. distinction) of all the el-

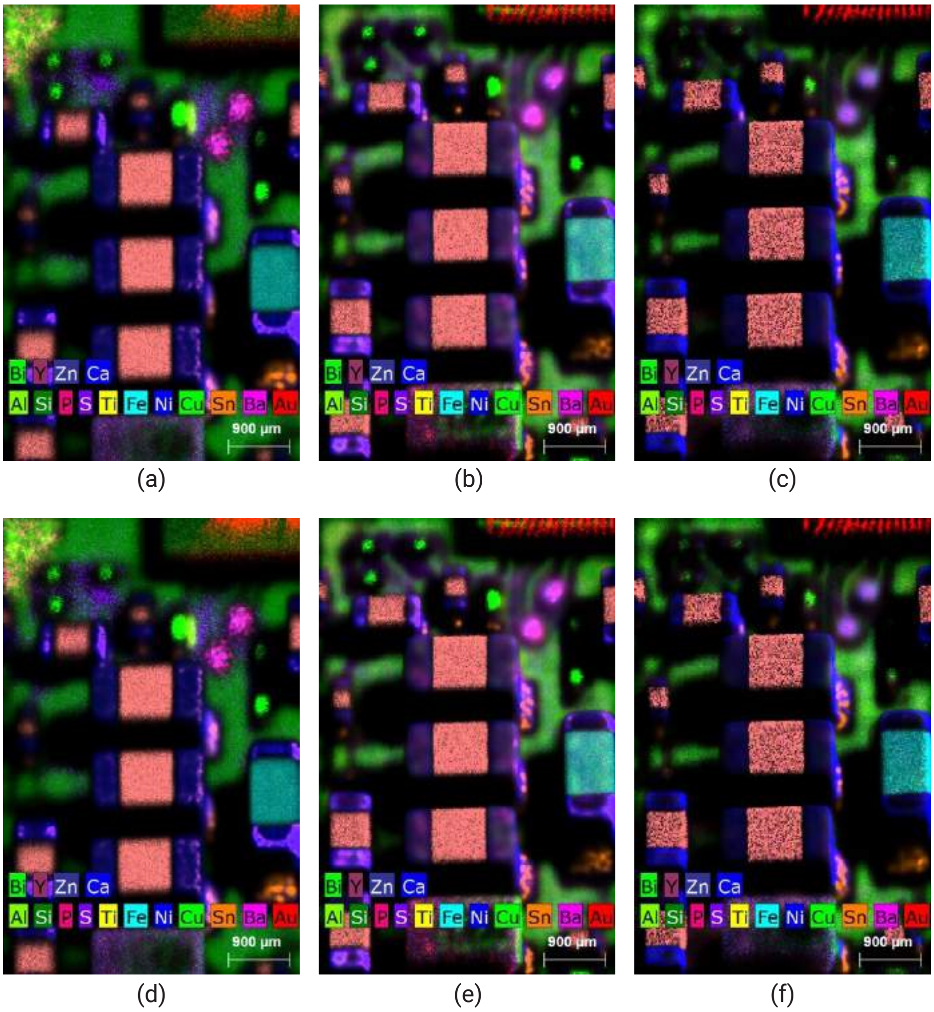



**FIGURE 3:** Hyper-maps of the elements detected by µXRF without filtering (a) and utilizing a Al100 (b) and a Al630 (c) filters, respectively. Detail of the distribution of gold and silicon without filtering (d) and utilizing a Al100 (e) and a Al630 (f) filters.


(a)

| Element | Series | [Norm.wt. %] |
|---|---|---|
| Silicon | *K-series* | 49.56 |
| Copper | *K-series* | 21.42 |
| Nickel | *K-series* | 7.71 |
| Iron | *K-series* | 6.60 |
| Zinc | *K-series* | 3.09 |
| Chromium | *K-series* | 2.90 |
| Barium | *L-series* | 2.86 |
| Tin | *L-series* | 2.56 |
| Calcium | *K-series* | 0.92 |
| Titanium | *K-series* | 0.77 |
| Tantalum | *L-series* | 0.38 |
| Palladium | *K-series* | 0.33 |
| Silver | *K-series* | 0.27 |
| Gold | *L-series* | 0.27 |
| Strontium | *K-series* | 0.14 |
| Aluminium | *K-series* | 0.14 |
| Yttrium | *K-series* | 0.04 |
| Lead | *L-series* | 0.03 |
| Bismuth | *L-series* | 0.02 |
| | Sum: | 100 |

(b)

**FIGURE 4:** Hyper-map of all elements detected by µXRF (a) and associated average concentration of the different elements as detected by acquisition (b).

ements, as well as a high uniformity for each class. The loadings of PC1, PC4 and PC6 (Figure 7b) show, in the region between 3 KeV and 15 KeV, the high variance of data, as a consequence 7 principal components are necessary to explain the variation of the calibration.

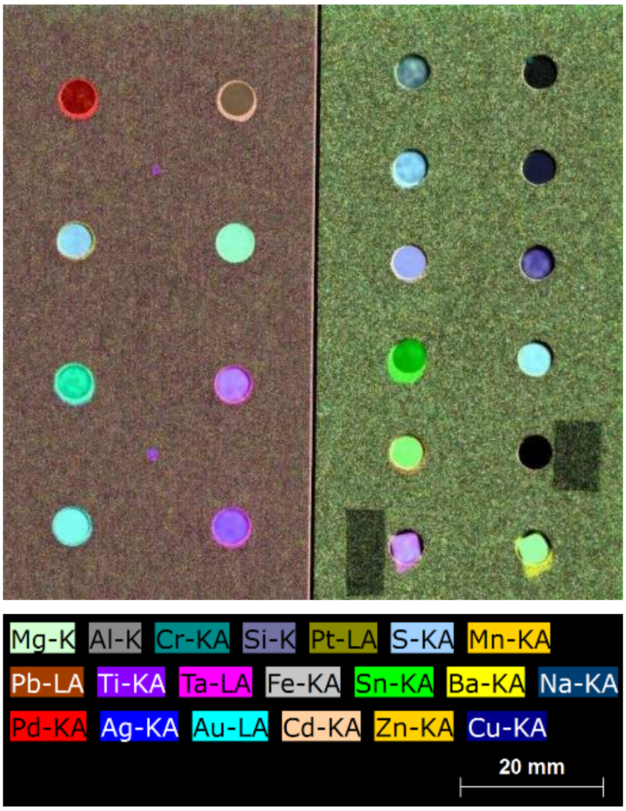



**FIGURE 5:** Elements hyper-maps of the calibration standards.

The selected energy spectra have been thus adopted as training dataset and a PLS-DA model was built. The obtained values of Sensitivity and Specificity are shown in Table 1.

The Sensitivity estimates the model ability to avoid false negatives (i.e. number of samples of a given type correctly classified as that type).

The Specificity estimates the model ability to avoid false positives (the number of samples not of a given type correctly classified as not of that type).

Sensitivity and Specificity can be assumed as model efficiency indicators: the more the values are close to one, the better the modelling is. In this study, the obtained values for Sensitivity and Specificity are very good. To verify its classification ability, the built PLS-DA model was applied to the electronic boards data set.

The results in terms of prediction (i.e. "Pred Probability") are shown in Figures 8-15: the class with the highest probability to belong to the chemical element, object of the detection/recognition, is assigned to each pixel in the image. The obtained results are very good for all the investigated elements, being comparable with those obtained following the classical "instrument-men-driven" approach. Misclassifications sometimes occur, but they are mainly due to the border effect or to the co-existence of several elements in the same pixel.

The element map of palladium (Pd) shows a low concentration but a wide distribution with a greater presence on micro-processors and electronic components according to their large utilization in electronic industry (Figure 8a). The prediction shows in the same area, mapped by µXRF, the presence of palladium confirming the good quality of the PLS-DA based modelling (Figure 8b).

The element map of gold (Au) shows a greater concen-

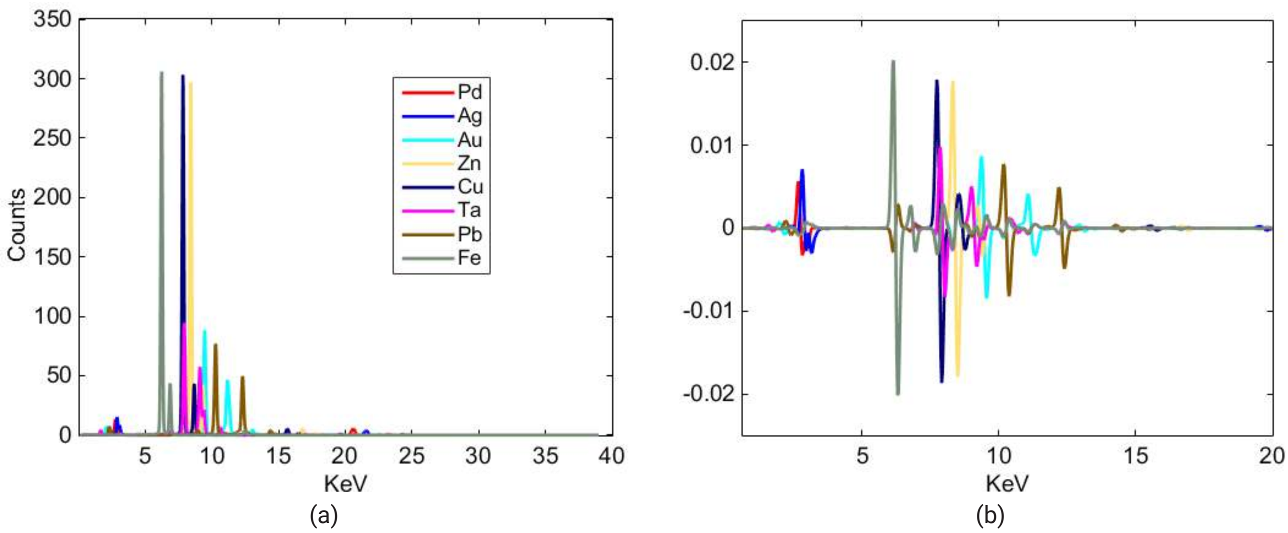



**FIGURE 6:** Raw (a) and pre-processed spectra (b), as resulting from the sequential application of the functions: Normalize (1-Norm, Area = 1), Baseline, Smoothing (order: 0, window: 5 pt, incl only, tails: polyinterp), 1st Derivative (order: 2, window: 7 pt, incl only, tails: polyinterp) and Mean Center.

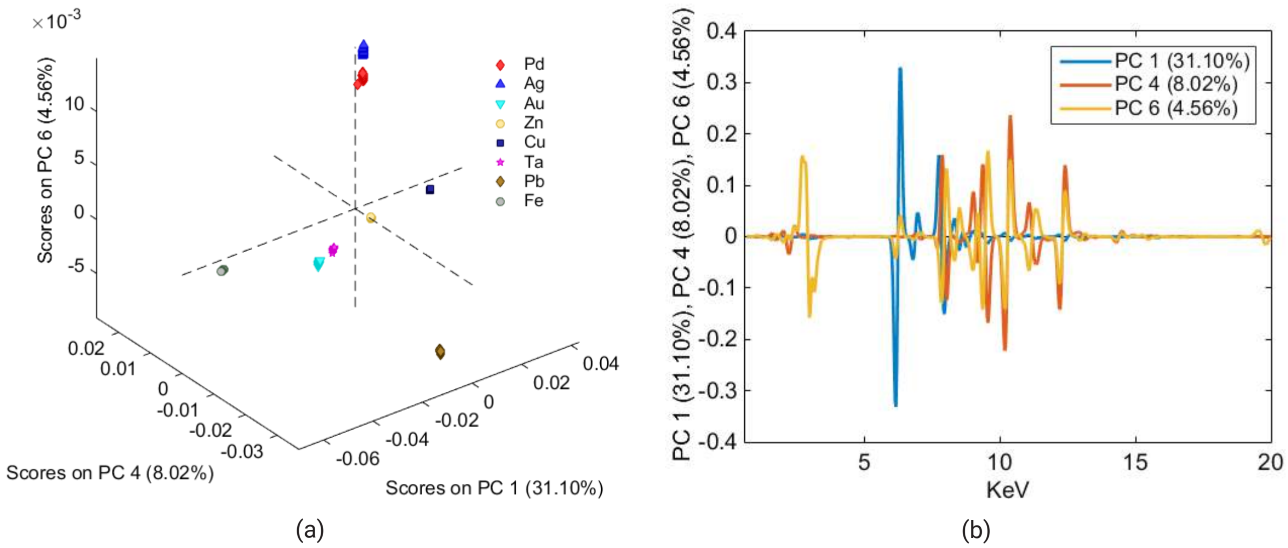



**FIGURE 7:** 3D PCA score plot (PC1-PC4-PC6) referred to the different investigated elements (a) and corresponding plot of the loadings (b).

tration of this element both with reference to component connector regions and also inside some components. In this latter case, detection is commonly realized when higher energies are utilized. The inner detection is also affected by the materials embedding gold elements (Figure 9a). The prediction map shows in the same area, mapped by µXRF, the presence of gold with a low error in classification related to its large presence (Figure 9b).

Silver (Ag) mapping shows a low concentration of this element, but a large distribution on electronic boards, as a consequence the signal is difficult to separate from background and border effects are significant (Figure 10a). The prediction map of silver shows the same characteristics, as detected by classical µXRF analysis. Silver topological assessment on the board is difficult to quantify, however, some electronic components show, in prediction, greater concentration (Figure 10b), if compared with the classical µXRF analysis.

The element map of zinc (Zn) clearly allows to identify the presence of this element both in the electronic components and in the protection structures (Figure 11a). The prediction maps show the presence of zinc in the same areas confirming the good discrimination and prediction power of the PLS-DA model (Figure 11b).

The element map of copper (Cu) shows a large concentration and distribution according to the high use of this material inside the electronic component (i.e. printed circuit tracks) (Figure 12a) The prediction maps show the same large distribution of copper to its topological assessment as resulting from classical µXRF maps (Figure 12b).

The element map of tantalum (Ta) shows its presence only in some electronic component. The total concentration of tantalum on electronic board is very low. Its primary peak (La=8.146 KeV) overlaps copper peak (Ka=8.046 KeV), therefore to perform tantalum mapping the secondary peak (Lb=9.343 KeV) was selected because it is not in-

**TABLE 1:** Sensitivity and Specificity for the PLS-DA built model.

| Modeled Class: 1 | Pd | Ag | Au | Zn | Cu | Ta | Pb | Fe |
|---|---|---|---|---|---|---|---|---|
| Sensitivity (Cal): | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |
| Specificity (Cal): | 1.000 | 0,984 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |
| Sensitivity (CV): | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |
| Specificity (CV): | 0.990 | 0.997 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 0.997 |
| Class. Err (Cal): | 0 | 0.008 | 0 | 0 | 0 | 0 | 0 | 0 |
| Class. Err (CV): | 0.005 | 0.002 | 0.012 | 0 | 0 | 0 | 0 | 0.161 |

*Cal: Calibration - CV: Cross validation*

fluenced by the presence, around the same KeV, of peaks representative of other elements (Figure 8a). The prediction of tantalum shows a correct identification according to classical element mapping results. Only in one electronic component its presence was not predicted by modelling, the reason is probably related of the high presence of other elements (i.e. zinc, gold), generating a different spectral shape compared to the reference one obtained by the tantalum reference calibration dataset (Figure 8b).

The element map of lead (Pb) shows its presence in some circuit components. The total concentration of lead results very low (Figure 14a). The lead prediction map

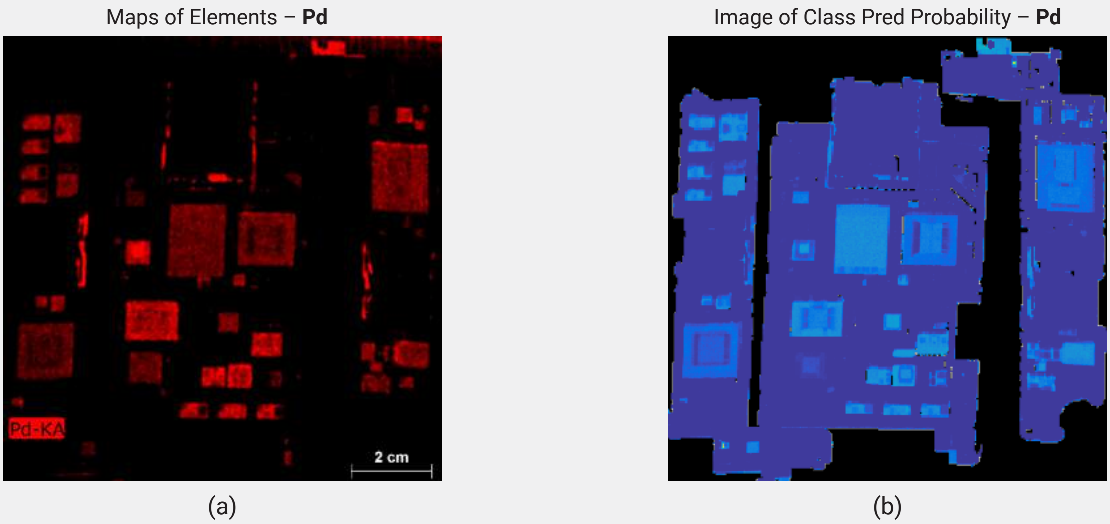



**FIGURE 8:** µXRF map of palladium (Pd) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

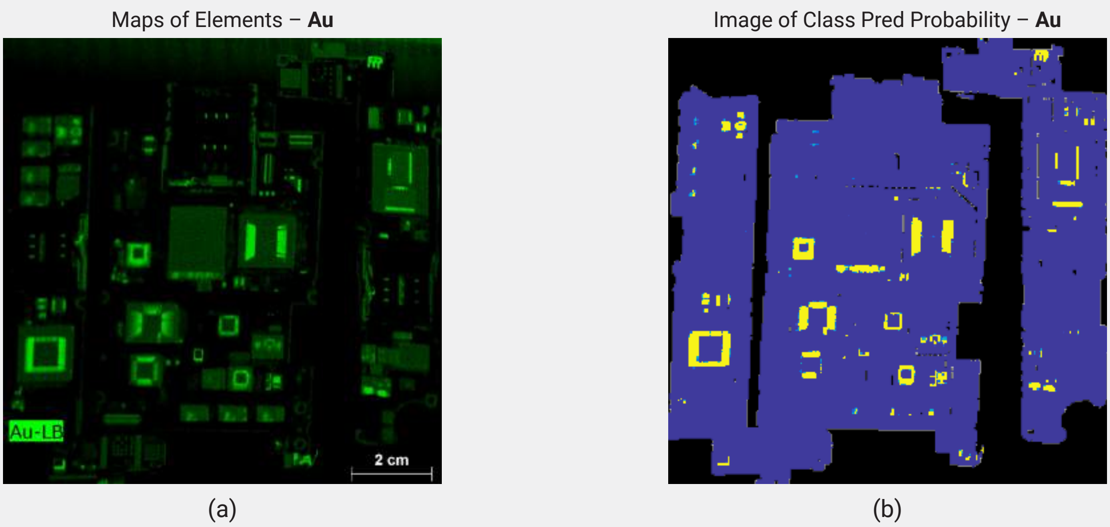



**FIGURE 9:** µXRF map of gold (Au) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

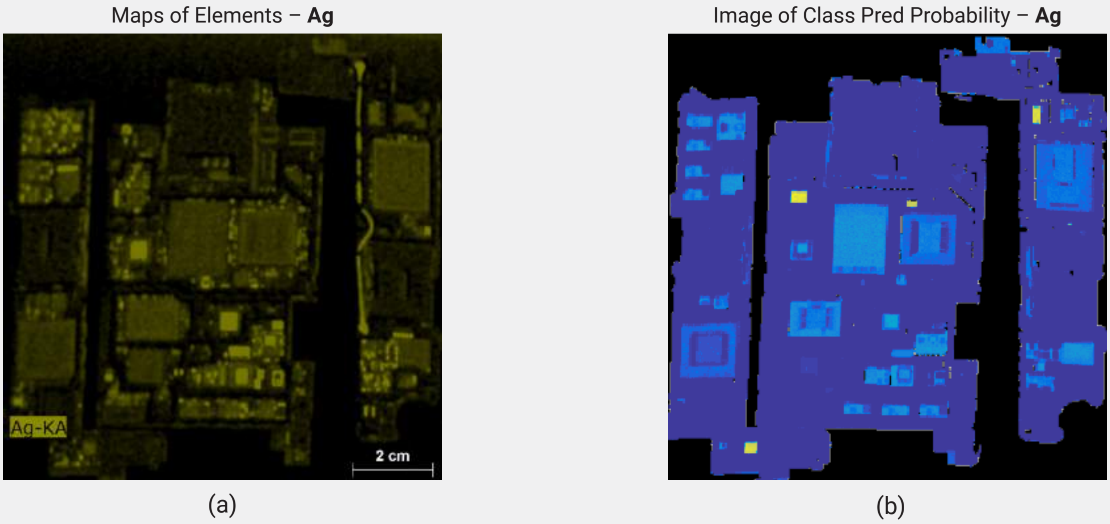



**FIGURE 10:** µXRF map of silver (Ag) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

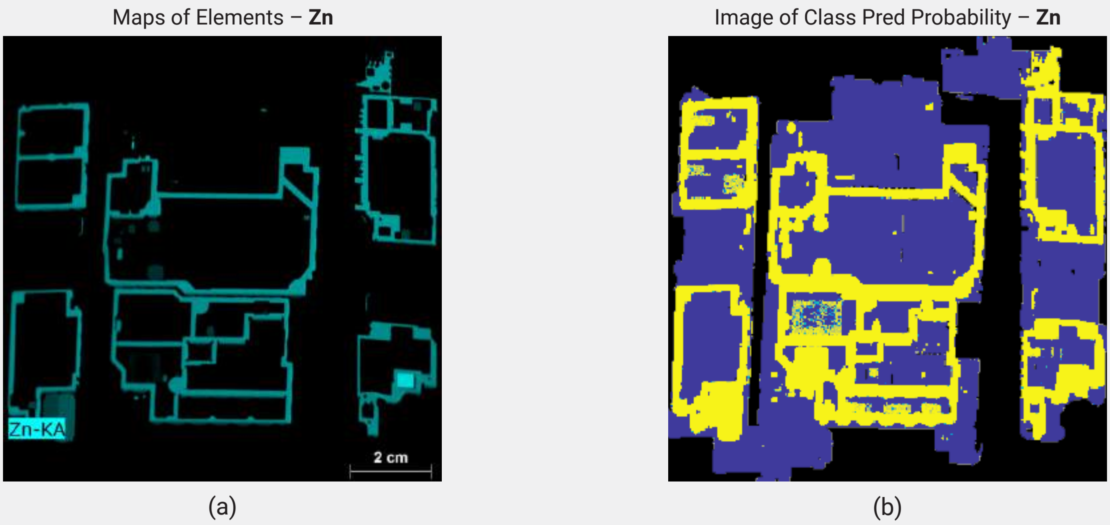



**FIGURE 11:** µXRF map of zinc (Zn) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

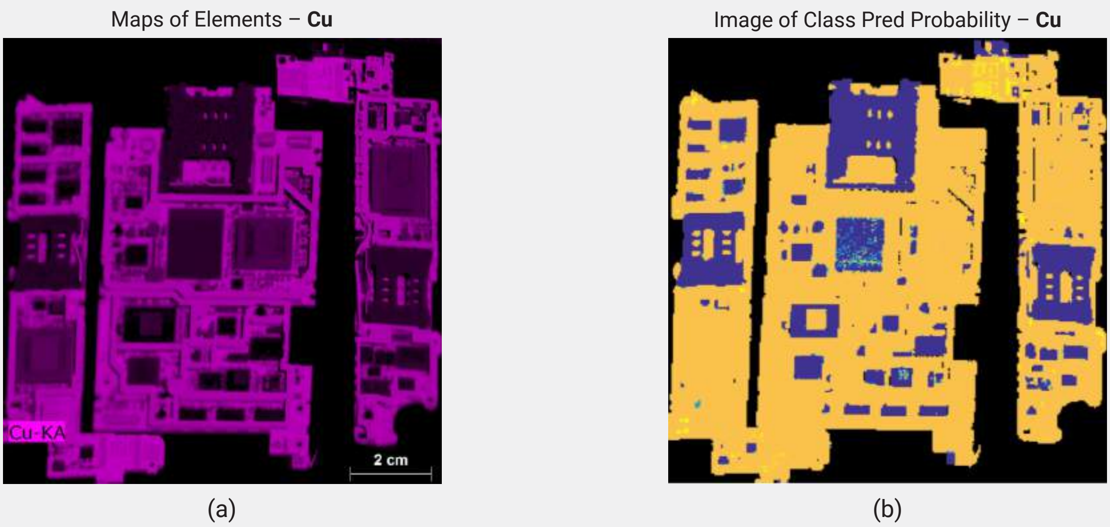



**FIGURE 12:** µXRF map of copper (Cu) element (a) following the classical analytical approach and as resulting from the PLS-DA prediction

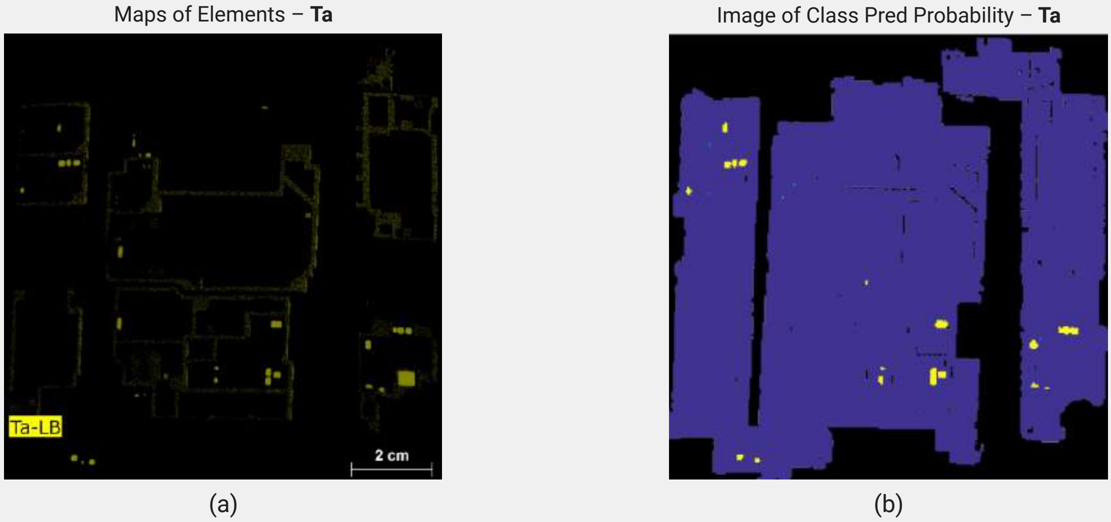



**FIGURE 13:** µXRF map of tantalum (Ta) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

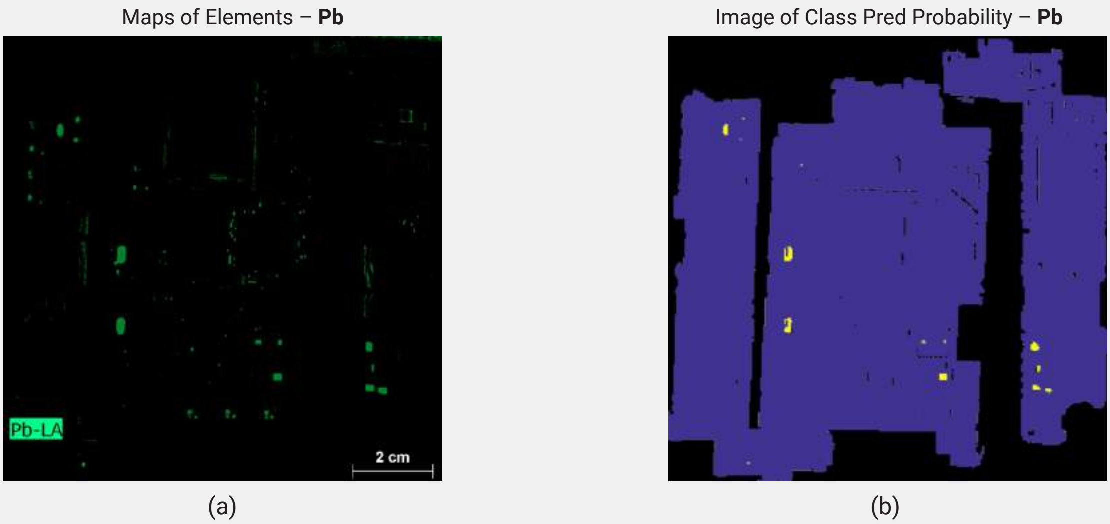



**FIGURE 14:** µXRF map of lead (Pb) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

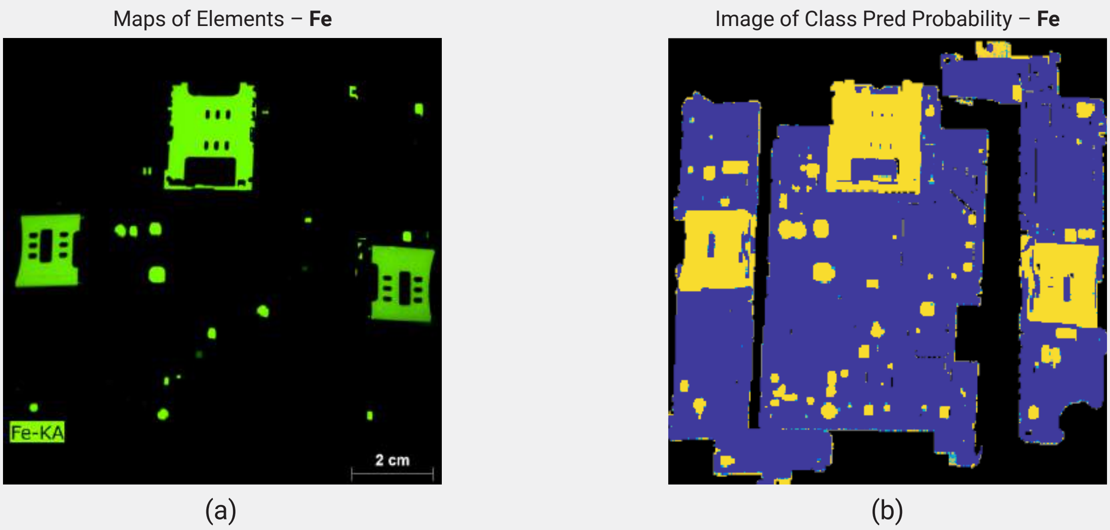



**FIGURE 15:** µXRF map of iron (Fe) element following the classical analytical approach (a) and the PLS-DA prediction modelling (b).

shows a correct identification in the electronic component characterized by a high lead presence. Its recognition is difficult in all those components of smaller dimensions, probably for the presence of other elements (i.e. Bismuth (La 10.839 KeV) whose peaks produce a masking effect (i.e. Pb La 10.551 KeV). (Figure 14b).

The map of iron (Fe) shows a high concentration of this element on card and security holders (Figure 15a). The prediction map, as resulting from PLS-DA modelling, produces its correct identification (Figure 15b).

## 4. CONCLUSIONS

The study was carried out to investigate the utilization of chemometric procedures, based on processing of data set generated by µXRF, in order to perform a laboratory scale preliminary (i.e. before mechanical-physical processing) automatic check of end-of-life (EOL) iPhone electronic boards characteristics (i.e. manufacturing and components presence), and related recovered products (i.e. particles) derived from processing. More in detail, PLS-DA, after PCA, was applied to build a model able to recognize/classify the precious and rare earth elements starting from the reference energy spectra representative of the different elements object of investigations.

The proposed combined chemometric-µXRF approach presents a lot of advantages: it is objective, it does not require any preliminary knowledge of the sample and it allows to assess, in a relative simple way, the quantity of precious and rare earth elements that is possible to extract by PCB derived products (i.e. iPhone electronic boards and products resulting from their mechanical processing).

The proposed prediction model performs a good classification. The procedure, after the preliminary model recognition set up, is easy to implement and it is characterized by low operative costs, being the procedure totally software, especially if compared with classical methods usually requiring sample chemical pre-treatment and longer analytical time (i.e. ICP-OES and SEM-EDX).

Further studies will be addressed to a systematic application of the proposed approach to particle resulting from comminution, classification and physical separation of dismissed iPhone boards, and more in general, PCB, in order to perform not only a qualitative control of the different flow streams, but also to set up pre-concentration actions finalized to separate particles characterized by different precious and rare earth elements composition and distribution. Following this approach, it will be thus possible to design more efficient and specialized strategies for final elements recovery by chemical processing.

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# ILLEGAL INTERNATIONAL TRADE OF E-WASTE - EUROPE

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**ABSTRACT**

The poor management of growing amounts of e-wastes has given rise to a series of consequences. One of the major consequences relates to illegal international trading of the wastes. This trade is a means for brokers, smugglers and organized crime groups to meet the demands of the waste producer for cheap disposal of used electronic products. The process is therefore frequently implemented through illegal means, with workers undertaking gross extraction processes in importing countries. The consequences of this include environmental harm, poor working conditions, yields for organized crime groups and loss of economic opportunities for exporting countries. This paper is focused mainly on e-wastes produced in Europe. To analyse and discuss a feasible solution to combat the illegal market, the article presents the three main reasons, i.e. European integration, corruption and ineffective customs inspections, underlying delivery of these wastes by criminals to the European continent, concluding that the best means of combating this trade is to apply a take-back system.

## 1. INTRODUCTION

Internationally, the most important legal instrument that describes electronic wastes (e-wastes) subjected to transboundary movements is the Basel Convention (BC). The latter movements relate to transport from an area under the national jurisdiction of one State to an area under the national jurisdiction of another State (UNEP, 1989). In terms of e-waste, this paper considers all types of discarded electrical and electronic equipment (EEE) (Adrian et al., 2014).

This type of waste is deemed hazardous by the BC mainly, in accordance with Annex I, due to the content of constituents such as mercury, copper, brominated flame-retardants and lead. Moreover, e-waste production has featured a steady growth, due to its fast obsolescence and increased production. It has been estimated that by the end of 2018, global production will have grown to 49.8 mega tons, i.e. , 6.7 kg/inhabitant, a 19% growth in relation to the 41.9 mega tons produced in 2014 (Baldé et al., 2015).

Therefore, this problem compels States to set up valid governance plan to cope with this issue because, since legal disposal is frequently expensive, disposers tend to seek cheaper solutions. One of these is the illegal international trading of the wastes and lack of relevant notification to all States concerned (UNEP, 1989).

These trades are concerning first and foremost as they are often outsourced to organized crime groups (OCGs) and small smugglers, both generally involved in other illegal activities in different parts of the world (Naim, 2006). Secondly, these trades result in gross handling of the waste and, hence, impact negatively on working conditions and the environment in importing countries (Geeraerts et al., 2015).

Accordingly, the BC was established with the aim of creating a legal responsibility for the European nations in terms of waste disposal and, thus, reducing illegal trading of the same and related consequences (UNEP, 1989).

Furthermore, the importance of analysing the illegal trading of e-wastes is linked to the challenges encountered in managing the wastes following inappropriate disposal, largely due to the complexity of the latter (UNEP, 2011). Governmental actions previously enacted in an attempt to thwart these transboundary movements are no longer effective (Naim, 2006). As a consequence, more adequate approaches should now be identified.

Based on these premises therefore, this paper concludes that the most feasible manner of addressing these trades is through the improvement of take-back systems for the collection and processing of e-waste either through direct regulation or by providing the necessary incentives (Mccann and Wittmann, 2015). To support the latter, section 5 will demonstrate how the European Integration, corruption – accepting bribes or undue advantages (Council

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of Europe, 1999), and ineffective customs inspections hamper the implementation of several governmental enforcements, consequently underlining the feasibility of adopting a take-back approach.

This approach is of considerable importance due to the complexity of e-waste transboundary movements (Bisschop, 2012) and, secondly, to the fact that concerted legal efforts are made to punish those involved in the illegal e-waste chain, European laws regarding transboundary movements of e-waste are not particularly effective. Moreover, the nature of environmental crimes itself is not punitive.

Moreover, due to the shortcomings in boundaries monitoring, there is no guarantee that enforcements, such as increase of trade bans, will succeed in coping with these trades (Rucevska et al., 2015). Hence, a focus on enforcements such as take-back systems, aimed at preventing the e-waste from reaching its international chain, is less costly.

Finally, it should be taken into account that the notion whereby illegal trades are merely a criminal issue is a misconception (Naim, 2006). It is also a matter of asymmetries in development and access to resources (Geeraerts et al., 2015). Consequently, the efficient solutions are those that counter the demand for these illegal services, such as take-back.

## 2. THE ILLEGAL INTERNATIONAL TRADE OF E-WASTE

By 2050 the waste sector is expected to employ between 23 and 26 million people, considering both legal and illegal activities (UNEP, 2011). This steady growth in the waste sector is concerning mainly in view of the crimes committed to illegally trade wastes, featuring a broad chain of legal operations, with criminals taking advantage of loopholes in control capacity (Rucevska et al., 2015).

These operations are executed by a series of agents and, with regard both legal and illegal operations, the e-waste chain is comprised of waste generators, waste collectors, waste management companies, transport and shipping companies, waste treatment operators, shipping agents, waste brokers, smugglers, small groups of people, and OCGs (Europol, 2015; Rucevska et al., 2015). In the case of these three last groups, particularly the latter, evidence has demonstrate bonds with the private and public sector, mainly in importing countries (Center for the Study of Democracy, 2012; Naim, 2006).

It should also be taken into account that illegal trades are facilitated by the cooperation established with both legitimate business, such as those in the financial, trade services and metal recycling industry; and illegitimate concerns, such as those specialized in document forgery for the acquisition of permits (Europol, 2011). However, even in the lack of a similar cooperation, legitimate businesses such as banks, carriers, lawyers and exchange offices (Bisschop, 2012) may participate unintentionally in this process (Naim, 2006). This unintentional nature however may be questionable, as the companies involved will know their customers and, hence, could use blacklists to avoid the transportation of illegal waste. Due to the cover provided by legal activities, this type of trade entails low risks of culminating in fines or prison, and results in the gaining of substantial profits (Bisschop, 2012).

When addressing the issue of why agents work in the e-waste chain, this should be viewed as a standard commodity, (Bisschop, 2012). In the case of a commodity, the producer sells the product to the consumer in return for money. In the case of e-waste, the producer provides both the waste and the money. This is a push factor for those involved to work on.

An additional push factor is represented by the fact that an increased environmental awareness has led to the development of new and more rigid laws, at the same time raising the costs of appropriate management of e-waste, primarily in developed countries (Bisschop, 2012). These circumstances have led to the creation of opportunities to get rid of the wastes quickly and cheaply, generally by means of illegal export (Naim, 2006). Italian companies, for instance, might pay about € 60.000 to legally dispose of a container of 15.000 tons of hazardous waste. Illegally, the same quantity could be disposed for approx. € 5.000 (Ciafani, 2012). These illegal exports therefore are linked to serious environmental crimes.

More specifically, agents who handle the e-wastes may vary according to factors such as place and quantity transported. OGCs, for instance, are generally more loosely structured than traditional mafia-like groups. Accordingly, small groups of up to ten people organized for a short period obtain financial benefits and then rapidly dissolve to form new groups (Europol, 2011; Geeraerts et al., 2015). Governmental enforcement plans also reveal how some criminal groups trafficking e-waste are involved in crimes related to human trafficking, fraud drugs, theft, firearms, and money laundering (Environmental Investigation Agency, 2011).

However, OCGs are not necessarily involved in the trafficking of e-wastes, with this role even being covered by other groups, including brokers. A pertinent type of broker is the waste tourist who, buys second-hand electronics and ships them to their relatives or business partners in developing countries. These people may be resident in the country of origin, carrying false passports and visas, and buy the products in thrift shops (Bisschop, 2012; Environmental Investigation Agency, 2011; Rucevska et al., 2015).

Furthermore, the widespread availability of Internet allows criminals to sell wastes on e-commerce websites. In Europe, over 70% of detentions relate to articles being shipped by express or postal services (European Commission, 2014). The high quantity of detentions in these services is a consequence of the high quantity of companies registered in free zones. These companies are neither public nor legally accessible, which makes it easier for them to hide illegal trades and delete evidence (Naim, 2006; Rucevska et al., 2015).

Finally, although these people may engage in illegal trades for several reasons, the main reason is related to economic aspects (Naim, 2006). For instance, in the Netherlands, second-hand televisions can be bought for US$ 4-5 each and sold in Africa for around US$ 10 per unit (Rucevska et al., 2015). Accordingly, the illegal disposal of e-waste economically attracts both the disposers and criminals who want to sell the waste. Moreover, low profitability of formal recyclers limits their financial ability to compete with informal collectors, who often purchase e-waste (Chi et al., 2011).

To conceal their illegal activity and products, actors in the e-waste chain adopt a series of methods (discussed below) to breach customs systems.

## 3. METHODS APPLIED TO BREACH CUSTOMS SYSTEMS

Article 4 of the Basel Convention gives parties the right to prohibit the import of hazardous waste and states that countries shall not permit its export. Ergo, to illegally trade in these wastes, criminals need to adopt methods to breach customs, notably in Europe, where all countries adhere to the convention (UNEP, 1989).

The two main methods brokers use is to mingle e-waste with legal materials and thus obtain a false classification (Rucevska et al., 2015). Together with this false classification, documents may also be forged (Naim, 2006).

In the strategy of mixing wastes, criminals attempt to hide illegal goods in the cargo, or, at least, hinder access to the same. Examples of this are doors of vehicles containing soldered e-waste (Rucevska et al., 2015). According to this strategy, criminals may even attempt to transport illegal wastes together with other illegal materials. For instance, INTERPOL (2015) has succeeded in seizing weapons concealed in illegally exported wastes in France.

In the case of false classification, it must be considered that, in the ambit of international trades, products are coded under a Harmonized System (HS), delineated by the World Customs Organization (WCO). Nonetheless, as Rucevska et al. (2015) stated, the HS does not encompass all existing wastes. Accordingly, the decision of whether the product traded is second-hand EEE or simply waste is a highly arbitrary decision, a situation that hampers the tasks of the inspectors (Geeraerts et al., 2015; Naim, 2006; Rucevska et al., 2015). Therefore, either by a lack of coverage in HS or by implementing an illegal trade, exporters may opt to provide a false declaration as to the nature of the waste or to use customs codes associated with goods falling outside the scope of the Basel Convention (Rucevska et al., 2015).

Exemplifying this situation empirically, the import of batteries and metal scrap mingled with other hazardous wastes (under the BC definition) to Indonesia (Japan Ministry of Environment, 2011) may be mentioned. Exporters used the code 7204 indicating ferrous waste and scrap; re-melting scrap ingots of iron or steel (Foreign Trade Online, 2018).

A cooperation has been set up between the Basel Convention Secretariat and the WCO to address this problem and to fill the loopholes in codes (Basel Convention, 2011). Nevertheless, since the main methods to breach customs systems are misclassification and the mixture of products, the creation of new codes would likely fail to constrain these breaches. Firstly, misclassification is not applied due to the lack of proper HS codes, but, rather, as an attempt to conceal goods. Therefore, it would still be possible to mingle the illegal e-waste with legal materials and trade them using the HS code of a legal material.

A series of governmental responses have been forthcoming with the aim of countering these methods. One of these is represented by the review of the WEEE Directive, adopted in June 7th, 2012, with specific regard to burden-of-proof. Following this revision, countries are able to request from the exporter evidence including a copy of the invoice and contract, to prove that the equipment is earmarked for direct re-use, and certificate of testing (European Commission, 2013). However, these documents are remarkably susceptible to forgery and corruption (Europol, 2011, 2015; INTERPOL, 2015; Naim, 2006).

When the illegal methods succeed, the exported waste generally reaches its destination; however, in addition to the waste, the trader also contributes to the creation of environmental and social problems (Vail, 2007). To analyse these issues, the subsequent section will focus on the waste importers and consequences produced by these trades, particularly with a view to promoting the application of take-back systems.

## 4. E-WASTE IMPORTERS AND CONSEQUENCES OF THIS TRADE

In agreement with the literature on illegal international trade of electronic waste, including those originated in Europe, the major destinations are Africa and Asia (Bisschop, 2012; European Environmental Agency, 2012; Geeraerts et al., 2015; Li et al., 2014; Lundgren, 2012; Rekenkamer and Voorhout, 2013; Rucevska et al., 2015). In general, small-scale exports are destined for West Africa, whilst the larger and sometimes more structurally organized transports are directed to South-East Asia (Lundgren, 2012).

Geeraerts et al. (2015) and Rekenkamer and Voorhout (2013) have indeed pointed out that the majority of European e-waste sent to Asia ends up in China. This is largely intended to boost the demand for raw materials created by a rapid economic growth in these importing countries (European Environmetanl Agency, 2012). E-waste is a valuable source of raw materials and China, as Early (2013) pointed out, controls approx. 70% of the global recycling market, a fact that is highly attractive for the e-waste market in this country.

However, due to the preferential status of a handful of countries for e-waste recycling, some, such as China, tend to strictly monitor the situation. In an attempt to overcome these monitoring processes, exporters have been seen to avoid the most common international flows of e-waste and use other sites as intermediaries to alleviate the suspicion of illegality (Geeraerts et al., 2015). For instance, Lundgren (2012) states that exporters often use Hong Kong, Taipei or the Philippines as entering sites and then transit the e-waste to smaller ports in China. Correspondingly, Dubai and Singapore also serve as intermediaries for the same purpose (Kalra, 2004).

On the other hand, the Chinese economic growth may also turn the tables and place the country as an e-waste exporter, since Chinese consumers increasingly buy new

EEE instead of second-hand products. As a consequence, African brokers go to China to collect second-hand EEE and ship them to African countries (Geeraerts et al., 2015).

Another source of attraction for the import of e-waste is the profit that informal recyclers make by dismantling these wastes. From this process, they extract precious metals such as gold, copper, nickel and rare materials, such as indium and palladium (Lundgren, 2012; Rucevska et al., 2015), thus creating a demand for the waste in both exporting and importing countries. The former feature a demand to get rid of waste cheaply, and the latter a demand to obtain revenue from waste by dismantling it.

In accordance with Geeraerts et al. (2015), exportation of the waste will result in an economic loss to the nations and enterprises that generate the waste. Sound recycling of 1 million cell phones can recover about 24 kg (50 lb) of gold, 250 kg (550 lb) of silver, 9 kg (20 lb) of palladium, and more than 9,000 kg (20,000 lb) of copper (Electronics Take-Back Coalition, 2014). Moreover, literature studies and data presented by Bisschop (2012) have reported how legal extraction is capable of achieving a 500% higher efficiency in terms of quantity of materials extracted, being able to extract approximately 280% more gold from a mobile phone as demonstrated , respectively, in part A and B in Figure 1.

Following the extraction of components from e-waste, these can easily be restored to a legal status, as the complexity of the chain makes it extremely difficult to learn the actual origin of the gold or copper extracted. Furthermore, the workers involved often have bonds with the manufacturing industry to sell the extracted materials (Geeraerts et al., 2015).

Considering the issue of workers, it is important to highlight that although for some this has become a lucrative industry, others it has served to reinforce inequalities, which intersect gender, race, class and age (Geeraerts et al., 2015). In terms of human health, Li et al. (2014) stated that these e-waste disposals, particularly in China, are responsible for the introduction of large amounts of pollutants into the air, drinking water, and food supply. With regard to working conditions, Pickren (2014) and Wang et al. (2013) affirmed that the majority of recycling labourers are rural migrants from outlying agrarian regions who have informal and precarious jobs and receive around $1.5 per day, many of whom women and children.

Furthermore, the environment is impacted by the consequences of this waste mainly due to the gross recycling methods used, which include:

- Heating circuit boards by blowtorch method (Puckett et al., 2002);
- Stripping of metals in open-pit acid baths to recover gold and other metals (Wong et al., 2007);
- Open-air burning of cables in order to recover copper and burning unwanted materials (Wong et al., 2007).

However, although informal and gross methods are much less effective, Chi et al. (2011) assert that they are highly 'cost-efficient' due to the use of non-skilled manual labour, and disregard any hazards to environment or health. Moreover, these informal practices contribute to the release of toxic metals and, consequently, expose workers to acids, lead and toxins released from burned debris (Naim, 2006).

Briefly, this analysis demonstrates, as mentioned previously, that the e-waste business reflects both the economic and social realities of different countries, and not only a criminal issue apropos of OCGs, small smugglers or brokers (Naim, 2006).

To analyse why these illegal trades are conducted in Europe, the next section will discuss three specific reasons and relate them to use of a take-back system.

## 5. REASONS UNDERLYING THE ILLEGAL TRADE

In addition to the above-mentioned reasons underlying the illegal international trade of e-waste, this paper will hereafter focus on three reasons encountered in a European context. This additional analysis will also serve as the groundwork to justify implementation of a take-back system. Firstly, European integration will be examined, followed by corruption and, finally, ineffective customs inspections.

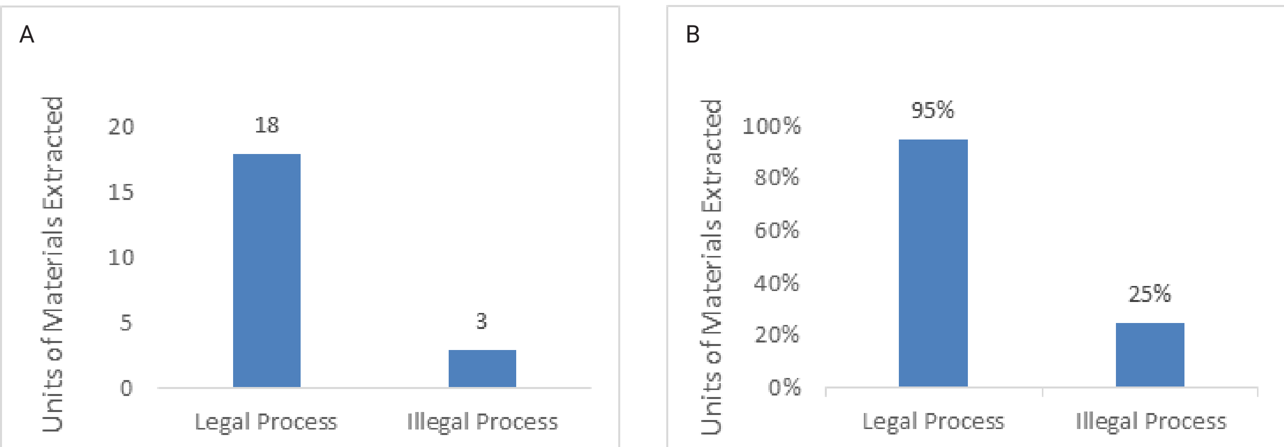


**FIGURE 1:** Extraction of materials from mobile phones and percentage of gold extracted from mobile phones - Adapted from Bisschop (2012).

### 5.1 The European Integration Process

The European Union (EU) is a customs union. According to the European Commission (2014), a customs union is created when a group of countries joins together to apply the same rates on import duties from the rest of the world. Additionally, the EU applies a wide set of common rules to imports and exports, and has completely removed all controls between member states (European Commission, 2014).

In legal terms, The Treaty on the Functioning of the European Union (TFEU) enshrines this removal of control and free movement. In Part Three, Title I, Article 26 the treaty states that the internal market shall comprise an area without internal frontiers in which the free movement of goods, persons, services and capital is ensured in accordance with the provisions of the Treaties (Official Journal of the European Union, 2008). However, despite this removal of barriers, inspections may be implemented between internal borders (Center for the Study of Democracy, 2012).

Consequently, although valuable for free trade between countries, this integration process may at times facilitate the action of e-waste brokers. IMPEL-TFS (2013) points out, on the one hand, that this process provides a lot of shared information between inspectors and organizations in the ambit of the EU. However, on the other hand, Rekenkamer and Voorhout (2013) and Geeraerts et al. (2015) state that the involvement of multiple organizations creates challenges with regard to enforcement, underlining how many Member States (MS) do not have well-trained staff, technical equipment or money to implement these inspections or enforcement. Indeed, even nations with more resources may face financial limitations and staffing issues (Lundgren, 2012). Moreover, the broad definition of waste used by the Waste Shipment Regulation (WSR) (internalization of BC into EU) may limit information sharing (Geeraerts et al., 2015).

Additionally, the prosecution of environmental crimes remains a national competence and MS do not have the same level of enforcement (Bisschop, 2012). This in turn leads to a process referred to by the cited author, port hopping, implying that brokers may choose ports in which controls tend to be less stringent.

For instance, the Netherlands is one of the busiest e-waste exports hubs in the EU and, consequently, is considered to have good enforcement policies (Geeraerts et al., 2015; Rekenkamer and Voorhout, 2013). In response to this, agents tend to look for ports in other countries.

As reported by Naím (2006), this process hampers the tracking of illegal cargoes, furthermore resulting in corruption at Border Crossing Points (BCP).. However, one important form of corruption with regard to the practice of port hopping is the overlooking of travel bans, as described by the Center for the Study of Democracy (2012), thus allowing criminals to move more freely throughout Europe.

The above issue however is aggravated by the fact that the legal system fails to prosecute all environmental crimes, with only a handful of countries having specific prosecutors for these sorts of crimes (Eurojust, 2014). Accordingly, in terms of WSR violation, prison sentences are very rare. In most cases, the offender is either fined or the charges are dropped (Rekenkamer and Voorhout, 2013). This may occur as waste crimes are frequently regarded as victimless crimes, which in most cases leads to waste crimes going unreported (Baird et al., 2014). Therefore, the loopholes in EU enforcement and legislation also act as driving forces of e-waste illegal trades (Geeraerts et al., 2015; Lundgren, 2012).

Accordingly, some ports may evolve into ports of transit, such as the port of Antwerp (Bisschop, 2012). The majority of waste handled at the port is in transit from countries such as Germany, Austria, Switzerland, France and the Netherlands. Antwerp is moreover used as an intermediary due to the presence of limited staff and limited availability of resources (Bisschop, 2016). Moreover, the author affirms that port hopping occurs largely between the ports of Rotterdam, Antwerp, Hamburg, Felixstowe, Le Havre and Bilbao. Following this process, the waste is forwarded to its main destination: Africa and Asia.

The occurrence of crimes in European countries is a crucial issue as the EU produces a substantial part of global e-waste. According to Baldé et al. (2015), this represented 22.72% in 2014. In addition, only one third of WEEE is appropriately disposed of in the EU, either in the country of origin or in other states (Eurostat, 2018). The remaining wastes might be collected by unregistered enterprises and properly or improperly treated or even illegally exported abroad (Eurostat, 2018).

The issue of European integration is likewise of importance in view of the contribution provided by waste transport policies and practices within individual nations and throughout the EU to the phenomenon of illegal waste shipment (Vail, 2007). Indeed, a tougher approach to recycling or treatment by the individual European countries may even encourage illegal shipment rather than stimulating appropriate management, particularly due to the relatively free flow of goods.

It is likewise important that, since illegal e-waste trades are a global problem, it was hoped that a European regionalization would be able to better identify the best solutions for the problem, since the efforts made would involve global measures rather than merely local and national actions (Naim, 2006). Nevertheless, as discussed above, this integration process may also facilitate an illegal trading in Europe, which produces a significant share of global e-waste.

Concisely, these loopholes, related mainly to inequality in enforcement among European countries, are one of the reasons for advocating the use of take-back systems as an efficient manner of combating illegal trades.

Finally, it should be borne in mind that not all illegal trades benefit from this integration process and relative free movement of goods. However, they may still be susceptible to corruption, as discussed in the next section.

### 5.2 Corruption

Corruption involves both legal and illegal activities in the e-waste chain and relates to both exporting and importing countries. Corruption involves a variety of individuals, including border guards, customs officials and port operators (Chêne, 2013). As reported by Chêne (2013), there is a broad consensus in the literature that port and border corruption may exert a detrimental impact on shipping costs and OCGs.

Sequeira and Djankov (2013) divided public officials in charge of public services according to the possibility of participation in collusive or coercive corruption. The first was related to the division of rent generated by an illicit transaction between public and private agents, and the second to the payment of an additional fee in order to gain privileges.

These two types of corruption produced a series of different reactions amongst the different agents. Some legal firms, for instance, are willing to travel additional distances to avoid coercive corruption at ports, chiefly as it may raise the cost of products (Sequeira and Djankov, 2013). Illegal businesses, however, tend to look mainly for collusive corruption, firstly because it is related to illicit transactions and, secondly, because it may represent a means of avoiding physical inspection of containers (Chêne, 2013; Sequeira and Djankov, 2013). Hence, it is an opportunity to reduce the possibility of customs discovering illegal activities.

With regard to OCGs, as stated previously, these groups commonly use intermediaries and native people who are better acquainted with the situation of the country in order to make corruption more effective. This outsourcing allows them to quickly withdraw their names from the transactions (Center for the Study of Democracy, 2012; Naim, 2006). In addition, a specific category of intermediaries is comprised of legitimate logistics and professional service experts, some of whom are employed (willingly or otherwise) by organized criminals to bribe border guards (Center for the Study of Democracy, 2012).

In the specific case of e-waste transport, corruption may include bribery, cybercrime, document forgery, identity theft and use of intimidation and violence (Geeraerts et al., 2015). During the 2000s, customs agencies around the world established a series of inspectorates to fight corruption. However, whilst they received no reward for fighting corruption, they were at risk of being 'rewarded' with death by OCGs (Michael and Moore, 2010).

As a result, according to the findings of the analysis of the European Integration process, corruption may also be capable of turning the country involved into an intermediary hub of e-waste export. One example of this is Italy. As a consequence of corruption, both in the public and the private sectors, mainly in the issuing of false certificates by laboratory technicians, the country has become a transit site of e-waste to Africa and Asia (Europol, 2011).

Illegal actions committed by border guards fall into categories including the sale of information, overlooking of travel bans, provision of false alibis and obstruction of investigations either actively or in a more passive manner. Active involvement could entail the providing of information about patrols, for instance, whilst passive involvement may relate to overlooking the presence of illicit goods after receiving bribes (Center for the Study of Democracy, 2012). By acting thus, these public agents clearly promote the action of OGCs, small smugglers and other agents in the illicit e-waste chain.

With regard to importing countries, the following factors should be taken into account in order to better understand corruption: presence of weak institutions, poor governance, under-resourced customs, operations in geographically dispersed places, lack of supervision, lack of training, low level of automation and limited staff (Chêne, 2013). In many African ports, for example, Omondi (2007) demonstrated how certifications and valuations are hugely prone to corruption, with bribes frequently being based on the consignment value. Moreover, in many of the importing countries, agents involved in the illicit chain may infiltrate the bureaucracies (Naim, 2006).

A similar form of corruption is also present in Asian countries, where governments are at times complicit in the actions undertaken by OCGs (Naim, 2006). Indeed, in the aftermath of attestation of the latter, the government of China created a rotation system of officials along the border with Vietnam (Geeraerts et al., 2015).

To counteract these problems, literature reports relating to corruption and anti-corruption in the customs area maintain that technology is one of the best means of achieving this goal, particularly as the processes would subsequently be automated (Michael and Moore, 2010; WCO, 2003). However, this specific use of technology may also aid the work of criminals, as demonstrated by the application of e-commerce described in section 2 of this paper. Moreover, technology has contributed towards a considerable geographical expansion of these illegal markets. Lastly, those operating illegally are often more flexible than governments and, consequently, are more willing to to take advantage of the benefits provided by technology (Naim, 2006). This in turn implies that automation may prove beneficial to both sides, and frequently may particularly enhance the work of those involved in illegal activities.

An additional factor heavily implicated in combating corruption is related to the raising of customs barriers or liberalization of trade. In the first case, the scarce flexibility of the waste should be taken into consideration. The use of barriers tends to raise the price of e-waste, whilst the market demand remains relatively unaffected (Baird et al., 2014). Additionally, illicit trades expand insofar as the profits increase. Consequently, in the presence of additional barriers, the traders tend to receive greater profits (Naim, 2006), due to the relative inflexibility of the product and the scarce effectiveness of customs barriers in hampering these trades.

Conversely, the onset of free trade, which may contribute towards reducing collusive corruption, largely due to the removal of tariffs (Chêne, 2013), would likely result in a decrease in the profits of illegal e-waste trades (Naim, 2006). However, Sequeira (2013) reported that liberalization may also be capable of replacing corruption by applying coercive methods to perform routine processes. Furthermore, although liberalization of trade may indeed result in a decrease in the illegal gains, this would in turn render customs more pervious to these crimes.

In addition to the aspects discussed above, other issues relate to the creation of codes of conduct, promotion of campaigns against corruption and customs investigations. Very few anti-corruption expert have ever been able to produce firm evidence demonstrating that the outcomes of these actions aimed at combating corruption, have outweighed their costs (Michael and Moore, 2010). Accordingly, the authors affirm that customs officers are rarely subjected to disciplinary actions following a breach of the code of conduct due to a somewhat abstract formulation of the former.

Consequently, corruption further underlines the need to implement effective take-back systems. Despite the presence of numerous anti-corruption programs and campaigns throughout customs, as demonstrated, the national governments are not able to counteract illicit trades (Naim, 2006).

In a nutshell, efforts such as campaigns against corruption and creation of codes of conducts should continue in the fight against corruption. However, based on the arguments presented, mainly relating to the lack of efficiency of these programs, it would be more appropriate to focus increasingly on take-back systems. This would undoubtedly represent a more effective way of preventing corruption and impeding the entry of e-waste into the international chain.

Analogue to the loopholes in the European integration, corruption will not always prove beneficial to all cases of illegal trading of e-waste. However, at any given time, the illegal activities will undoubtedly take advantage of the subject discussed in the next section: ineffective customs inspections. A scarce evolution in the efficiency of inspections throughout Europe indeed further supports the use of a take-back system.

### 5.3 Inefficiency in Customs Inspections

Given that a huge quantity of products pass everyday through customs worldwide, it is impossible for custom officers to inspect all shipments. For this reason, based on the methods illustrated previously in part three, it may be possible for illegal shipments of e-waste to pass through customs without being subjected to inspection.

This has been empirically demonstrated in Europe by the IMPEL data. Periodically, the organization performs Enforcement Actions (EA) to gather data relating to inspections as shown in Table 1 (IMPEL-TFS, 2011, 2013, 2015).

The actions undertaken in 2011 and 2013 focused solely on the physical inspections as part of the analysis. However, data illustrated in Table 1 has been adapted in line with the IMPEL reports, to consider both physical and administrative inspections. Data analysis failed to identify an improved efficiency in the inspection of e-waste shipments, although some countries have started to use data and other intelligence in preparing for inspections, as recommended by IMPEL-TFS (2015).

In EA II, for instance, 14.59% of all inspections related to waste, with 21.37% revealing a breach of some description. This detection of violations by waste shipments increased by approx. 10.6% from EA II to EA III. However, in EA IV, although the percentage of waste inspections had increased by 14.55% of all inspections conducted compared to EA III, the percentage of breaches detected had fallen by 15.42%.

Hence, despite the possibility of e-waste shipments undergoing inspection based on the findings of intelligence resources, data obtained in Europe continue to evidence a lack of customs efficiency. This may be linked to a problem with funding, as mentioned in section 5.1. Indeed, in spite of the market availability of new technologies to assist in the efficient inspection of cargoes, IMPEL-TFS (2011, 2013, 2015) has demonstrated that not all countries have access to sufficient resources to allow for a consistent carrying out of inspections.

In terms of technologies applied to improve customs inspections, with regard to e-wastes, some of these may give rise to controversy. For example, non-intrusive inspection equipment using x-ray and gamma-ray technologies are being deployed at border crossings and sea and airports to reduce the time taken in examining cargo shipments (European Commission, 2014).

Although useful for other crimes, in many cases x-ray and gamma ray are not specific enough to uncover the concealing of e-waste by pretending it is second-hand material. Moreover, as demonstrated in section 5.1 and above, not all countries have sufficient funding to implement these methods.

The inefficiencies highlighted are crucial, as until loopholes allowing the free trade of wastes labelled for recycling, coupled with weak enforcement procedures, are closed, there will continue to be a high probability of successful illegal transport (Vail, 2007).

All the aforementioned arguments demonstrate how even in the presence of improved monitoring and inspections, customs will still not be in a position to detect a substantial quantity of e-wastes illegally traded; this is largely due, to reasons such as the intense flow of products transiting through countries, inefficiency in inspections, shortfall of funding and inefficiency in targeting cargoes. Consequently, it would be a significantly more cost efficient solution to prevent wastes from reaching this point by reducing the flow of e-wastes and investing increasingly in take-back systems.

## 6. DISCUSSION

To enhance the understanding of the results of this study, a multidisciplinary approach should be applied. International trade is fundamental as it allows brokers to dispose of a huge quantity of illegal e-wastes. Further, a

**TABLE 1:** Waste violations detected by customs inspections in the European Union.

| | Enforcement Action II 2008-2010 (IMPEL-TFS, 2011) | Enforcement Action III 2012-2013 (IMPEL-TFS, 2013) | Enforcement Action IV 2014-2015 (IMPEL-TFS, 2015) |
|---|---|---|---|
| **Total Inspections** | 26705 | 22414 | 17183 |
| **Waste Inspections** | 3897 (14.59%) | 3162 (14.1%) | 4923 (28.65%) |
| **Waste Violating WSR** | 833 (21.37%) | 1011 (31.97%) | 815 (16.55%) |

detailed analysis of the trade has confirmed the unfeasibility of expecting customs to be effective in identifying all e-wastes subjected to illegal trading.

Additionally, the EU itself may unwittingly promote this illegal trade by allowing a relatively free flow of goods. Briefly, these considerations provided valuable insights into the disposal of e-waste through illegal trade routes.

Based on this multidisciplinary feature, we conclude that the best way for countries to counteract illegal e-waste trades is by preventing influx of these wastes into the international chain; indeed, once the wastes reach the distribution chain governmental enforcement systems are called upon to cope with much more complex scenarios. These scenarios highlight a need for international cooperation, increased financial availability and increased efforts. To conclude, in spite of the failures of the take-back system, the investment of capital aimed at improving the system, and widespread application of the same would undoubtedly prove to be the most feasible measure in combating these illegal trades.

Of course, there may be other means of preventing the entry of e-waste into the international chain. However, these means would necessarily imply considerable changes in economic dynamics, such as decrease of production and consumption of EEE, and, consequently, be associated with a need for long-term changes.

Long-term changes may likewise be required to improve the take-back system. That might also be true. However, as explained previously, valuable materials may be extracted from e-waste, and consequently, public policies implemented in this context may serve to stimulate enterprises to collect wastes, profit from collection and avoid a huge part of the impacts caused by the illegal international trading of the same. An empirical example is provided by Apple, which in 2015 recovered 2204 pounds of gold and 6612 of silver via take-back initiatives. The value of these extractions was pegged at $ 40 million (Szathmary, 2016).

Based on the investigations undertaken in this study however, any public policies implemented will need to make participation in take-back systems more profitable for the brokers than selling the wastes illegally. A successful empirical example is the Chinese Home Appliance Old for New Rebate Program. The program involved the setting up of a governmental fund for recovery; through this fund, people would get a ten percent discount on a new home appliance on delivering an old appliance to an authorized collection company (China, 2009). According to Rucevska et al. (2015), twenty months into the program, 49.9 million obsolete home appliances had been collected. Additionally, this process raised the sales of new products because the governmental funds allowed enterprises to sell with discounts and still profit.

## 7. CONCLUSIONS

The importance of studying the illegal international trades of e-waste is irrefutable, notably when considering those generated in Europe, where only one third of these wastes are appropriately managed (Eurostat, 2018).

Specifically, the multidisciplinary analysis of this article provides relevant insights to the subject by filling gaps in the literature with regard to this issue. Indeed, future research should focus on the definition of an economically feasible take-back approach for both the national authorities and disposers. This would consequently serve to attract government-certified agents rather than illegal brokers, with the agents or companies being in a position to profit from the legal extraction of valuable materials from e-wastes, and consequent reduction in the illegal trading of these wastes.


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# ORGANIC WASTE AND BIOELECTROCHEMICAL SYSTEMS: A FUTURE INTERFACE BETWEEN ELECTRICITY AND METHANE DISTRIBUTION GRIDS

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**ABSTRACT**
In a very near future, renewable electricity produced by photovoltaic and eolic is destined to be the cheapest form of energy. As these sources can't be constant in time, new industrial research challenges have already been shifted to electricity storage from the grid. Here we present an innovative concept of electricity storage system, based on the field of bioelectrochemical systems. Electromethanogenesis is one of the most recent applications in this field, where methanogenic microorganisms of the Archaea domain can fix $CO_2$ to methane, under electrical stimulation. In other words, electricity can be efficiently converted into $CH_4$, i.e. one of the most commonly used fuels, territorially-distributed with a capillary grid in most EU-Countries. What is needed, to implement this process, is a relatively concentrated source of $CO_2$ in an anaerobic acqueous environment. Currently in our society, huge concentrated streams of $CO_2$ are released into the atmosphere every day from wastewater and waste treatment facilities, as well as from landfills. To treat sewage and organic waste, organic matter is degraded to inorganic carbon, mainly by microbial oxidation processes, which are strongly energy-intensive. In perspective, every wastewater treatment, anaerobic digestion, organic waste composting facility and controlled ladfill could be a key hotspot to transform excess grid electricity into biomethane, while treating waste with the same energy. Biomethane could be injected to the distribution grid and the waste-management facilities would become the interface between the two grids. To achieve this scenario, efforts in scaling up electromethanogenesis systems and new bioelectrodes materials (e.g. electro-active biochar) are needed. Here, we summarize some key steps in this field of research and the constraints that are to be overcome.

## 1. INTRODUCTION

In the near future, photovoltaics and wind turbines will be considered as primary sources of energy. Their cost and sustainability will soon stably meet the grid parity. Electricity storage capacity will be the real challenge, to buffer intermittent productions and consumptions (Breyer and Gerlach, 2013).

Traditional electricity storage capacities rely mainly on hydropower facilities (pumping). Batteries life cycle and sustainability have already been improving and their costs are also decreasing exponentially. Other technologies and solutions are urgently needed as alternatives to widen the spectrum of possible storage capacity for the electrical grid. Among a range of possible technological solutions, power-to-gas technologies have attracted attention. Water electrolysis to produce gaseous hydrogen as chemical energy vector (power-to-gas, P2G) has been proposed as the solution, to be coupled to fuel cell to reconvert $H_2$ into electricity. The abiotic electrocatalysis of the reaction $2H^+ + 2e^- \rightarrow H_2$ works theoretically at a cathodic potential of -0.410 V vs Standard Hydrogen Electrode (SHE, pH=7). However, in applications with high current-densities, according to the catalyst and the electrode properties, this reaction may require much lower potentials (-0.7 – 1 V vs SHE), due to consistent overpotentials of both cathodic and anodic reactions (Zoulias and Varkaraki, 2004). Also, safely handling,

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transporting and storing molecular hydrogen is still a technological unresolved challenge and too expensive to think about applications in the near future (Gahleitner, 2013).

An alternative option for more realistic applications is power-to-methane. The conversion of electrical current into methane (i.e. $CO_2$ methanation) has been for long proposed as alternative, considering that many Countries already count on a capillary methane distribution grid (Zoss et al., 2016). High-temperature and pressure catalytic power-to-methane conversions are the state of the art, but they encounter serious constraints in P2G applications: a) relatively small-scale plants are too expensive; intermittent use, as needed for day-night grid variations, are not viable; and large amounts of high purity $CO_2$ gas streams are required to avoid hindering the metal catalysts (Götz et al., 2016). Also, the territorially distributed availability of concentrated and pure $CO_2$ streams is not guaranteed, if we exclude fossil-based power plants.

Biological methanation has been proposed as an alternative for smaller-scale applications (Götz et al., 2016). Several methanogenic microbes of the domain Archaea are known to catalyze the reduction of of $CO_2$ to methane (autotrophic methanogenesis), in strictly anaerobic environments (Bapteste et al., 2005). Hydrogenotrophic methanogenesis, for example, has been longely known and it is widely used in full-scale anaerobic digestion (AD) facilities as one of the two main metabolic routes towards methane production from biomass (Hara et al., 2013; Premier et al., 2013). Electro-trophic methanogenesis (or electro-methanogenesis) was also recently discovered as an alternative path towards $CO_2$ reduction (Cheng et al., 2009a). Different innovative biotechnological applications of these biochemical pathway were proposed as options for methane generations by reduction of $CO_2$ through electrostimulation of methanogenic microbial communities in bioelectrochemical systems (BES) (Blasco-Gómez et al., 2017).

In this article, we discuss the future challenges and potentials of this innovative perspective and we propose a scenario for the integration of such technology in AD process.

## 2. BIOLOGICAL POWER-TO-GAS

Methanogenic microbes of the domain Archaea can catalyze the conversion of $H_2$ and $CO_2$ to methane (hydrogenotrophic methanogenesis), in strictly anaerobic environments. Electrochemical $H_2$ evolution by water electrolysis and subsequent $H_2$-sparging in anaerobic digesters (where the environment is saturated with biogenic $CO_2$) could be a smart solution for P2G at a territorial scale. Today, the EU counts on a year-by-year increasing number of anaerobic digestion plants sufficiently distributed on the territory (EURObserv'ER, 2014). Biogas upgrading to biomethane for injection to the methane grid is also a reality, both under technological and regulatory point of view (EURObserv'ER, 2014). In this scenario, the organic matter contained in waste or wastewater streams would be an inexpensive source of concentrated $CO_2$, which can be directly converted into biomethane by room-temperature/pressures and easily scalable processes.

Unfortunately, the first step of this transformation chain (i.e. water electrolysis in electrochemical cells, based on abiotic catalysts) is a relatively inefficient process. At cathodic potentials in the range -1 – -1.5 V vs SHE and current densities in the order of 1-10 kA/m$^2$, the electricity-$H_2$ conversion efficiency is currently around 4-5 kWh/Nm$^3_{H2}$. This is due to the high overpotentials of the cathodic electrocatalysis at such high current densities (Zoulias and Varkaraki, 2004). Even considering a stoichiometric conversion of this $H_2$ to methane:

$$4H_2 + CO_2 \rightarrow 2H_2O + CH_4$$

the electricity-$CH_4$ conversion efficiency would result of 16-20 kWh/Nm$^3$$CH_4$. Additionally, the final conversion efficiency would be even lower due to the low solubility in water of $H_2$ (Bo et al., 2014).

Also, water electrolysis suffers of poor efficiency in the counter reactions, at the anode. The use of water as electron donor, with $O_2$ evolution, is definitely not a thermodynamically favorable reaction (+0.82 V vs SHE, pH=7, Figure 1).

## 3. ELECTROMETHANOGENESIS (OR BIOELECTROCHEMICAL METHANATION)

To overcome these constraints, a new generation of biological methanation process was recently (year 2009) introduced and called Electromethanogenesis (Blasco-Gómez et al., 2017; Cheng et al., 2009b). It results from the integration of electrochemical systems and microbial autotrophic methanogenesis (hydrogenotrophic route). Electroactive microbial communities, grown as biofilms on solid electrodes, were demonstrated to be able of direct electron transfer towards the fixation of inorganic carbon to methane (Cheng et al., 2009b), following the reaction:

$$CO_2 + 8H^+ + 8e^- \rightarrow CH_4 + 2H_2O$$

This reaction theoretically happens at a cathodic potential of -0.224 V vs SHE, i.e. it takes half of the energy, with respect to water electrolysis (Rabaey and Rozendal, 2010). This is well represented in Figure 1.

The most common microbial species able to perform this reaction belong to the Archaea domain and are normally found in regular anaerobic sludge in biogas-producing facilities. Direct electron transfers towards inorganic carbon fixation to methane, in fact, was demonstrated as a mechanism that happens in natural anaerobic environments, between different microbial species. The so-called DIET (direct interspecies electron transfer), mediated by membrane bound proteins and conductive extracellular filaments (called e-pili), was demonstrated between acetoclasts (e.g. Geobacter sp., Shewanella sp.) and methanogens (e.g. Methanobacterium sp., Methanosaeta sp., Methanosarcina sp.) (Holmes et al., 2017; Lovley, 2011). In few words, microbes create a network of nanowires to exchange electrons among different species. Where conductive solid materials are present, this connection is favored, as compared to water-suspended cells (Chen et al., 2014).

Additionally, if the electron flow is forced from externally imposed electrochemical potentials, methane

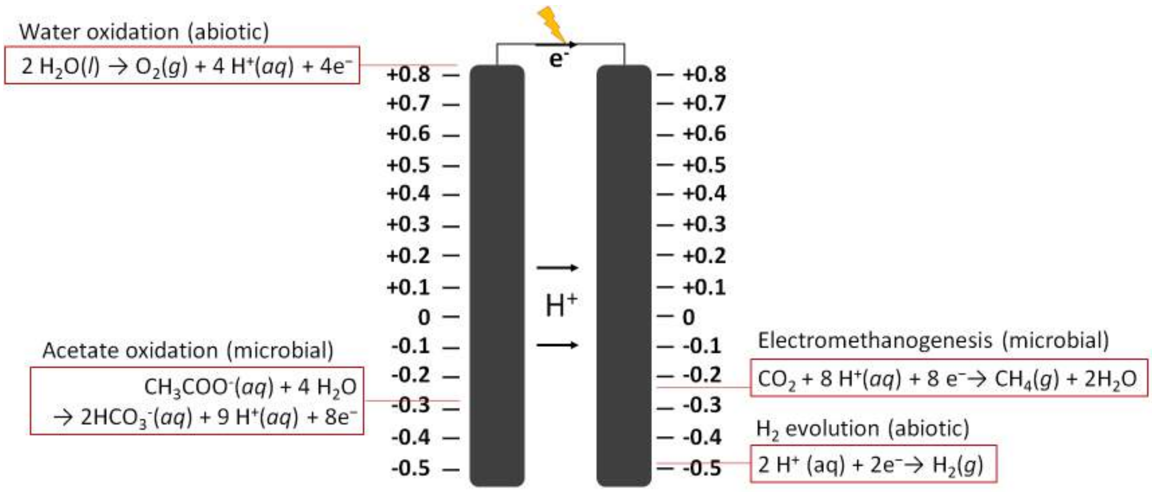


**FIGURE 1:** Electrochemical reaction standard potentials (E0, V vs standard hydrogen electrode, pH7) for anodic and cathodic main reactions in electromethanogenesis versus water electrolysis.

formation can be favored, even in absence of favorable electron donors, such as organic molecules (e.g. acetate) (Bajracharya et al., 2015). However, the thermodynamics of the system are favored when the electron donor at the anode is an organic molecule (Jadhav et al., 2017; Logan and Rabaey, 2012). In this case, the oxidation reaction is mediated by acetoclastic electro-active microbes, that discharge electrons to the conductive surface of the anode (Andrea Schievano et al., 2016; You et al., 2014). Acetoclastic methanogens were often found in anodic biofilms of bioelectrochemical systems (Rago et al., 2017; Rago et al., 2018) where they compete with electro-active microorganisms for the same electron donor (acetate). For this reason, several studies focused on different strategies for methanogenisis inibition in recent years (Chae et al., 2010, Rago et al. 2015).

## 4. ORGANIC WASTE AND BIOGENIC $CO_2$: THE NEW BATTERY

Important agro-industrial sectors by-produce huge amounts of wastewaters that in many cases are cause of environmental concerns and/or imply expensive and energy-consuming purification processes, using traditional technologies. Only in Italy, food-production sectors such as animal production, winery, olive oil and dairy by-produce nearly 200 million m$^3$ per year of polluted wastewaters (EUROSTAT, 2014). Nowadays, due to the high management costs related to proper disposal and treatment, this amount of polluted water is mainly spread to agricultural land as-it-is and/or, after inefficient purification processes, discharged to superficial/groundwater bodies. In many cases (ex. olive oil and winery wastewaters), the prolonged use of such high organic loads on agricultural soil implies decrease of soil fertility, eutrophication of water bodies and other environmental problems (Carey et al., 2016).

According to various reports, in the EU, as well as in the US, the electrical power consumption dedicated to wastewater treatment and water management is estimated in the range 2-5% of the total electricity consumption (McCarty et al., 2011). This corresponds to typical electrical consumption for aerobic activated-sludge process around 0.6 kWh/m$^3$ of treated wastewater.

If electromethanogenesis was implemented, in substitution to traditional aerobic treatments, at least half of this amount of energy (the fraction used for organic matter oxidation) could be completely converted into methane. Potentially, even higher amounts of reducing equivalents and $CO_2$ could be found in all solid organic waste to be treated in anaerobic digestion, composting and landfill plants. All these territorially-distributed hotspots of concentrated streams of biogenic $CO_2$, could represent a potential interface between the electricity network and the methane grid.

## 5. NEW CONFIGURATIONS OF ANAEROBIC DIGESTION

Two-stage anaerobic digestion process was suggested as an option to maximize the amount of energy recoverable from biodegradable organic waste in terms of hydrogen ($H_2$) and methane ($CH_4$) (Schievano et al., 2012; Venetsaneas et al., 2009). $H_2$ can be produced from organic materials in a process called dark fermentation (DF) (Manzini et al., 2015). Pre-treatments and a DF step (primary process) can be applied to enhance biomethane productivity of substrates, by increasing the overall biodegradability and equivalent balance. In DF, organic substances are hydrolysed and fermented by anaerobic bacteria to $H_2$, $CO_2$, and simple organic molecules such as volatile fatty acids (VFAs). Optimization of DF may lead to improved hydrolysis and therefore higher energetic exploitation of waste materials (Schievano et al., 2014).

The advantages of two-stage digestion systems, if compared to single-stage AD, are typically shorter substrate retention time, enhanced solids degradation efficiencies, enhanced hydrolysis with a subsequently higher $CH_4$ production and potentially higher organic loading rates (Pognani et al., 2015). In particular, the possibility of such biological process steps towards biogas upgrading (increasing $CH_4$ concentrations) to high-grade biomethane for grid injection, rather renovates the interest of this kind of plant configuration. As part of $CO_2$ is 'stripped' together with $H_2$ during DF, $CH_4$ concentrations in a biogas produced by a secondary methanogenic reactor shows tipically higher $CH_4$ contents, as compared to a single-step AD (Pognani et al., 2015; Schievano et al., 2012).

Therefore, DF should act as primary process to obtain a highly bioavailable hydrolizate (rich in VFA) for secondary bioprocessing, such as bioelectrochemical conversions (A Schievano et al., 2016). Among others, the most interesting possibility is represented by electromethanogenesis (Blasco-Gómez et al., 2017), where $CO_2$ is fixed to $CH_4$ by electro-stimulated microbial communities, towards further increase in biogas $CH_4$ content. In the applications of electromethanogenic processes, $CO_2$ contained in the biogas is used as electron acceptor: the electron flow is forced from externally imposed electrochemical potentials and methane formation from $CO_2$ can be favored, either in presence or in absence of favorable electron donors, such as organic molecules (e.g. acetate). This opens two possible scenarios:

a) With $O_2$-evolution from $H_2O$ as counter reaction, electromethanogenesis could be applied to a concept of bio-electrochemical power-to-gas, where peak electricity from the grid could be converted to methane by fixation of the excess $CO_2$ contained in the biogas. In this case, the total amount of bioavailable equivalents yielded by the dark fermentation step should undergo methanogenesis in the second stage of AD (Figure 2a), to maximize biogas ($CH_4$ + $CO_2$) production. This way, all $CO_2$ molecules are available to receive reducing equivalents from the electricity grid and be reduced to $CH_4$.
b) If bioavailable organic matter is sent to the anode, the thermodynamics of the system are favored, and the anodic oxidation reaction is mediated by acetoclastic electro-active microbes, that discharge electrons to the conductive surface of the anode (Schievano et al., 2016; Zhao et al., 2014). In this case, only a fraction of the total bioavailable equivalents yielded by the dark fermentation steps will be sent to the methanogenic AD stage

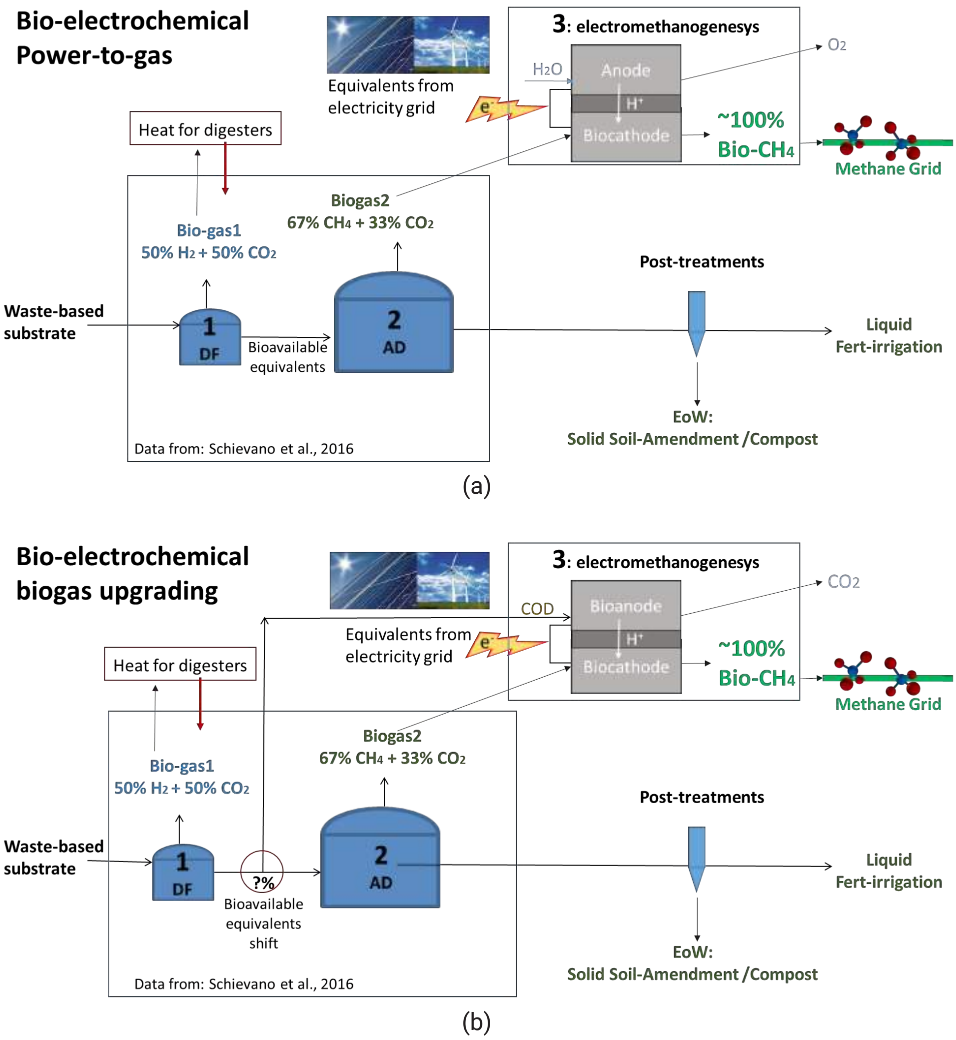


**FIGURE 2:** The concepts under investigation in this project: a) electromethanogenesis is used as bioelectrochemical power-to-gas system and all bioavailable equivalents yielded from dark fermentation are sent to AD methanogenesys; b) electromethanogenesis acts as bioelectrochemical biogas upgrading system and a fraction of bioavailable equivalents is sent to the bioanodic compartment, as counter-reactive.

**TABLE 1:** Electrode materials and characteristics of most performing lab-scale electromethanogenesis trials.

| | Cathodic poised potential (V vs SHE) | Volumetric surface area $m^2/m^3$ reactor * | Current density $A/m^2$ * | Volumetric methane generation rate $Nm^3/m^3$ reactor $day^{-1}$ | Current capture efficiency | Reference |
|---|---|---|---|---|---|---|
| Graphite fiber brush | -0.5 | 54 | 0.3 | 0.2 ÷ 0.3 | 96 | (Cheng et al., 2009b) |
| Graphite granules | -0.8 – -0.9 | 1350 | 0.07 | 0.1 ÷ 0.3 | 75 | (Batlle-Vilanova et al., 2015) |
| Carbon felt | -0.7 | 104 | 2.9 | 0.5 ÷ 1 | 80 | (van Eerten-Jansen et al., 2015) |

** total outer active surface area of the cathode, not including pores*

and the rest will serve as electron donors at the anode (Figure 2b). 'The right fraction of the total bioavailable equivalents might change significantly, depending on the type of biomass and the efficiency of both DF and AD. This particular point should be object of future research efforts. Under such conditions, the three-stage AD process can serve as a bio-electrochemical biogas upgrading system, because $CO_2$ can be fully converted to $CH_4$, with minimal external power supply. In such systems, the typical electrical power consumption is of 0.1 – 0.2 kWh/kg BOD removed, with cathodic conversion efficiency of power to methane in the range of 0.3 – 0.5 kWh/$Nm^3_{CH4}$ (Blasco-Gómez et al., 2017).

## 6. LARGE-SCALE BIOELECTRODES: ELECTRO-ACTIVE BIOCHAR?

Typical current densities of electromethanogenic systems are of the order of 0.1 – 1 $A/m^2$, i.e. 3 – 5 orders of magnitude lower than those of commercial abiotic water electrolysis systems. Finally, the counter reaction at anode (oxidation) can be also bio-electrochemical: as in microbial electrolysis cells, waste organic matter coming from waste and wastewater streams can be oxidized, with much higher thermodynamic gain, as compared to water oxidation to $O_2$. In such systems, the typical electrical power consumption is of 0.1 – 0.2 kWh/kg BOD removed, with cathodic conversion efficiency of power to methane in the range of 0.3 – 0.5 kWh/$Nm^3_{CH4}$ (Blasco-Gómez et al., 2017). In Table 1, we report the most successful materials used to design electrodes and their main characteristics.

The major challenge to scale up applications of this biotechnology consists in the fabrication of electrodes, with high geometric surface area per volume of bioreactor (of at least $10^2$ $m^2/m^3$). In fact, to counterbalance the low current densities, as compared to abiotic water electrolysis processes, an electrode should have enough surface area where the microbial reactions happen.

New low-cost and biocompatible conductive materials with such characteristics are currently under study (mostly based on graphite and char-coal derived from biomass pyrolysis) in many laboratories. Depending on different properties, such as quinone and aromatic structures, the degree of graphitization, high porosity and the presence of different superficial heteroatoms, biochar can be redox-active and electrically conductive (Kloss et al., 2012). It was recently demonstrated how biochar acts as soil-fertility promoter, not only indirectly by changing the soil structure and chemistry, but also by directly mediating electron transfer processes, i.e., by functioning as an electron shuttle (Kappler et al., 2014). In this experiment, effective biochar stimulated microbial reduction of the Fe(III) oxyhydroxide mineral ferrihydrite by Shewanella oneidensis (a well-known electro-active prokaryote). Meanwhile, Chen et al. demonstrated that biochar act as promoter of direct interspecies electron transfer for different syntrophic associations of microorganisms, thanks to its electrical conductivity and stimulating the capacity of extracellular electron transfer by pili or other direct membrane mediators (Chen et al., 2014). Swarnalakshmi at al. recently prepared a cyanobacterial-biofilmed (Anabaena) bio-fertilizer based on charcoal and soil, containing Azotobacter, Mesorhizobium, Serratia and Pseudomonas strains (Swarnalakshmi et al., 2013). Enhancements of up to 50% of nitrogenase activity were observed, as compared to control experiments.

A recent study, applied biochar to AD, demonstrating how the mere presence of 25 $g/L_{reactor}$ of biochar, derived from pyrolysis (500 – 800°C) of different ligno-cellulose biomass types, significantly improved AD process of food waste fermentate (Cruz Viggi et al., 2017). Other studies reported that several methanogenic communities are favored by the presence of biochar (Lü et al., 2016; Mumme et al., 2014).

Biochars are typically characterized by high BET specific surface areas, in the range of 10-100 $m^2/m^3$. Depending on the distribution of such area versus pore diameters, this surface might be available to microbes for extracellular electron transfer, and thereby work as a bioelectrode. Deeper studies are needed, to understand the potential use of biochar as low-cost substitute of highly technological materials such as carbon fibers or graphite, for the fabrication of large scale bioelectrodes, to be used in AD systems.

## 7. PERSPECTIVES AND CHALLENGES

In a near future, every single anaerobic digestion plant, landfilling site, wastewater treatment plant will be a potential spot for a highly efficient interface between the electricity and the methane grids (Figure 3); while treating waste organic matter "for free". This field of research is still in its infancy. However, the results obtained by several studies, even at a real-scale, promise near-future application of this concept. The main issues related to the process scale-up are given by the choice of the electrode materials and the

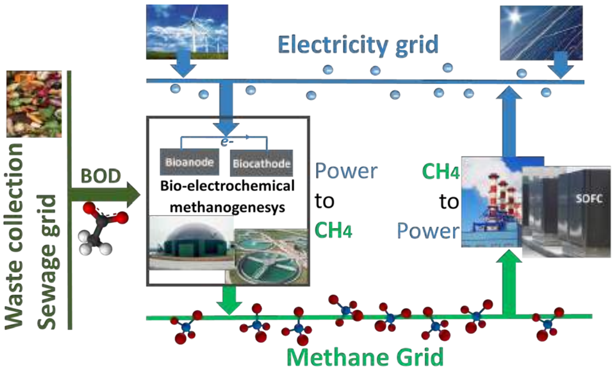


**FIGURE 3:** Scheme of the potential interface between the electricity and methane distribution grids, based on converting all organic waste/wastewater treatment facilities into Electromethanogenesis units.

reactor design. The use of electro-active biochar, as material for the development of large-scale high-surface area electrodes, is one of the most promising path of research and development for success in this field.

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# ENERGY PRODUCTION FROM RESIDUES: ECONOMIC ASSESSMENT OF BIOGAS INSTALLATIONS - DEVELOPMENT OF A CALCULATION TOOL

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**ABSTRACT**

The changing situation in Germany with the future limitation of funding by the EEG (Erneuerbare Energien Gesetz - Renewable Energies Act) (EEG 7/21/2014) has resulted in the situation whereby solutions for the operation of biogas plants lacking public funding need to be evaluated more precisely. The economics of biogas plants, powered exclusively by residues such as biowaste or municipal solid waste should be investigated with a focus on economic and business factors.

The economic assessment approach applied here examines business opportunities for regional biogas operations in different European countries. A calculation tool was developed that shows the profitability of biogas plants and considers main and typical characteristics to provide a reliable basis for a future business case analysis. The main purpose of this tool is to estimate the profitability of future biogas installations at the project planning stage; consequently, application of this tool should minimize the economic risks for investors prior to undertaking the main investment. As biogas plant lifetime is typically assumed to be 20 years, the model allows the creation of profitability models over the same period of time.

## 1. INTRODUCTION

Anaerobic digestion is a well-established method for energy production from biomasses. The biogas production technology has been implemented to varying degrees in accordance with the EU policies for green and sustainable energy supply (Ahrens et al. 2017a).

In 2013 9,200 biogas plants were in operation in Germany (GreenGasGrids, n.d.) and more than 12,000 biogas plants worldwide (incl. Germany) (ecoprog, 2016). Currently an installed capacity of approx. 7,000 $MW_{el}$ exists (ecoprog, 2016). According to the European Commission greenhouse gas emissions should be reduced by 20% (compared to 1990) and the share of renewable energies increased up to 20% of the total energy demand by the year 2020 (European Commission, n.d.). Therefore, additional biogas plants with an overall installed capacity of about 2,600 MWel will be built worldwide up until 2025. Although the German market has declined sharply Europe is still the undisputed leader in the field of biogas technology (ecoprog GmbH 2016).

In comparison with other European countries, biogas technology in Germany is well established. The question therefore arises whether the technology applied in Germany could be transferred to other countries or - if not - which changes and adjustments would be required. Apart from technical adaptions, the crucial question of economy and feasibility of the process under different European conditions is considered. Consequently, this study illustrates an appropriate method for use in forecasting the economic situation of a biogas plant installation in line with a series of critical economic parameters (such as tariffs or operating costs) and legislation (funding).

Although the field of anaerobic digestion has been widely investigated, the FNR (Fachagentur Nachwachsende Rohstoffe, Germany) has highlighted a need for further research following the development of technology, and particularly due to changes in market conditions (FNR, 2012).

The changing situation in Germany with the future limitation of funding by the EEG (EEG 7/21/2014) has resulted in the situation whereby solutions for the operation of biogas plants which are not reliant on public funding should be identified. The changes made to this legislation provide plant operators with less positive prospects than before. Therefore, operators of already existing biogas plants and especially new biogas plants will be forced to find new

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business opportunities.

While this new situation in Germany has been caused by modifications to the legislation, and therefore public funding, most European countries have no access to this kind of funding . The profitability of biogas plant operations devoid of public funding is therefore a crucial requirement to be taken into account on opting to implement this energy producing technology with a view to the future. A calculation tool that takes into account aspects relating to both current and future conditions has been developed and will be presented here.

## 2. USER-ORIENTED APPROACH

To date no studies have been undertaken to assess the economic feasibility and business opportuntites provided by biogas plants powered exclusively by residues (biowaste, municipal solid waste etc.) and crucially, not supported by public funding,. The economic assessment approach illustrated here intends to close this knowledge gap by examining (in addition to various input factors, such as availability of substrates, funding or operating costs) the regional opportunities for biogas business case studies in different European countries.

For the first time targeted assessments of the economy will be revealed, which provide significant decisional support not only for operators.

Consideration of the profitability of biogas plant operations is affected by numerous factors, including high costs or the generation of significant income.

First, substrates for the process need to be carefully considered and selected. The use of waste materials (organic residues from industrial or agricultural production processes, as well as for example, biowaste from households or other sources) represents the focus of the present paper. The use of these waste materials for the production of energy (particularly biogas) provides the possibility to avoid disposal and produces a positive effect on the profitability of planned business operations.

Present discussion on the widespread use of agricultural areas for maize cultivation (especially in Germany) underlines that the current situation needs to be changed, and alternatives such as the use of biogenic residues encouraged. In this context, the question arises whether it is economical to use waste-based materials as substrates for energy production in biogas plants.

An additional highly important factor impacting on the profitability of biogas plants is the feed-in tariff that can be obtained for the energy produced, which may be biogas itself or electricity and heat.

These aspects underpin the development of a comprehensive calculation tool for feasibility and profitability analysis. The economic aspects are based on a widespread data research of different operated biogas plants in Germany and Sweden (Abowe, 2014), as well as literature data and country-specific data from Lithuania and Estonia (Abowe, 2013). Moreover, all impacts on the economy such as legislation, country-specific aspects, technology and others, have influenced this study

All these investigations have led to the development of the calculation tool, capable of creating profitability models for the implementation of biogas technology over a period of 20 years.

Technology aspects, such as operation modes of biogas plants (wet/dry digestion, plug flow or garage fermenter) in particular, are likewise crucial in the consideration of profitability. Experiences and results from several lab tests, supervision of operating large-scale biogas plants as well as pilot plant tests (especially during the ABOWE project) were taken into consideration.

All the above-mentioned aspects were taken into account for the development of the calculation tool, which allows estimation of the profitability of a series of different biogas business cases.

The standard method for the execution of company evaluations is the discounted cash flow method. This method is based on calculating the value of future cash flows by discounting to the valuation date and deduction of the initial net investment (Schacht, Fackler 2009). The discounted cash flow method provided the basis for the developed profitability calculation tool.

The widely adaptable applicability for conditions to be found in different countries makes the tool and the background considerations an outstanding one.

The major objective underlying development of the calculation tool was to identify a means of demonstrating the profitability of biogas plants by means of a model taking into account the main typical characteristics, thus providing a reliable basis for a future business case analysis. The developed model has been successfully implemented and tested in several scientific applications in different European regions (Sweden, Lithuania, Estonia, Finland, Germany) – corresponding results have been published by ABOWE (2013, 2014), and Ahrens et al. (2017b), Ahrens et al. (2016).

The main aim of this study was to develop a calculation tool to enable the user to ascertain whether investment in a biogas plant would be profitable or not. The singular feature of this tool should be the consideration of operative cash flows over a lengthy period of time, ease of handling with readily available data and, a crucial factor, transferability to other European countries and – as a very far-sighted aim – application to other technologies.

A calculation model aimed at helping potential plant installers to determine profitability may represent a useful instrument, particularly when applicable to various types of input materials and design. In particular, application of the model in a European context constitutes the goal of this work.

A general model that takes into account all these potential issues provides an important tool, especially for less developed European countries in the field of biogas technology, to implement the EU Directive, and promote the establishment of biogas plants to improve the living conditions of the population.

All considered influence variables will be taken into consideration in the economic calculation. On the basis of this widespread approach a selection of the best and most profitable technologies is yielded.

Ostfalia University of Applied Sciences operates a series of pilot plants (plug flow fermenter for wet or dry fermentation as well as garage fermenter) for the production of biogas. In these pilot plants, test runs verify the results provided by theoretical calculations. Data determined by these tests form the basis for biogas potentials and therefore possible obtainable revenues.

On the basis of the results of the pilot plant tests the calculation tool was applied to large-scale projects and - with the aid of these tests - improved and advanced (Abowe 2013, 2014).

The EU-funded project ABOWE (Implementing Advanced Concepts for Biological Utilization of Waste) was fundamental in the preparation of this work and in determination of the necessary data. One objective of the ABOWE project was to promote investment decisions by performing pilot plant tests and related activities. The results were applied to evaluate cash flow calculations.

Figure 1 shows a simplified presentation of the necessary data and sequence to be adhered to in using the calculation tool. This flow chart represents the frame of the tool. After entering data relating to available substrates and selecting the operating mode and gas use, tariffs for the sale of the products should be chosen. Calculations and necessary data, which run in the background, are not shown here. Following this data entry, the cash flow over a time period of 20 years is obtained.

In performing the calculation a series of aspects (including necessary replacements of parts of the biogas plant, country specific aspects) were considered and different key values (e.g. investment costs, operating costs, energy demand, substrate data) were taken into account. Depending on the scenarios and the region all variables influencing economy or economic feasibility can easily be changed and adapted to ensure the calculation model provides an impression of the profitability of a planned biogas implementation project.

## 3. CASE STUDY - RESULTS OF THE ASSESSMENT

The following case study shows an example of assessment of biogas plant operations.

Two scenarios have been calculated: biogas plant projects with plug flow fermenter and CHP-unit (combined heat and power unit) for the production of electricity and heat, which only differed in the chosen tariffs for electricity.

In this easy example the costs for investment, maintenance and others - such as personnel or transport - were based on average data acquisition for biogas plants under different operating modes (mainly from literature studies and according to information from plant operators). These data were used as an operational background in the calculation tool and are not presented here. However, the tool provides the opportunity to vary crucial input parameters and allows the identification of different perspectives towards variation of resulting impacts on plant feasibility in line with variation of different cost parameters.

The calculated and verified economic key value data, used as a background in this tool, are based on the above-mentioned widespread data search.

Table 1 shows some of the assumptions made in the case of 2,004 tons of biowaste (methane potential 102 $Nm^3$ $CH_4$/Mg FM) (FNR 2017) and 680 tons of corn silage (methane potential 104 $Nm^3$ $CH_4$/Mg FM) (FNR 2017) being used as input materials for a biogas plant per year, yielding production of a theoretical amount of approx. 250,000 $Nm^3$ $CH_4$.

In these case studies, the produced electricity was assumed to be sold at a feed-in tariff of 0.247 €/$kWh_{el}$ (a possible electricity rate currently paid by customers in Germany (Stadtwerke Wolfenbüttel, 2017)) and that the feed-in tariff for heat is 0.03 €/$kWh_{th}$ (Ahrens 2017). Further assumptions related to an income from biowaste collection and utilization of 40 €/ton and expenses for corn of 30 €/ton. The results of the cash flow calculation are reported in Figure 2 (cumulative discounted cash flows, initial value = investment costs). The result of the calculation represents the possible profit after a defined period of time.

The second scenario proceeded based on the assumption that the feed-in tariff was much lower as the plant op-

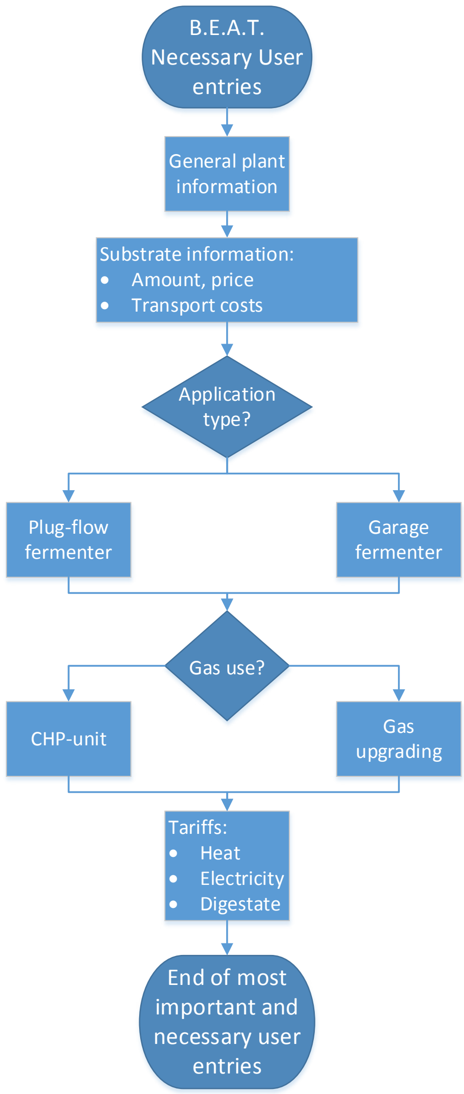


**FIGURE 1:** Flow chart of the calculation tool B.E.A.T. (Biogas installations Economic Assessment Tool).

**TABLE 1:** Tariffs and further background assumptions (extract of all used data).

| | Tariffs | | | |
|---|---|---|---|---|
| | Costs | | Income | |
| | Case 1 | Case 2 | Case 1 | Case 2 |
| Substrates: Biowaste Corn silage | 30 €/Mg FM | 30 €/Mg FM | 40 €/Mg FM | 40 €/Mg FM |
| Electricity | - | Grid charge and taxes: 0.169 €/$kWh_{el}$ | 0.247 €/$kWh_{el}$ | 0.078 €/$kWh_{el}$ |
| Heat | - | - | 0.03 €/$kWh_{th}$ | 0.03 €/$kWh_{th}$ |
| Produced energy | 984,518 kWhel/year and 984,518 kWhth/year; efficiency rate of the CHP-unit: η = 40% | | | |
| Biogas plant: Plug flow fermenter system, investment costs (average amount depending on size of biogas plant (per kW installed electrical capacity)) plus 30% for waste hygienisation (assumption) | | | | |
| Sale of electricity and heat (minus own demand); handling of digestate was not considered in this example | | | | |

erator was obliged to pay grid charges and taxes, implying a total lack of financial gain. In this case the feed-in tariff was taken as 0.078 €/$kWh_{el}$.

Figure 2 shows the possible result of an assessment made with regard to biogas plant operations using biowaste and corn silage.

Here the two case studies demonstrate the influence of the feed-in tariff for electricity. In case 2 the operator was obliged to pay grid fees and taxes, consequently the curve will not reach the zero line within 20 years.

## 4. RESULTS AND DISCUSSION

Wide-ranging experiences and investigations have resulted in the development of the calculation tool, which has enhanced the calculation of profitability estimates for a series of different biogas business cases.

The calculation provides the basis for a decision-making process relating to the financial implementation of advanced digestion technologies, particularly in the absence of public funding. Particular focus is placed on the potential demands of implementers and investors and the production of renewable energy throughout the EU member states. Data relating to influencing factors such as operating costs, market prices or substrate characteristics are continuously collected from existing biogas plants, and therefore the future results of the tool can be verified against real data. The particular aim in developing this tool was ease of handling with only a small amount of necessary input data.

The two case studies are intended to provide a rough idea of the purpose of the calculation tool and illustrate the strong dependency on important input factors such as feed-in tariffs. Therefore, the paper contains only an excerpt of all acquired data and assumptions required during calculations.

An approximation of the profitability of planned biogas projects is provided. Detailed calculations and estimates may only be obtained by applying models of real systems. The assumptions given are not intended for generalized use, but relate to scenario-specific aspects chosen for the examples presented here.

Applicability of this tool to new biogas concepts such as a demand-driven operations (Ahrens et al., 2016) and

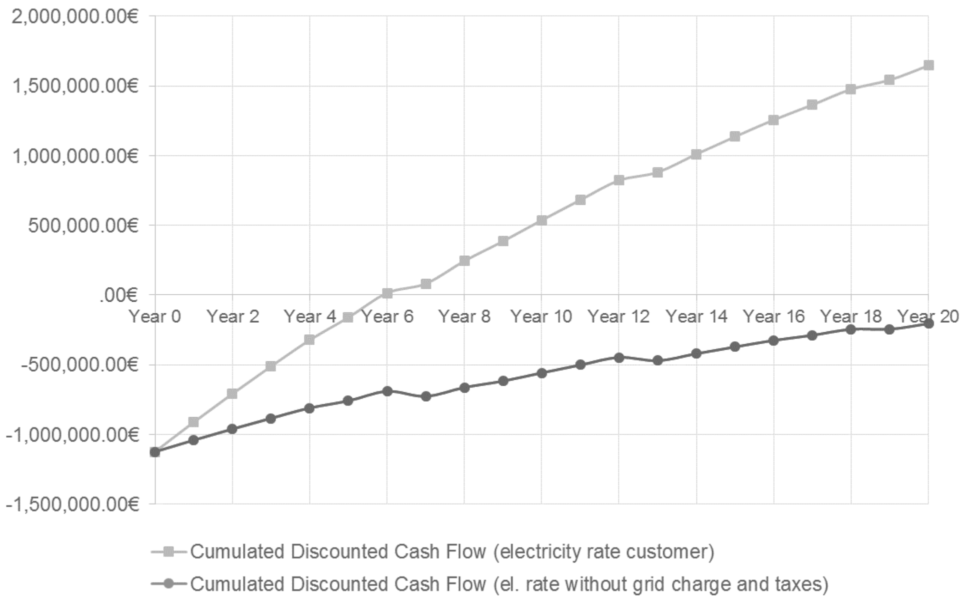


**FIGURE 2:** Cumulative discounted cash flows of two scenarios.

use of a series of CHP-units (combined heat and power plants) should results in a greatly enhance user-convenience, possibility of assessing existing large-scale plants, as well as the implementation of additional key data.

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# FEASIBILITY OF LANDIFLL GAS UPGRADE FOR USE AS A FUEL SOURCE FOR REFUSE TRUCKS: A CASE STUDY IN SOUTH AFRICA

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**ABSTRACT**

The overall aim of this study is to determine the feasibility of the upgrade of landfill gas extracted from a typical large MSW landfill in South Africa, to be used as a fuel source for the fleet of municipal refuse trucks. The outcome of this study aimed to determine the economic feasibility of the upgrade of landfill gas for use as fuel as well as the associated environmental benefits. Landfills contribute to 11% of the total (GHG) greenhouse gas emissions globally. The reduction of GHG emissions can be achieved through the extraction of landfill gas. At landfills in the major municipalities in South Africa, landfill gas is primarily extracted and flared or used for electricity generation. As a non-renewable highly priced commodity, the use of fossil fuels to haul refuse trucks proves to be an expensive endeavour for municipalities. Therefore, there is a need for alternative fuel sources.
The feasibility of the use of landfill gas as a fuel source was achieved by the analysis of the case study municipal facility and its process. The Waste Resource Optimization and Scenario Evaluation (WROSE) model, developed at the University of KwaZulu-Natal, was employed to determine the environmental, technical and economic benefits of implementing alternative waste treatment technologies. Municipal weighbridge and emissions data was collected from a number of landfills across South Africa and the WROSE model was used to analyse the environmental and financial benefits of three scenarios: 1. Landfilling of unsorted waste with LFG extraction and flaring, 2. Landfilling with LFG extraction for energy generation and 3. Landfilling of unsorted waste with LFG extraction and upgrading to fuel for municipal trucks. The outcome of the research shows that the landfill used as a case study presented great potential for the upgrade of landfill gas to a fuel source.

## 1. INTRODUCTION

Due to rapid urbanization and the increased generation of waste, large volumes of greenhouse gases (GHG's) associated with the waste sector are released into the atmosphere every year. In developing countries like South Africa, the primary method for municipal solid waste management is the collection and disposal of waste to landfills. Landfills are responsible for 11% of the total GHG emissions globally.

According to Bogner et al. (2008), the global waste sector contributes to 3% of all GHG emissions, and as much as 18% of global methane emissions. The waste sector in South Africa contributes approximately 4.3% of greenhouse gas emissions (Nahman, et al., 2012). However, its emission rate of 9 million tons of $CO_2$ equivalents is far greater than the African average of 5.8 tons and more than 6 times the sub-Saharan average of 1.4 tons (WRI, 2016).

The South African government has therefore expressed the need to reduce the amount of waste disposed of at landfill sites and has encouraged the use of internationally funded schemes, such as the Clean Development Mechanism (CDM) or similar, under the Kyoto Protocol, for the implementation of biogas-to-energy and waste-to-energy projects that would not have been possible without such incentives.

However, solid waste management in developing countries/emerging economies is generally characterized by highly inefficient waste collection practices, variable and inadequate levels of service due to limited resources, lack of environmental control systems and appropriate legislations, limited know-how, indiscriminate dumping, littering and scavenging and, most of all, poor environmental and waste awareness of the general public (Matete and Trois, 2008).

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South Africa, as an emerging economy, is also facing the challenge of meeting high standards in service delivery with limited resources. The disparity in service coverage between different communities in the same area is a characteristic of waste management practices in South Africa.

The Polokwane Declaration in 2001 has set as very ambitious targets the reduction of waste generation and disposal by 50% and 25%, respectively, by 2012 and the development of a plan for Zero Waste by 2022, forcing South Africa to invest in the valorization of waste as a resource.

The rationale for this research stems from several factors influencing the waste management sector in South Africa, including legislative developments, national imperatives and international obligations (Kyoto Protocol, Basel Convention etc.): the growing emphasis on GHG mitigation; landfill space shortages; waste diversion and zero waste goals increased focus on waste to energy technology implemented under the CDM and similar schemes, and the requirement for waste quantification and development of a national Waste Information System as mandated by the 2008 Waste Act.

Moreover, municipalities in South Africa, despite being among those geared towards innovation in waste management through effective implementation of the waste hierarchy, face many challenges, those in line with other developing countries which include: lack of capacity to improve service delivery and lack of investment in the use of waste as a by-product. These have hampered the development of the biogas and bioenergy sector in South Africa.

The province of KwaZulu-Natal (among the most populous in South Africa) houses 68 registered landfills, however only 3 landfills in the eThekwini Municipality (City of Durban) have registered CDM projects and are among the very few across the African continent. LFG is generally extracted and flared in most engineered landfills across the country.

In addition, the recent drop on the CDM market has shifted the uses of LFG therefore local authorities are now looking at cleaning the landfill gas for use in applications such as fuelling fleet for long hauling/collection trucks.

The University of KwaZulu-Natal, in collaboration with national and local government as well as the private sector has been involved in mapping the potential for bio-energy/waste-to-energy projects across the country since 2010 and developed the Waste Resource Optimization and Scenario Evaluation (WROSE) model to assist municipalities and industry in the decision-making process of implementing waste management strategies.

This article has the dual objective to explore the potential for the optimization of biogas as a fuel source using a typical South African case study municipality by assessing the viability of producing biomethane for co-generation or use in the automotive industry, in comparison to current biogas management practices used in municipalities such as LFG extraction and flaring, as well as LFG extraction for energy co-generation. The economic and environmental sustainability/feasibility of the three scenarios, for an averaged typical waste stream of a local municipality, have been evaluated using the WROSE model.

The model aids municipal officials in determining the best alternative waste treatment strategies relevant to their specific needs using waste volumes and composition. The outcome of the model is greenhouse gas emissions reduction potentials and economic assessments of each potential scenario (Trois & Jagath, 2011).

In addition to assisting municipalities in the optimisation of biogas management options, the study also looks at the use of waste as a resource, the overall reduction of greenhouse gases into the atmosphere.

Therefore, this study is intended to provide data and information to municipal waste managers with regard to different use of the landfill gas produced by the existing landfills, considering the carbon footprint and potential for GHG reduction as discriminants for their choice.

## 2. LANDFILL GAS MANAGEMENT IN SOUTH AFRICAN MUNICIPALITIES

According to the National Waste Information Baseline Report 2012, South Africa generates 108 Million tonnes (Mt) of waste per annum, of which 98 Mt (90%) is landfilled (DEA, 2012). Of this, 55% was 'general waste', 44% was 'unclassified waste' and 1% 'hazardous waste'. Furthermore, approximately only 10% of the total waste generated is recycled (DEA, 2012). The average South African produces 0,7 kg of waste a day, with 42 million cubes of general waste being produced per year (DEA, 2012).

GHG emissions data for South Africa also reflect these trends, with the waste sector contributing to 2% of the total emissions and waste management activities contributing to 12% of total methane emissions as seen in (DEAT, 2011). South Africa's primary mode of waste management is still landfilling (Greben and Oelofse, 2009) and available land near areas of large waste generation is becoming scarce, the GHG emissions and leachate from organic fraction of municipal solid waste (OFMSW) makes the use of landfill sites less attractive (Greben and Oelefse, 2009).

Moreover, within South Africa, 26%±2.6% of the total waste generated is organic waste, which, when digested and decomposes produces biogas. Biogas has the potential to be used as a fuel source to produce electricity or fuel. At present, South Africa is heavily dependent on the import of petroleum products. The rising costs of these products put strain on an already struggling economy. It is therefore necessary to find alternative fuel sources that are renewable and locally produced to reduce the country's reliance on imported products.

Zietsman et al (2008) postulate that the use of landfill gas as a vehicular fuel has the potential to make the process of landfilling more self-sustaining in addition reducing the dependence on fossil fuels and ultimately reducing GHG emissions. The liquefaction of natural gas is viewed as a safer and economical alternative for use in transportation technology. Liquefied natural gas (LNG) is an eco-friendly cryogenic fuel for sustainable development. Significant progress has been made in developing more energy efficient, low cost, small-scale natural gas liquefiers. However, the process of the conversion of biomethane to LNG is higher in electricity usage than that of compressed natural gas (CNG), therefore in this study the conversion technology considered will be for CNG. Heavy-duty vehicles are high volume fuel users and rely solely on fossil fuels. Technological advancements are being made globally to convert landfill gas to fuels in addition to developing natural gas vehicles.

There is an increase in efforts made towards zero waste management strategies and the use of waste as a resource. Through the use of processes such as landfill gas extraction, methane gas is converted into electricity or flared to reduce the GHG impacts on the atmosphere. However, South Africa, as a developing country, faces another important issue, which is the high cost of fossil fuels that, as a non-renewable resource, constitute a high demand commodity in the country according to the 2016 Energy Price Report. Due to the fluctuation in costs of fossil fuels the transportation costs at municipal solid waste facilities are high. This is due to the standard waste management practice being the collection of waste from households and the disposal to landfills. Therefore, it is important to assess the potential for alternative measures to assist municipalities in cost cutting measures. Various South African municipalities have invested in waste to energy technology, one such example is the extraction of landfill gas for the generation of electricity. However, in many cases the methane gas is extracted from the landfill facility and flared. It is therefore necessary to explore the alternative use of methane gas a fuel source for hauling refuse trucks. Not only does this address the issue of GHG emission reduction into the atmosphere but it also has the potential to reduce the cost of fuel for municipalities.

## 3. METHODOLOGY

The WROSE model was developed by UKZN to assist Local Authorities in the design of appropriate waste management strategies for the implementation of waste-to-energy projects, by providing a quantitative estimate of the potential for GHG reductions and landfill space savings that can be achieved through ad hoc zero waste strategies, assessing their economic feasibility and so addressing specific knowledge gaps regarding the quantity and quality of the local MSW stream. The WROSE model is a multi-criteria analysis model. The WROSE model is designed to evaluate five waste management scenarios:

- Scenario 1: Landfill disposal of unsorted, untreated MSW
- Scenario 2: Landfill disposal of unsorted, untreated MSW with landfill gas recovery
- Scenario 3: Mechanical pre-treatment of MSW, recovery of recyclable fraction through a Material Recovery Facility (MRF) with landfill gas recovery
- Scenario 4: MBT (MPT, recovery of recyclables through MRF and anaerobic digestion of biogenic food waste with landfill gas recovery).
- Scenario 5: MBT (MPT, recovery of recyclables through MRF and composting of biogenic food waste with landfill gas recovery).
- (Trois and Jagath, 2011)

At present, the model provides the user with information such as landfill space savings and GHG emission reduction figures as well as basic economic viability of alternative waste treatment technologies present in each scenario (Trois & Jagath, 2011).

The methodological approach adopted in developing WROSE was based on a dry-wet zero waste model that required the selection of zero waste strategies suitable for MSW management in South Africa and the development of possible waste management scenarios incorporating these strategies. Input to the model is a detailed qualitative and quantitative Waste Stream Analysis (WSA). The GHG impacts of every scenario were estimated using emissions factors that were developed into a GHG quantification model/tool (Friedrich and Trois, 2016).

The model can calculate also other indicators such as landfill space savings resulting from each scenario, as well as the potential income, capital and operating costs produced by each strategy.

The purpose of this research was to determine if the use of landfill gas for power generation or the upgrading of biomethane to be used as fuel source for the municipal trucks fleet in South African municipalities is a more competitive option than conventional biogas management strategies.

A large district municipality was selected as representative case study for this scenario analysis, and averaged weighbridge data across the province was used as input in the WROSE model.

The methodology used to achieve this included the following steps:

- Development and selection of waste management scenarios relevant and representative of typical South African Municipalities. Three scenarios were selected after an extensive review of waste management strategies in South Africa, as presented in Figure 1.
  - Scenario 1: Landfilling with gas extraction and flaring
  - Scenario 2: Landfilling with gas extraction and electricity generation
  - Scenario 3: Landfilling with gas extraction and upgrading
- Data collection of average waste data consisting of waste quantities and waste composition disposed of into the selected case study landfills (Figures 2 and 3).
- Using the information represented above from the case study municipality, an analysis was conducted to determine the economic indicators and environmental sustainability of each of the selected scenarios. The economic analysis takes into consideration the capex and opex costs to be incurred and the revenues generated by each scenario. While the environmental analysis is based on prediction of emissions of $CO_2$ generated or avoided in each scenario. Table 1 and 2 present the basic economic and environmental parameters used for the scenario analysis with a descriptive narrative of the rationale used in the definition and evaluation of each indicator.

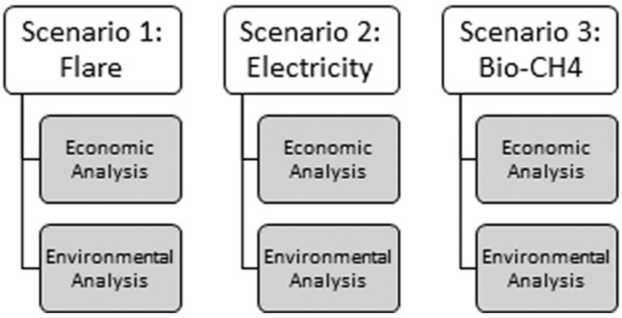


**FIGURE 1.** Selected scenarios representative of a typical South African municipality

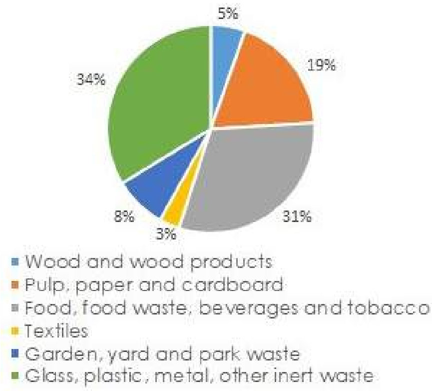


**FIGURE 2:** Average waste composition of the selected case study municipality.

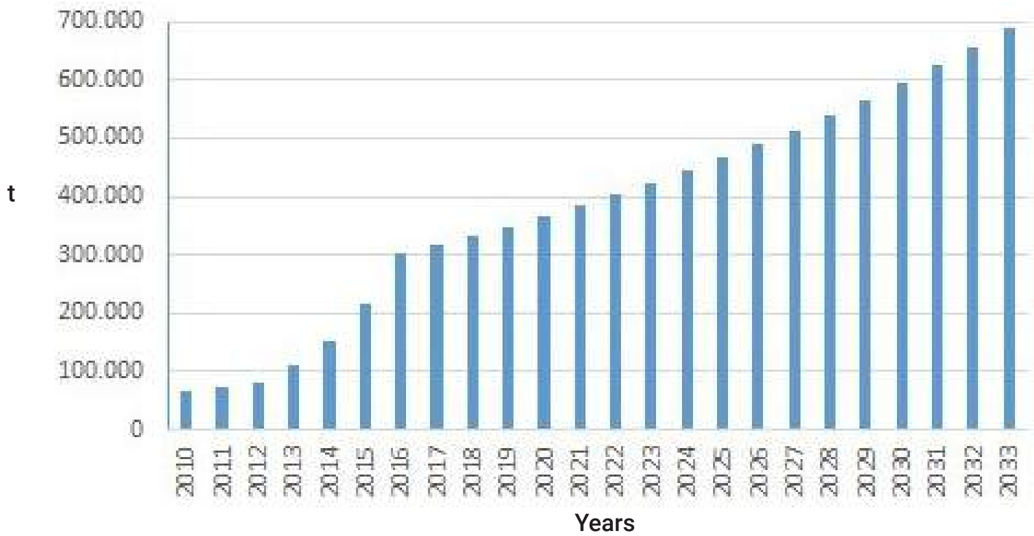


**FIGURE 3:** Waste quantities disposed of at the case study landfills to date with projected quantities over the next 15 years.

## 4. RESULTS AND DISCUSSION

### 4.1 Economic Analysis

Table 1 and Figure 4 clearly indicate that Scenario 3, with biomethanation for transportation, has similar costs as the previous 2 scenarios without LFG upgrading, however it also realises positive and higher savings from the use of biomethane as fuel for municipal trucks which offsets the costs for oil fuel, as compared with the scenario with production of biomethane for energy co-generation.

### 4.2 Environmental analysis

Table 2 and Figure 5 clearly indicate that all three scenarios realise similar $CO_2$ emissions reduction potential, with Scenario 2 (LFG upgrading for electricity generation) presenting a better performance than the other two scenarios assessed. The result of the scenario number 2 depends on the benefit of the $CO_2$ equivalent (the $CO_2$ not produced by a fossil fuel).

It is however to note that in neither of the scenarios the $CO_2$ emissions due to collection and transportation of MSW is considered.

## 5. CONCLUSIONS

This study assessed the feasibility and environmental impacts of applying the waste management strategy of upgrading LFG into a source of fuel to power a fleet of refuse trucks for a typical municipality in South Africa, in comparison to baseline strategies for biogas management. The principal environmental impacts were greenhouse gas emissions quantified by the WROSE model.

The research intended to provide South African municipalities with a methodology and a decision-making framework for the comparison of various waste management scenarios/strategies using the WROSE model. The three scenarios were found to be comparable in terms of environmental benefits but upgrading of biomethane as trucks fuel proved to realize higher revenues than the other two scenarios.

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**TABLE 1:** Economic indicators used for the cost-benefit scenario analysis.

| | | Scenario Flare<br>€ cent/kg waste | Scenario Electricity<br>€ cent/kg waste | Scenario Bio-$CH_4$<br>€ cent/kg waste |
|---|---|---|---|---|
| COSTS - CAPEX | | 0,020 | 0,079 | 0,154 |
| COSTS - OPEX | Waste collection | -5,830 | -5,830 | -5,830 |
| | Transportation | -5,830 | -5,830 | -5,830 |
| | Suction and flare | -0,006 | -0,006 | -0,006 |
| | Power generation plant | | -0,112 | |
| | Upgrading and dispensing | | | -0,082 |
| REVENUES or COSTS REDUCTION | CERs | * | * | * |
| | Selling of electricity | | 0,312 | |
| | Use of biomethane to power the trucks to collect and transport the waste. The costs referred to fuel oil consumption for waste collection and transportation | | | 11,660 |
| | Use of the biogas generated for the generation of electrical energy and heat necessary | | | 0.022 |
| **RESULTS (excl. CAPEX)** | | **-11,666** | **-11,466** | **-0,066** |

**TABLE 2:** Environmental indicators ($CO_2$ emissions) used for the scenario analysis.

| | | Scenario Flare<br>$t_{CO2}/t_{waste}$ | Scenario Electricity<br>$t_{CO2}/t_{waste}$ | Scenario $CH_4$<br>$t_{CO2}/t_{waste}$ |
|---|---|---|---|---|
| EMISSION OF $CO_2$ | Waste collection with trucks | 0,011300 | 0,011300 | 0,011300 |
| | Transportation to landfill with trucks | 0,011300 | 0,011300 | 0,011300 |
| | Emission of CO2 for the upgrading | | | 0,012 |
| REDUCTION OF $CO_2$ | t $CO_2$ avoided | -0,2066 | -0,2066 | -0,2066 |
| | t $CO_2$ equivalent | | -0,0394 | |
| | Use of biomethane to power the trucks working with waste collection as an alternative to diesel. | | | -0,023 |
| | Reduction for the use of Bio $CH_4$ istead of diesel. | | | -0,012 |
| **TOTAL** | | **-0,184** | **-0,223** | **-0,207** |

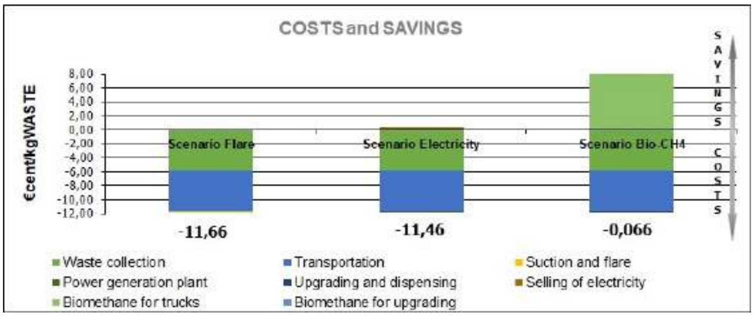


**FIGURE 4:** Economic analysis of the three scenarios.

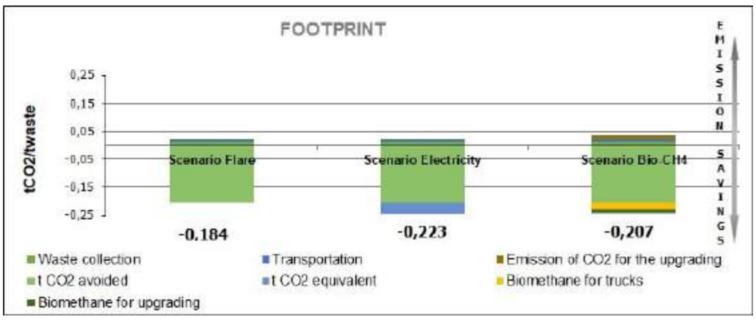


**FIGURE 5:** Environmental analysis of the three scenarios.

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# SIGNIFICANCE OF IMPLEMENTING DECENTRALIZED BIOGAS SOLUTIONS IN INDIA: A VIABLE PATHWAY FOR BIOBASED ECONOMY

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**ABSTRACT**

In recent years, world economy has achieved considerable economic and social development, but this has resulted in the widespread degradation and depletion of our natural environment. Solid-waste generation increases exponentially due to the rapid urbanization, but inappropriate waste handling causes health hazards and urban environment degradation. The goal of strengthening bio-based economy is potentially related to biogas solutions with respect to solid waste management in several ways. Bio based economy demands new ways of philosophy and co-operation within and across sectors to minimise the environmental footprint and climate change throughout the value chain. The range and significance of biogas technologies has increased rapidly during the last 30 years for the treatment of organic waste. The typical way of perceiving the role of biogas solutions is as a last step in cascading the biomass where renewable energy in the form of biogas is produced along with bio-fertiliser. Among the existing technologies for waste treatment, anaerobic digestion (AD) plays a key role in reducing waste along with the generation of renewable energy as it plays a key role in reducing the adverse effects on environmental and climatic changes of the biosphere such as the reduction of NOx and $CO_2$ emissions in the atmosphere, leading to reduced carbon foot print and reduction in solid waste accumulation. However, AD of organic waste with conventional digesters is not practicable for the processing of large quantities of waste generated in India. The aim of the present work is to present knowledge and technological gaps between slow rate and high rate biogas solutions that emerged in India, used to replace conventional fuels with renewable and green fuel (biogas) through initiatives taken up by the governmental and private organizations.

## 1. INTRODUCTION

Energy demand in various sectors is continuously increasing due to enourmous growth in population and industrial development. Natural energy resources like coal, oil and gas are fulfilling the energy demand globally, however, due to complete dependence on natural reserves and continuous exploitation, it is leading to depletion of resources besides deteriorating the natural environment. Hence, there is the need to explore alternative forms of energy which are renewable and, at the same time, subside the usage of conventional energy forms. One of the solutions to such problem is the evolution in alternative renewable and sustainable energy technologies like biogas and bioenergy (Cantrell et al., 2008, Begum et al., 2017, Juntupally et al., 2017). Currently, 10% of the world's energy demand is met by bioenergy (Thomas et al., 2017). Sustainable and economically viable way of producing biogas and bioenergy is to purposefully utilize the wastes as resources that are enormously increasing due to urbanization and rapid industrialization. For instance, about 62 Million Metric Tons (MMT) of waste is generated in India per year of which about 50-55% is biodegradable in composition (Begum et al., 2016). Current global municipal solid waste (MSW) generation levels are approximately 1.3 billion tonnes per year, and are expected to increase to approximately 2.2 billion tonnes per year by 2025 (Daniel and Perinaz, 2012). The production of biogas from biode-

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gradable waste through anaerobic digestion (AD) process presents significant advantages over other forms of renewable energies. AD is an established technology that is proven to be most energy efficient and eco friendly, but it is under used in majority of the developing countries. A few biogas installations for the treatment of animal wastes of smaller capacitites (50 to 200 kg/day) operated in the past two decades ago but now it is highly important to set up decentralized biogas plants to treat the waste at source to obtain benefits in terms of methane rich biogas for decentralized cooking applications and organic fertilizer with notable N, P and K content to replace chemical fertilizers. This kind of approach not only results in effective waste management and renewable energy generation but also avoids transportation and disposal costs involved in handling wastes. The waste management plan lays out quantitative targets for recycling, as well as many initiatives deal with encouraging the development of green technologies, improve waste policies and an increase focus on citizens and their role in the waste management cycle. Energy recovery from organic waste can offset operational costs, generate revenue, increase the share of renewables in the energy mix and reduce GHG emissions. A salient reason for advocating biogas technology exploiting waste as a resource is to effectively manage waste, reduce the burden on natural energy resources through environmental friendly process and drive the nation towards improving biobased economy. The idea of creating a bio-based economy where waste derived renewable resources are used for industrial purposes and energy production, without compromising on food and feed provision, has already been set out by the Organization for economic co-operation and development (OECD) in 2009. Around the world, countries are promoting the circular and biobased economy agenda and the number of solutions provided by companies is increasing rapidly. To implement these solutions several barriers need to be overcome to shift the system towards one aligned with biobased economy principles. These include regulatory barriers – such as inconsistent and biased definitions of waste, and economic hurdles, including the absence of accurate externality pricing – which tilt the field towards traditional systems, rather than designing it to obtain biologically derived materials and energy. Overcoming such barriers will further enable the technological advances required to realise the economic opportunities. It is estimated that the total number of biogas plants in India has increased from 1.23 million in 1990 to approximately 4.54 million in 2012, despite an estimated potential of 12.34 million digesters (Lohan et al., 2015). Apart from the direct benefits gleaned from biogas systems, there are other, perhaps less tangible benefits associated with this renewable technology. By providing an alternative source of fuel, biogas can replace the traditional biomass based fuels, notably wood. Benefits can also be scaled up, when the potential environmental impacts are also taken into consideration; significant reductions in emissions associated with the combustion of biofuels, such as sulphur dioxide ($SO_2$), nitrogen dioxide ($NO_2$), carbon monoxide (CO), total suspended particles (TSPs), and poly-aromatic hydrocarbons (PAHs), are possible with the large-scale introduction of biogas technology unlike small scale biogas plants. However, AD of biodegradable waste helps to reduce massive amounts of $CO_2$ eq. emissions, produce valuable renewable energy and recover nutrients. It is estimated that per every ton of organic waste disposed off in an unsuitable manner about 1.2 ton of $CO_2$ is released in to the atmosphere leading to global warming (Mahapatra et al., 2009; Pathak et al., 2009; Hashmi et al., 2007). The aim of the present work is to present the decentralized biogas solutions that emerged in India to replace conventional fuels with renewable and green fuel (biogas) through initiatives taken up by the governmental and private organizations. It also highlights the proactive factors in spreading the biogas technology, tackling the waste management issues and barriers that hindered the process of adopting biogas technologies, e.g. lack of awareness, financing etc. A comparison between the conventional and high rate digesters in terms of its design, process, product yield and its economic feasibility at small and big scale is also presented. This paper also describes a case study emphasizing practical experiences witnessed in installing, commissioning and working of a decentralized biogas plant for the treatment of organic waste, together with its output in terms of biogas, digestate, techno economic and environmental assessments combined with the discussion on technology transfer and commercialization. An understanding on the significance of setting up decentralized biogas plant that could strengthen the biobased economy is also presented.

## 2. METHODOLOGY

The initial part of the study focused on investigating the type of biogas solutions that evolved in India, its installations in rural areas promoted by government projects and on barriers that hindered the spread of biogas technology due to limited access to information and lack of awareness on the scientific innovations. The study was based on the documents that were available in the internet, formal discussions with the experts in biogas technology who developed the biodigesters suitable for Indian environment based on design principle and rawmaterials and a few spontaneous responses of participants on the biogas concept in the bioenergy conferences. A second part of the study focused on presenting a case study describing the installation and commissioning of the biodigester at site (Bellary, Karnataka, India), operation and maintenance, retrofitting the digester during unstable conditions by the technology contractor, an estimate on its products utility and pay back period by the beneficiary of the project.

## 3. RESULTS AND DISCUSSION

### 3.1 Necessity for promoting biogas technology in India

About 70% of the population in India lives in rural areas. The rural energy scenario is characterized by inadequate, poor and unreliable supply of energy services and large dependance on traditional biomass fuels (Begum et al., 2017). Every year several large and small scale farmers in India burn agricultural crop residues after harvesting, to reduce the costs involved in storage and handling that resulted in release of atmospheric pollutants such as aerosols, suspended particulates, $SO_x$ and $NO_x$. In addition, burning crop residues leads to decrease in soil nutrients and fertility. As India is considered an agriculture-based country, huge quantitites of animal waste is generated. However, it is possible to convert this waste into a valuable resource by exploiting modern waste to energy conversion technologies. Considering the technological innovations and the availability of waste like biomass, animal manure etc., Indian government started to install biogas (gobar gas) plants exploting dung as substrate for the production of biogas as a cooking fuel. Hence, necessity for promoting biogas technologies in India arose and accordingly the Indian government under different renewable energy schemes has taken initiatives to promote further use of biogas solutions in rural India. The detailed information on the biogas solutions emerged is discussed in subsequent sections.

### 3.2 Biogas solutions that emerged in India

To tackle the waste disposal issue, various biogas solutions emerged in India. The biogas solutions that emerged in early 1960s in India are based on two types of digesters i.e. fixed masonry dome type and floating drum type. A cattle dung-based floating drum type biogas unit named Khadi and Village Industries Commision (KVIC) was the first solution evolved to treat cattle dung in rural India. This technology was approved by the Ministry of New and Renewable Energy (MNRE) and promoted under National Programme on Biogas Development (NPBD) (MNRE, 2010). Though it was a popular technology, high plant cost and the need to regularly apply paints on metal biogas holder to protect from corrosion were the drawbacks of the technology. Other biogas solutions, including Deenabandu and Pragati models, overcame the drawbacks of the aforesaid technology. The majority of researchers reported that the aforesaid biogas models are slow rate conventional digesters, suitable to treat only animal wastes and characterized by low organic degradation efficiency and biogas yield, while the associated disadvantages include choking and scum formation in the reactor. Hence, the main necessity was to develop high rate anaerobic digester model that could treat any kind of segregated organic waste for the generation of methane rich biogas and nutrient rich organic fertilizer with a high degradation efficiency and product yield. Hence, the different research organizations in India promoted an initiative aimed at the development of high rate anerobic digesters that could meet the standards set by the advanced digesters in European countries (EBA, 2016).

### 3.3 Principle and working of a biogas plant

The process exploited by a biogas plant is AD or biomethanation, which is defined as a sequential process including hydrolysis, acidogenesis, acetogenesis and methanogenesis. A typical biogas plant has a digester in which the slurry (substrate mixed with water) is anaerobically digested for the generation of methane rich biogas and digestate. Intially the slurry from feed preparation tank is pumped in to the digester and subject to microbial action. The microbes consume the available organic matter from waste and produce methane, carbon dioxide, nitrogen and traces of hydrogen sulphide which is collected in a gas holder. The digestate from the digester is collected in outlet tanks, for further use as organic fertilizer if the quality meets standard criteria (Arelli et al., 2018).

### 3.4 Slow rate V/s High rate digesters

It is a known fact that anaerobic digesters currently employed for solid waste treatment are broadly classified into two distinct categories, such as conventional (slow rate) and high rate digesters. Slow rate digesters are based on two basic designs, the floating dome type and the fixed dome type digesters, and are designed for very long hydraulic retention time (HRT) with large reactor volume and low volatile solids (VS) reduction. These digesters were suitable for the treatment of cattle manure with a low treatment capacity i.e 50 to 200 kg/day (Tiwari et al., 1988; Sonam et al., 2014). It was reported that majority of the plant installations in rural India were based on conventional slow rate digesters, which have been found to be failed due to chocking and scum formation due to lack of proper maintanence (Begum et al., 2017). Hence, for the biomethanation of organic solid waste in any case, conventional digesters are not suitable for the processing of large quantities of waste. Therefore, High rate digesters are basically designed to minimize HRT and increase the rate of biogas production by AD features optimization, such as appropriate mixing, ensuring high density of active biomass, providing good buffering capacity, controlling food to microorganism ratio and suitable slurry concentration in the digester, monitoring possible microbial culture inhibition mechanisms etc. High rate digesters also address the bottlenecks involved in conventional digesters. The high rate digesters design allows to treat waste from 250 kg/day to 10 ton/day. The innovative features that make the high rate digesters more suitable from conventional digesters are shown in Table 1.

### 3.5 Economic feasibility/viability

The applicability of any green technology is difficult to quantify unless its economic feasibility is studied. The costs associated with biogas technology can be categorized under the following heads:

1) Initial costs of construction and installation: cost of labor, excavation cost, costs of construction materials, pipes and their set up for biogas supply, transportation cost of materials.
2) Operative Costs: costs associated with mixing of feedstock in the slurry tank with water, pH boosters to operate the reactor in stable conditions, collecting the digested slurry from outlet tank, drying the digested slurry to obtain solid fertilizer (Braimakis et al., 2014). The economic feasibility along with remunerative benefits is discussed in detail in subsequent section.

### 3.6 Strategies for implementing decentralized small and medium scale biogas solutions

India has implemented a large biomass energy program, which involves the promotion of several bioenergy technology programmes (BETP) through several policies,

**TABLE 1:** Comparitive analysis of slow rate and high rate digesters.

| Parameter | Slow rate digeters | High rate digesters |
|---|---|---|
| Type of substrate | Animal wastes like cow dung, cattle dung | Any kind of organic wastes like food waste, animal wastes, agricultural waste and organic fraction of MSW |
| Capacity of treatment (kg/day) | 50 – 200 | 250 – 10,000 |
| Material of construction (MOC) of digester | Concrete | Mild steel or stainless stell |
| MOC of biogas holder | Metal, plastic | Poly vinyl chloride balloons |
| HRT (days) | 40 - 60 | 15 - 24 |
| Volatile solids loading rate (kg VS/$m^3$ of digester/day) | 1 -2 | 5 - 6 |
| Total solids loading rate (kg TS/$m^3$ of digester/day) | 3 - 4 | 6 - 8 |
| Volatile solids reduction (%) | 40 -50 | 70 - 80 |
| Biogas generation ($m^3$/ton.day) | 30 - 40 | Cow dung: 40 – 50 |
| | | Poultry litter: 70 – 75 |
| | | Food waste: 120-140 |
| Methane content (%) | 45 - 50 | 60 – 65 |
| Foot print area required ($m^2$) | - | 55 (for one ton of waste treatment plant) |
| Payback period for one-ton plant | - | 2.5 - 3 years |

institutional and financial incentives and interventions. The importance of high rate digesters, developed by various research institutes/organizations to address the limitations of the slow rate digesters, increased in the past decade. Government has undertaken strategical approach by implementing a sequential process, in which Technology transfer, demonstration and dissemination leading to commercialization have been the corner stones of Ministry of New and Renewable Energy (MNRE) for the deployment of renewable energy technologies in India. Most of the BETP were implemented with direct capital subsidy support from the MNRE (MNRE, 2017). Other policy incentives, (e.g. income tax reduction, accelerated depreciation, concessional duty/custom duty-free import, soft loans for manufacture and state level policies on wheeling and banking, etc.), were also used to facilitate market development. Despite several supportive policies and incentives, the rate of spread of bioenergy technologies has remained low. About 3.83 million household biogas plants were installed till 2006 against a potential of 12 – 17 million. The total number of large community and institutional biogas plants installed until 2006 was 5,500 for the treatment of organic solid waste such as food waste (Ranjit et al., 2013; Viswanathan and Kavi, 2005). Implementing the aforesaid approach in diseminating biogas solutions has the aim oftracing the path towards the improvement of the national biobased economy. The slow rate of spread has been attributed to the existence of several barriers, which have been identified in several studies. To improve the spread of BETP it is essential to address the most important of the barriers and develop appropriate measures to overcome them.

## 3.7 Barriers: Factors hindering spread of biogas plants

### *3.7.1 Limited capacity to assess, adopt, adapt and absorb scientific options*

Implementation of biogas solutions was primarily targeted at rural areas, who have limited capacity and knowledge to absorb these technologies. Due to the intervention of governmental subsidy schemes, the rural community gradually adopted the biogas technologies for the waste treatment and consequent biogas generation. Nevertheless, due to the increasing waste quantities and its associated disposal issues, biogas technologies are pursued by the urban bodies as well (Tiwari et al., 1988). A general resistance to change is noted, which is magnified due to lack of capacity to understand, adopt and adapt the technologies for improved benefits. The limited capacity towards the technological shift is not only linked to its use but also to its production and maintenance (Ravindranath and Ramakrishna, 1997). The limitations in knowledge and manufacturing capacity resulted in no significant innovation. However, nowadays, it is possible to record modern high rate plants, characterized by affordable costs and benefits.

### *3.7.2 Insufficient information and financing possibilities to assess the technological needs*

The requirements correlated to technology differ from one stakeholder to another. For example, a user of a cooking stove might have different information needs as compared to a manufacturer of stoves. The generic approach of disseminating information on biogas solutions in rural and urban areas had only limited impact and hence the access to information also remains a key issue. High initial capital costs and investments associated with mass manufacturing of reactors and its accessories are additional barrier in promoting biogas solutions. Both users and manufactures have a very low initial capital; this fact is further accentuated by the rigid lending procedures that limit access to financing even when financing is available on standard norms. Large part of farmers are unaware of the possible ways in which farm and cattle wastes could be efficiently utilised. In this context, government agencies and NGOs are major stakeholders in creating awareness. Moreover, many farmers find difficult to bear the construction and operational costs of setting up the digester (Ravindranath and Ramakrishna, 1997). This again requires the government to introduce incentives (like soft loans) and subsidies to enhance the approachability of the technology and thus increase its market diffusion.

### 3.8 Sustainability of decentralized biogas solutions

The efficiency of solid waste management could be enhanced by the installation of decentralized small and full scale biomethanation systems (Sonam et al., 2014). Decentralization offers several advantages, such as allowing immediate implementation of decision taken on site, and exploiting a higher degree of organic content of the OFMSW, available from the avoidance of possible degradation during transportation (MSW rules, 2000). Operational cost occurring in decentralized biomethanation systems is low compared to centralized biomethanation systems, since decentralization completely avoids the transportation costs and accidental hazards (Braimakis et al., 2014). Transportation cost linked to centralized approach would be reduced if the MSW collection system was designed focusing on local and decentralized management. The avoided cost will allow the municipalities to focus more on disseminating available green technologies for a safe waste treatment and disposal, thus contributing to achieve sustainable development. On the other end, employees can be empowered by having more autonomy to make their own decisions, giving them a sense of importance and making them feel as if they have more input in conducting the organization. Also, It allows them to make better use of the knowledge and experience they have gained and implement some of their own and creative ideas. The common people in rural areas who are ignorant about all these treatment methods and technologies could acknowledge the improvements occurred in their respective areas and support the government in undertaking solid waste management projects by implementing sustainable and decentralized biogas solutions.

## 4. CASE STUDY

Several institutions, research organizations and companies took up an initiative to develop high rate biodigesters for the treatment of organic solid wastes. Indian companies offering different biogas technologies, characterized by various working principles and design aspects (e.g. capacity, type of waste, output etc) is shown in Table 2. Subsequently, a case study has been presented to elaborate in detail the benefits (technical and economic benefits in terms of biogas and digestate production and utilisation) associated with decentralized high rate biogas plants for the treatment of waste at source of production.

### 4.1 High rate biometanation of organic solid waste based on Anaerobic Gas lift Reactor (AGR) Technology

CSIR-Indian Institute of Chemical Technology (IICT), Hyderabad is a research organization extensively working on transnational multidisciplinary research to resolve the societal issues. Considering solid waste treatment and disposal as a primary issue that should be be resolved, CSIR-IICT has made intensive research efforts to develop a novel high rate biomethanation technology called "ANAEROBIC GAS LIFT REACTOR (AGR)" (PT-609/0207NF2012) for the generation of biogas and digestate from organic solid waste (CSIR-IICT, 2014; Gangagni Rao and Swamy, 2012). This technology is superior in terms of biogas and digestate production as it incorporates novel pre and post processing processes. A pilot plant based on this technology has been installed at Toofran near Hyderabad and operated with various sustrates like poultry litter, cattle manure, napier grass, cooked and uncooked food waste to understand the performance of the digester in terms of biogas and digestate production, volatile organics degradation rate, hydraulic residence time etc. Subsequently, the technology has been licensed to M/s Ahuja Engineering Services Private Limited (AESPL), Secunderabad, (AESPL, 2017) that executes projects on turnkey basis. The Akshaya Patra Foundation is a non governmental organization (NGO) operating under the aegis of ISKON, and serves mid-day

**TABLE 2:** Comparative analysis on biogas solutions offered by different companies (Begum et al., 2017; GPS Renewables, 2017).

| Company Name | Capacity (kg/day) | Type of kitchen waste | Working principle and design | Power consumption to operate the plant (kWh/day) | Products | Biogas production ($m^3$/day) |
|---|---|---|---|---|---|---|
| M/s AESPL | 1000 and above | All type of organic waste (cooked and uncooked food waste, poultry litter, cattle manure, organic fraction of municipal solid waste) | All type of organic waste (cooked and uncooked food waste, poultry litter, cattle manure, organic fraction of municipal solid waste) | 12 - 15 | Biogas and Digestate | 120 - 140 |
| M/s GPS | 200 - 2000 | Cooked food waste | Cooked food waste | 20 - 25 | Biogas | 90 - 100 |
| NEERU | 100 - 1000 | Animal manure, kitchen waste | Animal manure, kitchen waste | 10 - 15 | Biogas and Digestate | 50 - 60 |
| Mailhem Ikos | 100 - 1000 | All types of organic waste including animal manure, organic MSW | All types of organic waste including animal manure, organic MSW | 20 - 25 | Biogas | 50 - 60 |
| Xeon Waste Managers LLP | 250 - 2000 | Cooked food waste | Cooked food waste | 20 - 25 | Biogas | 70 - 80 |

meals to children in schools (TAPF, 2018). Several large-scale kitchens are set by TAPF across the country generating 1 Ton of organic waste per day. M/s AESPL, under the technical guidance of CSIR-IICT, has installed a plant at one of the kitchens of TAPF, in Bellary in Karnataka in 2015, for the generation of biogas and digestate from food waste (cooked and uncooked) produced by the kitchen, exploiting AGR Technology. Approximately 1000 kg of food waste (400 kg cooked food waste, 600 kg uncooked food waste) and 500 L of organic wastewater (rice water/ganji) is used for the generation of 120 to 140 $m^3$ of biogas per day to replace 50 to 60 kg of LPG (Kuruti et al., 2017).

**Equipment required for the installation of biogas plant:**

- *Major equipment*: Anaerobic digester with accessories.
- *Pre and post-treatment accessories:* Waste crusher/shredder, conveyor arrangement for loading waste into crusher, process pumps, biogas scrubber, biogas gasomether, biogas compressor, biogas pressure tank, biogas flare unit/gas blower.

The aim of this plant is to serve as a sustainable technology to provide a suitable waste disposal system to the kitchen as well as utilize the clean fuel (biogas) produced as a cooking fuel to replace LPG consumption and promote decentralized solid waste management through biogas installations. The plant runs like a closed-loop system where the kitchen's waste is converted into biogas and fertilizer on day-to-day basis. Based on the success at Bellary and satisfaction of beneficiary in terms of product yield, payback period, operation and maintenance and other intangible benefits achieved, TAPF has given consent to install and commission the biogas plants at its kitchen at Hubli in Karnataka, Ahmedabad, Surat and Bhavnagar in Gujarat. The techno-economical and the economic benefits created by the biogas plant is shown in Table 2. The full-scale biogas plant described above is shown in Figure 1.

## 5. FUTURE PERSPECTIVE ON DECENTRALIZED BIOGAS SOLUTIONS

Sustainability of a decentralised waste management and biomethanation system requires satisfaction of a minimum of three sustainability bottom-line factors, namely: economic, environmental and social sustainability. High rate biomethanation plants can be introduced within the community (colony, housing society, institution premises, local area etc.) through Resident Welfare Association (RWA), Community Based Organization (CBO), Non-Governmental Organization (NGO), Advanced Locality Management (ALM), Self Help Groups and so forth. However, certain factors limit its widespread application in rural societies in India. The federal and stage governments need to be more proactive in providing easy access to these technologies to the poor farmers. The policies and support of the government are decisive in persuading the farmers to adopt such technologies and to make a transition from wasteful traditional approaches to efficient resource utilization. The concept of creating biobased circular economy through biogas solutions unites the aspirations of delivering economic growth, job creation and protecting the global environment, which is under pressure, with eco friendly waste to energy systems. If the people in rural communities are aware about all the consequences of excess waste generation and the benefits of waste treatment at source, then the burden on local municipal bodies would fall. The central and state governments may jointly demonstrate how decentralized approach can work by setting up at least one decentralized waste processing facility in each state at full government cost and technical assistance and also utilizing it as a training ground and opportunity for other local bodies to follow. However, the journey towards a bio based circular economy has only just started, efforts need to be taken to overcome the barriers related to the spread of biogas solutions and enhance bio based circular economy by encouraging BETP.

## CONCLUSIONS

It is worth concluding that the goal of strengthening bio-based economy is potentially related to biogas solutions with respect to solid waste management in several ways. Biogas solutions could contribute towards creating a bio-based economy provided that the importance and socio-economic benefits associated with biogas solutions is realised by the rural and urban population, citizens, policy makers and the respective governments. Among the existing technologies for waste treatment, anaerobic digestion (AD) plays a key role in reducing waste along with the generation of renewable energy as it mitigates direct and indirect GHG emissions, recycles nutrients in the form of organic fertilisers, prevents nitrogen leakage into groundwater and avoids the spread of harmful diseases through land filling. Valorization of organic solid waste by implementing decentralized biogas solutions in rural and urban India resolves the need for energy along with a potential solution for the disposal of wastes. Biogas solutions are generally versatile, flexible and cost efficient from a societal perspective and when adapted to local conditions may contribute to sustainable development as well.

**TABLE 2:** Technical features and revenue obtained from biogas plants based on AGR technology installed by M/s AESPL.

| |
|---|
| Quantity of food waste: 1 ton/day |
| Biogas generation: 120 to 140 $m^3$/day |
| Bio-manure generation: 150 kg/day |
| LPG replacement: 50 - 60 kg/day (4 domestic LPG cylinders) |
| Land area requirement for the plant: 55 $m^2$ |
| Manpower requirement (semi-skilled): 2 people |
| Revenue/day from biogas = (4 LPG cyliners x USD 12) = USD 48 per day |
| Revenue/day from Bio-manure: 150 kg x USD 0.08 (8 cents) = USD 12 per day |
| Total Revenue per day: USD 48 + USD 12 = USD 60 |
| Total Revenue per annum (considering 300 working days): USD 18,000 |
| Capital cost – USD 46,153 |
| Flat pay back of capital investment: about 2.5 years |

*Note: One USD = 65 INR; revenue created by the plant is calculated based on LPG replacement with biogas excluding biomanure sales.*

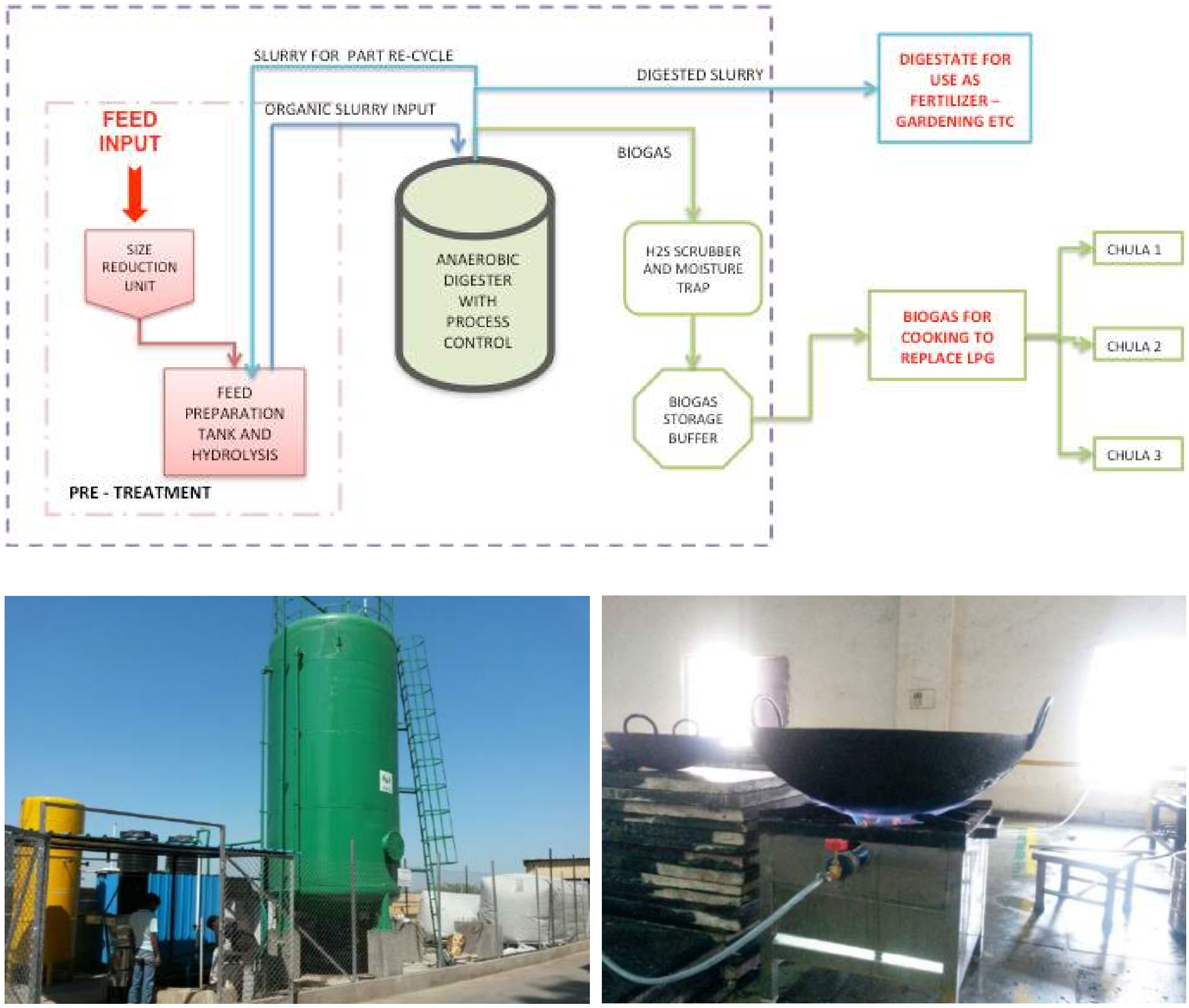


**FIGURE 1:** Biogas plant layout and full-scale installation at site based on AGR technology at the kitchen of TAPF at Bellary in Karnataka.

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# INVESTIGATION OF MUNICIPAL SOLID WASTE (MSW) AND INDUSTRIAL LANDFILLS AS A POTENTIAL SOURCE OF SECONDARY RAW MATERIALS

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**ABSTRACT**
Many of the secondary raw materials (SRM) in landfills constitute valuable and scarce natural resources. It has already been recognised that the recovery of these elements is critical for the sustainability of a number of industries and SRM recovery from anthropogenic waste deposits represents a significant opportunity. In this study, the characterisation of the different waste fractions and the amount of SRM that can potentially be recovered from two landfill sites in Finland is presented. The first site was a municipal solid waste (MSW) landfill site and it was specifically investigated for its metals, SRM, plastics, wood, paper, and cardboard content as well as its fine fraction (<20 mm). The second site was an industrial landfill site containing residual wastes from industrial processes including 1) aluminium salt slag from refining process of aluminium scrap and 2) shredding residues from automobiles, household appliances and other metals containing waste. This site was investigated for its metals and SRM recovery potential as well as its fine fraction. Results suggest that the fine fraction offers opportunities for metal (Cr, Cu, Ni, Pb, and Zn) and SRM extraction and recovery from both landfill site types while the chemical composition of the industrial waste landfill offered greater opporutinity as it was comparable to typical aluminium salt slags. Nevertheless, the concentrations of rare earth metals (REE) and other valuable elements were low even in comparison with the concentrations found in the Earth's crust. Therefore mining landfill sites only for their metals or SRM content is not expected to be financially viable. However, other opportunities, such as waste-derived fuels from excavated materials especially at MSW landfill sites, still exists and fosters the application and feasibility of landfill mining.

## 1. INTRODUCTION

The issue of resource security has come to the forefront of the debate as Critical Raw Materials (CRM) and Secondary Raw Materials (SRM) supply is fundamental to maintain and develop EU economy. For example, recent studies predicted that the depletion of silver, copper and platinum group metals (PGM´s) reserves will occur in the next 17-30 years and a peak in supply will be required within the next 20-35 (Sverdrup et al., 2014a,b; Sverdrup and Ragnarsdottir 2016). Sverdrup and Ragnarsdottir (2016) further stressed the importance of metals recycling, metal conservation and elimination of dissipative losses so that the society can become more sustainable with respect to metals supply. For PGM alone, they predict that extraction will reach a maximum in the period 2020–2050 and that market supply will peak in 2070–2080. For copper, the peak production estimates are much closer, from 2031 to 2042. In a longer perspective, taking into account price and recycling, the supply of copper to society is expected to run out sometime after 2400 (Sverdrup et al., 2014b). Thus, considering the increasing scarcity and raising prices of SRM, their recovery from anthropogenic deposits such as urban and mine

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wastes disposal sites is essential. In fact a great amount of waste can be regained as practical and valuable SRM by enhancing the recovery processes from industrial, extractive and municipal solid waste (MSW) landfill sites especially if we consider that Europe is highly dependent on the imports of certain raw materials including rare earth elements (REE) and SRM. Europe has between 150,000 and 500,000 landfill sites, with an estimated 90% of them being “non-sanitary” landfills, pre-dating the EU Landfill Directive of 1999 (Jones et al., 2013). These older landfills tend to contain municipal solid waste and often lack any environmental protection technology. To avoid future environmental and health problems, many of these landfills will soon require expensive remediation measures. This situation does present us with an exciting opportunity for a combined resource-recovery and remediation strategy, which will drastically reduce future remediation costs, reclaim valuable land, while at the same time unlocking billions of tonnes of valuable resources contained within these landfills (Gutiérrez-Gutiérrez et al., 2015; Dino et al., 2016; Dino et al., 2017). There is however to date no inventory available of SRM and CRM present in EU landfills. There has been only very limited knowledge around best practice and how to manage the excavation and recovery of valuable materials.

There are still challenges in enhanced landfill mining (ELFM). We need to understand more about each of the stages involved: the exploration, separation, transformation and up cycling technologies, and how these can be applied in the best way in dealing with the differing urban and industrial landfill sites. For instance, to recover recyclable materials such as metals and plastics, we need to consider their chemical degradation as they may not be suitable for conventional recycling. The recyclable materials are in a moist environment and emerge with soil/clay covering and attachments. As such, we need to find the best cleaning approaches.

There are also policy challenges in establishing legal frameworks for ELFM but concerted action is underway to overcome them (EURELCO, 2017). The potential of ELFM was presented to the European Parliament last year. It has received backing from the the European Commission in May 2017 by acknowledging in their ‘Closing the Loop – EU Action Plan for the Circular Economy’ that an increment in reuse and recycling of key waste streams has to be undertaken and made a specific reference to ELFM. When considering ELFM, some actions should be forecasted to asses the sustainability of the mining opportunity: (i) the estimation of the amount of types of waste materials; (ii) the characterisation and localization of the different wastes present in landfill; (iii) their potential recovery and treatability for their utilization (Kaartinen et al., 2013). Burlakovs et al. (2017) further critically summarised LFM challenges from historical sites and driving paradigms of LFM from ‘classical hunting for valuables’ to ‘perspective in ecosystem revitalization’. Krook et al. (2012) also reviewed the main trends, objectives, topics and findings of 39 research papers on landfill mining published during the period 1988–2008. Together with a follow up review by Krook and Baas (2013), they conclude that urban mining and landfill mining show high potential but the state-of-the-art is still too theoretical and requires large pilot scale demonstration to assess and demonstrate the performance of such activities in practice.

Several materials and energy fractions can be exploited from landfills. Wagner and Raymond (2015) are among the first authors to present an economically succesful LFM case study where metals were recovered. In another study, Leme et al. (2014) showed that the integration of waste-to-energy plant to a LFM project is highly dependent on the MSW treatment fees due to high installation, operation and maintenance costs. Quaghebeur et al. (2013) investigated different valorization options for the excavated materials from the REMO site in Houthalen (Belgium). The results showed differences in the composition and the characteristics of the waste materials with regard to type of waste (MSW versus IW) and the period during which the waste was stored. For the plastics, paper/cardboard, wood and textile recovered, waste-to-energy was the most suitable valorization route. Quaghebeur et al. (2013) also concluded that metals, glass/ceramics, stones and other inert materials can be valorized as well if the materials will be separated from each other. The amount of combustibles in the excavated waste confirmed the large potential of waste-to-energy for landfill mining. Fine fractions from the industrial waste contained high concentrations of heavy metals (Cu, Cr, Ni and Zn) and thus offer opportunities for metal extraction and recovery. Also the biochemical methane potential (BMP) of the fine fractions from two Finnish landfills have been investigated and showed good opportunities for further energy from waste recovery (Sormunen et al., 2008; Mönkäre et al., 2016). The present paper presents two ELFM pilot case studies carried out in Finland. One of the site was a MSW landfill site (Metsäsairila) and the other one an industrial waste (IW) landfill site (Vierumäki). Detailed site investigation of the two sites was carried out to evaluate the potential SRM resources that can be exploited from different landfills. The described characterisation was part of a wider activity related to the Smart Ground H2020 project (Grant number 641988) which aims, together with other objectives, to foster resource recovery from both urban solid waste landfill sites and mine waste disposal sites by (i) improving the availability and the accessibility of data and information on SRM amount in EU anthropogenic deposits and (ii) integrating data from existing databases and new information collected into a single EU database.

## 2. MATERIALS AND METHODS

### 2.1 Landfill sites description

The first site, Metsäsairila, is a MSW landfill site located in the South-Eastern region of Finland, nearby the City of Mikkeli. MSW buried in the site is collected from approximately 55 000 inhabitants. The site has been operating since beginning of 1970´s and is divided in two distinct cell areas: a closed one and an active operational one. The active cell area is located in the northern part of the landfill. The active cell is membrane-lined with a mixture of bentonite and moraine on the bottom structure; in contrast the closed area is located on swamp. Both active and closed cells have collection system for leachate. Landfill gas which is mainly collected from the closed cell and used for combined heat and electricity production on site. The height of the waste filling was estimated to be around 20-25 meters in the closed cell and between 6 and 10 m in the active cell. The closed cell is currently being capped with a layer of clay and silt moraine and will be completed in 2018. The surface area of the closed cells is around 8 ha while the active cells surface area is around 3 ha. The active area has received waste since 2007. The second site, Kuusakoski Oy's, is an industrial landfill site located in Vierumäki, southern Finland. The site started receiving waste in 1974 and has been closed in three stages in 1989, 1990 and 1991. The wastes disposed of in the landfill are residues from industrial processes including 1) aluminium salt slag from refining process of aluminium scrap and 2) shredding residues from automobiles, household appliances and other metals containing waste. The area of site is estimated to be approximately 2.5 hectares. Typical to a landfill of this age, there are no engineered bottom isolation layers at the landfill, and a peat layer has been used as a compacting bottom structure. The height of the waste filling was estimated to be ranging between 5 to 8 meters. After completion, the waste was covered with a layer of clay functioning as a sealing layer, moraine and a layer for vegetation. Today, the landfill site is reminiscent to a typical young forest.

## 2.2 Sampling, sorting and analysis of collected samples

Geophysics characterization was carried out at Metsäsairila landfill site as described previously by Lahti et al. (2005). By using geophysics it was possible to direct the sampling to the most appropriate points of the landfill site and also to get broader information of the physical properties of the landfill material. The geophysics characterization was carried out only in the closed cell area as in the active cell it was too many confounding factors to make the geophysical field measurements. Electrical resistivity tomography (ERT), Induzed polarization tomography (IPT), Magnetic and Electromagnetic (EM) methods were used together in order to get the best result in searching the metal containing areas (Lerssi et al., 2016). Gravity method was used to determine the bedrock level and also the thickness of the landfill material (Vanhala et al., 2005; Valli and Mattsson, 1998). By using gravity it was possible to determine the maximum drilling depth to avoid the damages on the landfill bottom. Five sampling points were then drilled by hydraulic piling rig in the areas with the highest conductivity and total magnetic intensity (Figure 1). Samples with codes DH1, DH2a and DH3 were from the closed part of the landfill site and samples with codes DH6 and DH7 were from the currently operational part of the landfill for waste disposal.

The amount of waste materials collected at each sampling point is summarized in Table 1. Samples were moved to sorting point where they were manually sorted by sieves to different particle size categories (>100 mm, 20-100 mm and <20mm) and waste fractions (metals, wood, paper, plastics, textile, soil and others). Waste fraction separation was done to fractions size of 20-100 mm and >100 mm.

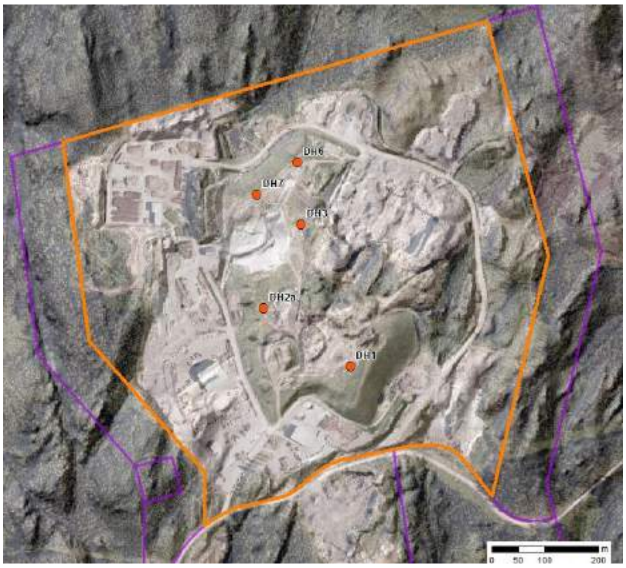


**FIGURE 1:** Topographic map of Metsäsairila MSW landfill site obtained using laser scanning based DEM with 2 metres resolution overlaid with aerial photography (orange dots: location of the sampling points).

Analysis of the fine material samples (<20mm) for critical raw materials (CRMs) content was carried out by an external laboratory (ALS Finland Oy, Finland). Reference method used was based on US EPA 200.8, CSN EN ISO 17294-2 and US EPA 6020 (measurements were done by inductively coupled plasma mass spectrometry (ICP-MS).

Unmanned Aerial Vehicle (UAV) photogrammetry survey of the Vierumäki industrial landfill site was conducted for visualisation of topography before the physical exploration of site was carried out. Topographic and morphologic 3D characterization of the site will obtain a detailed reconstruction of the topographic surface. This will give better overview about structure and composition of the investigated pilot site. Photogrammetry is a viable alternative for calculating landfill volume which is useful for the SRM´s volume evaluation on site. Figure 2 shows an orthophoto of the Vierumäki industrial landfill site with cell size of 5 x 5 cm.

Sampling at Vierumäki industrial landfill site was done with an excavator from five sampling points to cover the landfill area as well as possible with limited amount of time and resources (Figure 3).

During the excavation, it was noticed, that the landfill had well defined layers which were attributed to the aluminum salt slag and the shredding residues (Figure 4). The

**TABLE 1:** Amounts of aggregate waste materials collected at the Metsäsairila MSW landfill site.

| Sample ID | Sample depth (m) | Amount of aggregate waste materials (kg) |
|---|---|---|
| DH1 | 3.5-17 | 406.0 |
| DH2a | 3-12 | 192.3 |
| DH3 | 2.5-10 | 277.4 |
| DH6 | 0.2-5 | 282.2 |
| DH7 | 0.2-5 | 284.4 |

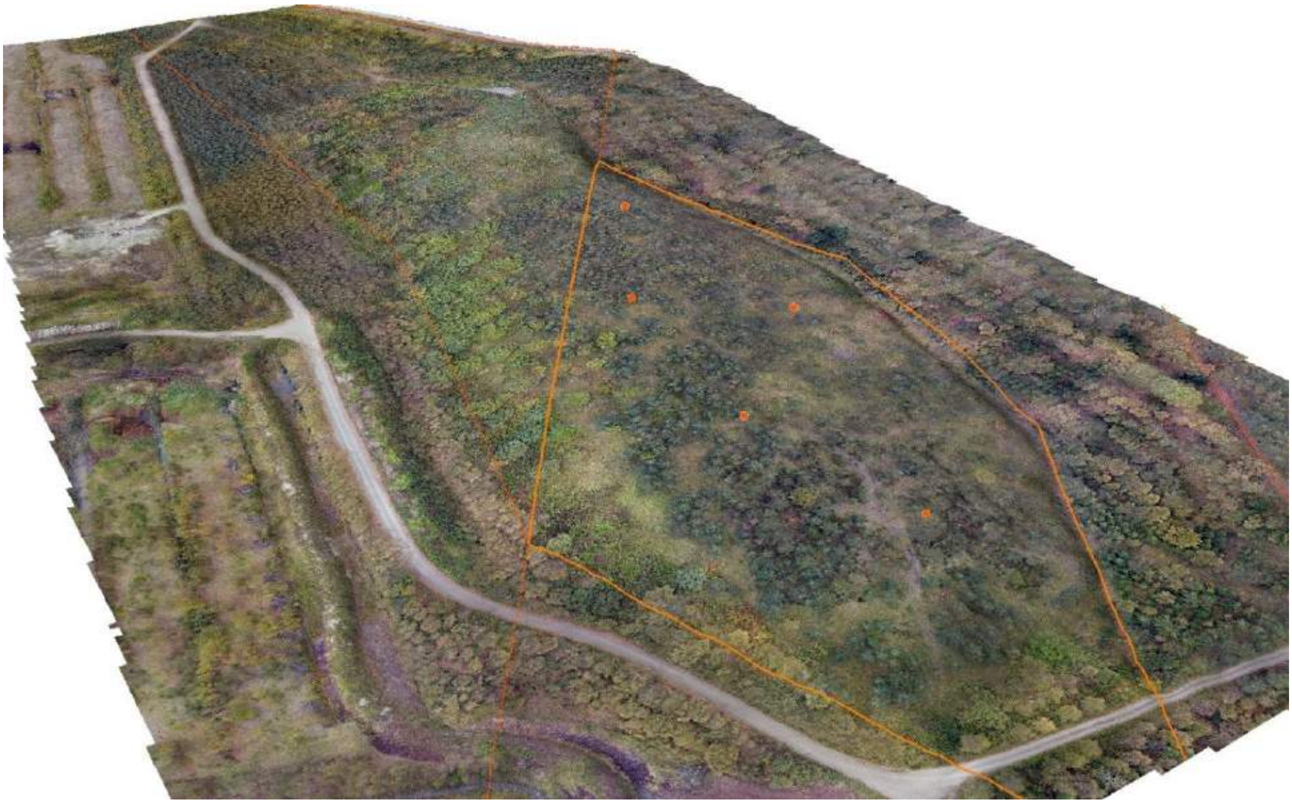

**FIGURE 2:** 3D site topography based on DTM with 50 cm resolution and draped orthophoto.

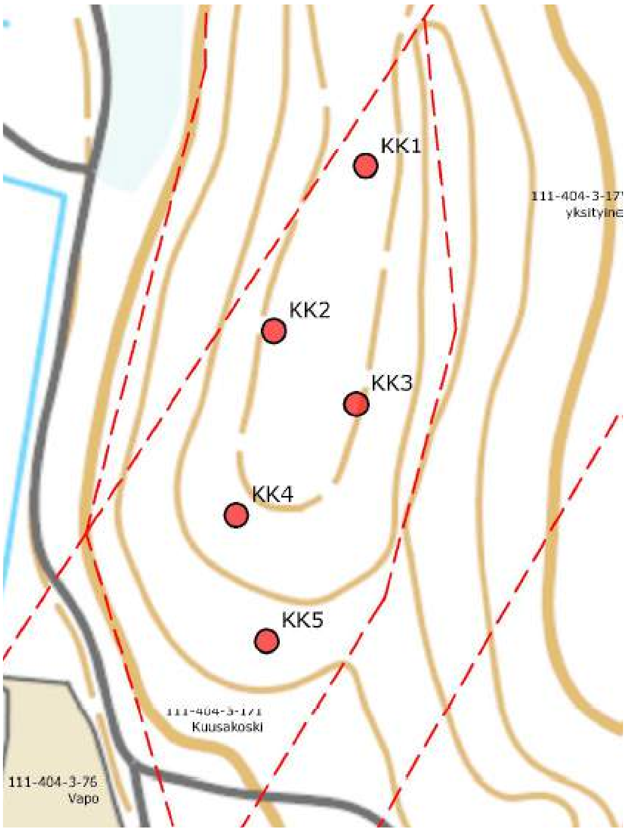


**FIGURE 3:** Locations of sampling points at the industrial landfill. Red dots show the sampling points and the dashed line surrounding the dots show the rough borders of the industrial landfill.


**FIGURE 4:** Landfill layers at the Vierumäki industrial landfill.

estimated height of each layer on each sampling spot was recorded to enable calculations of material amounts in the landfill.

After excavating the cover layers (moraine and clay) the two waste layers were mixed together to obtain a representative composite sample from the excavated waste materials as summarised in Table 2.

The composite samples were manually sieved to different particle size categories >100 mm, 20-100 mm and <20mm. The two largest particle size categories, >100 and 20-100 mm, were sorted to different waste fractions (metals, combustibles, soil and others). The fine fractions <20 mm and the combustible fractions (20-100 mm and >100 mm combined from each sampling point) were analysed for Al, Mg, Cu, Sb, Co and Cr by XRF. A composite sample of all fine fraction samples and one composite sample from all combustible samples were also analysed for Sc, Y, La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Yb, Pt, Pd, Ru, In, Ag and Au by Aqua Regia dissolution and subsequent analysis by induced coupled plasma mass spectrometry (ICP-MS) as described in Kaartinen et al. (2013). In addition, the calorific values of the combustible samples were determined with a bomb calorimetry by ALS Finland Oy (European Standards, 2011). The calorific value is an important quality attribute as it indicates the amount of recoverable energy from waste.

## 3. RESULTS AND DISCUSSION

### 3.1 Metsäsairila MSW landfill site

The geophysical characterisation carried out at Metsäsairila provided significant new information of the landfill waste layers composition in both horizontal and vertical directions, especially for identifying the best locations for the presence of metals and determining the dimensions waste materials that should be excavated. Figure 5 shows the 3D ERT results together with the magnetic data and the bedrock topography interpreted from gravity data. The sampling places were selected within the areas with high magnetic intensity and electrical conductivity as it was indicating high metal content within the buried waste materials.

The mass distribution of the different waste fractions from the five core samples is summarised in Table 3. The mass distribution was relatively similar between the samples from the different sampling points despite their location and the aged of the buried waste (i.e. DH3 was located in the old closed area while DH6 was in the active area). Sorted fractions >100 mm and 20-100 mm which were combined from all wells together consisted mainly from

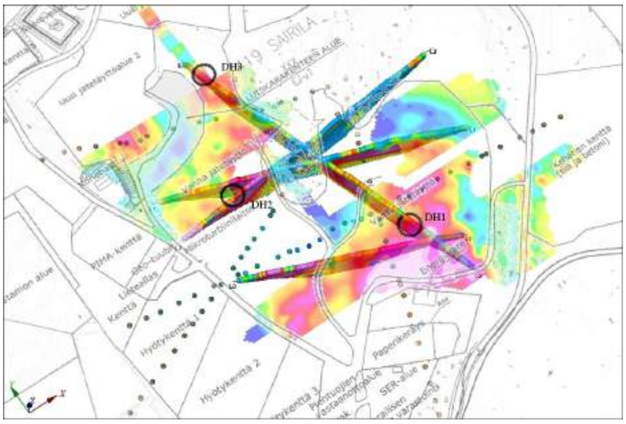


**FIGURE 5:** Electrical conductivity cross sections from 3D ERT profiles and total magnetic anomaly 2D map data with realized drill holes D1-D3 (sampling points). Interpreted bedrock topography of gravity profiles is visualized in the picture as coloured circles. In all the data the red colours show the high and blue colours show the low values.

energy fraction (plastic, paper, wood, cardboard, 76%), metals (5%), soil (17%) and others 2%. Results are following similar trend than research implemented by Kaartinen et al. (2013) in MSW landfill in Kuopio, Finland. They observed that amount of fine material (<20 mm) was found to be ca. 50% (w/w) which also supports previous reports of the amount of the fines (Quaghebeur et al., 2013). Fine material consisted mainly of landfilled wastes but also of the landfill cover materials (usually soil). In our study case fine fraction varied from 37% to 47% depending on the sampling point. Mainly the fine fraction included soil material but also small particles of plastic, paper and wood were present. The two main fractions were fine material (<20 mm) and the energy fraction comprised of wood, paper and cardboard, plastic and textiles. Sorted size fractions >100 mm and 20-100 mm from every sampling point had a similar waste distribution and main interesting fractions were the one considered for energy recovery and the fine material fraction (<20 mm).

The closed area of the Metsäsairila MSW site represents about 960 000 t of MSW of which metals account for 3.7% and the combustible energy fraction (wood, paper and cardboard, plastic and textiles) 42% (Table 4).

According to the analysis of metals and the fine fractions of the sorted samples, the excavated waste samples contained primarily Ba, Cr, Cu, Zn and Pb. Amounts of Ag, Au and In were rather low. The concentrations of heavy metals were lower than in the studies of Quaghebeur et al.

**TABLE 2:** Vertical distribution of the waste layers at Vierumäki industrial landfill.

| Sampling point | KK1 | KK3 | KK4 | KK5 |
|---|---|---|---|---|
| Cover layers (m) (organic growth layer+moraine+clay) | 0 – 1.8 | 0 – 1.0 | 0 – 1.0 | 0 - 1.0 |
| Waste layer depth (m) from - to (in meters from ground) | 1.8 – 5.5 | 1.0 - 3.5 | 1.0 – 5.0 | 0.8 - 4.5 |
| Shredding residues layer (m) from - to | 1.8 - 3.0 | 1.0 - 3.0 | 1.0 - 1.7 | 0.8 - 2.8 |
| Aluminum salt slag layer (m) from - to | 3.0 - 5.5 | 3.0 - 3.5 | 1.7 - 5.0 | 2.8 - 4.5 |
| Mass of composite sample to manual sorting (kg) | 531 | 252 | 288 | 242 |

**TABLE 3:** Weight distribution of different waste fractions in collected aggregate samples.

| Waste fractions | DH1 | DH2a | DH3 | DH6 | DH7 | Average |
|---|---|---|---|---|---|---|
| **>100 mm** | **111.51** | **68.23** | **50.03** | **69.57** | **81** | **76.1** |
| metal | 6.54 | 9.3 | 3.75 | 2.45 | 1.7 | 4.7 |
| wood | 8.9 | 11 | 3.4 | 5.06 | 13.6 | 8.4 |
| paper and cardboard | 8.15 | 11.92 | 4.27 | 5.52 | 8.8 | 7.7 |
| plastic | 44.2 | 30.4 | 30.96 | 41.2 | 27.8 | 34.9 |
| textiles | 13.92 | 4.38 | 5.73 | 8.99 | 28 | 12.2 |
| soil | 29.8 | 1.23 | 1.92 | 6.35 | 1.1 | 8.1 |
| others | 0 | 0 | 0 | 0 | 0 | 0 |
| **20-100 mm** | **124.71** | **52.7** | **101.76** | **78.84** | **75.18** | **86.6** |
| metals | 2.82 | 3.19 | 6.76 | 2.22 | 1.54 | 2.8 |
| wood | 25.49 | 8.02 | 12.6 | 20.9 | 14.52 | 13.6 |
| paper and cardboard | 12.37 | 6.77 | 12.5 | 8.7 | 10.5 | 8.5 |
| plastic | 30.2 | 19.8 | 36.4 | 20.5 | 14.44 | 20.2 |
| textiles | 18.11 | 4.07 | 4.41 | 2.8 | 5.2 | 5.8 |
| soil | 34 | 10.12 | 25 | 19.9 | 26.18 | 23 |
| others | 1.72 | 0.73 | 4.09 | 3.82 | 2.8 | 2.6 |
| **<20 mm** | **169.8** | **71.4** | **125.6** | **133.8** | **128.2** | **125.8** |
| **Total mass (kg)** | **406.02** | **192.33** | **277.39** | **282.21** | **284.38** | **288.5** |

**TABLE 4:** Estimated amounts of the different waste materials in the closed area of the Metsäsairila MSW landfill site.Uc iam te

| | Average (%) | Estimated total amount (t) |
|---|---|---|
| Metals | 3.70 | 35 474 |
| Wood | 7.93 | 76 088 |
| Paper and cardboard | 6.39 | 61 366 |
| Plastic | 21.92 | 210 430 |
| Textiles | 5.78 | 55 490 |
| Soil | 11.66 | 111 891 |
| Others | 0.75 | 7 169 |
| Fine fraction | 41.40 | 397 440 |
| Total | 100 | 955 348 |

(2013) and Gutiérrez-Gutiérrez et al. (2015). Specifically, Gutiérrez-Gutiérrez et al. (2015) assessed the content of four UK MSW landfill sites and reported concentrations for rare earth elements (REE) (including Sc, Y, La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu) of 220 ±11 mg/kg, PGM (Pt, Pd, Ru) 2,1 ± 0,2 mg/kg, and other critical metals such as Li, Ln, Sb, Co of 156 ±7 mg/kg. In this study at Metsäsairila MSW landfill site, the REE concentration was twice lower (87±13 mg/kg). Concentrations of Pt, Pd and Ru were also lower, less than 1.5 mg/kg. However Ce was found to be the most abundant rare metal in our study as in Gutiérrez-Gutiérrez et al. (2015). This finding underlines the differences in the composition and the characteristics of the waste materials in different MSW landfills with regard to type, location and the period connected to landfilling activity. Based on these results, it is obvious that the quantity of REE and PGM that can be recovered from the waste materials will be much lower than what would be extracted by mining natural ores. In addition, at the end of the extraction process the metals are concentrated in acid solution which must be treated to separate the metals of interest. Achieving a level of purity above 99% becomes a major inconvenient to make the recovered metals highly valuable. Also concentrations are in same range than for ordinary soil so it is predicted that extracting them from fine fraction would not give extra benefit for MSW landfill mining.

### 3.2 Vierumäki industrial waste landfill site

In contrast to the Metsäsairila MSW landfill site, the fine fraction <20 mm had by far the greatest mass share of all the samples in the Vierumäki industrial waste landfill site (on average 74 ± 7% (n=4)). The 20-100 mm fraction represented 20 ± 7% and the >100 mm fraction 6 ± 3% of the waste samples. From visual observation, the fine fraction consisted mainly of the aluminium salt slag. Based on the field observations of the landfill layers (Table 2), a simplified cross section of the landfill site was estimated as follows: 1 m of cover layers, 1 m of shredding waste and 3 meters of aluminum salt slag. Together with the estimated landfill area of 2.5 hectares and the results from manual sorting, the masses of different material types at the landfill were estimated as shown in Table 5. Here the fine fraction <20 mm is regarded as aluminium salt slag.

The average concentrations of the critical metals, REE and PGM in the fine fraction and the combustible fraction of the samples are summarized in Table 6. The fine fraction <20 mm had characteristics comparable to typical aluminium salt slags. The concentrations of REE and other valuable elements were in contrast very low even in comparison with the concentrations found in the Earth's crust (USEPA, 2012).

The average calorific value of the combustible fractions

**TABLE 5:** Estimation of the material amounts at Vierumäki industrial landfill site.

| Material | Mass (t) |
|---|---|
| Cover layers | 25 000 |
| Shredding waste total of which | 20 000 |
| Fine fraction (Al salt slag) | 15 600 |
| Combustibles | 4400 |
| Metals | 72 |
| Soil | 578 |
| Other (mainly large pieces of Al salt slag) | 379 |
| Al salt slag class (from separate layer) | 75 000 |

was 22±4 MJ/kg which is good compared to the heating value of other materials such as lignocellulosic materials normally ranging between 12.2 and 20.6 MJ/kg, biochar between 27.4 and 32.6 MJ/kg, plastics and synthetic rubber between 37.8 and 38.00 MJ/kg and cardboard 13.81 MJ/kg (Boumanchar et al., 2017).

## 4. CONCLUSIONS

Based on the characterisation of the exacavated waste samples from both landfill site types the amounts of critical metals, REE and PGM were not high enough to justify landfill mining and recovery alone. The main fractions from the MSW landfill site were by order of contribution, the fine fraction (<20 mm), followed by the combustible fraction and the metallic products (ferrous and non-ferrous products). Thus the economic viability of landfill mining could be increased by recovering additional material fractions such as plastics, paper, cardboard and wood for energy production. In contrast the aluminium content of the fine fraction of the IW landfill site is offering good opportunity for recovery. Overall, other opportunities exist that together form the concept of ELFM. Waste-derived fuels from excavated materials have the potential to be highly energetic. From both landfill sites investigated, the energy potential is comparable to the levels of energy of Refuse-Derived Fuels (RDF) produced from non-landfilled wastes. Ultimately, the mining and recovery approach leads to a further commercial opportunity in the land itself, reclaimed and the soil remediated, making it available again for housing, industrial estate development or other forms of development. Abandoned landfill sites present environmental and human health risks that can involve large taxpayer investments to clear up. Nonetheless, there are still several challenges in ELFM, which means that further research and development is needed before the full potential will be realised.

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**TABLE 6:** Average content (%) of REEs, PGM, critical metals and others found in the Vierumäki industrial landfill site.

| Element (%) | Fine fractions <20 mm average, n=4 (standard deviation) | Combustible fractions average, n=4 (standard deviation) | Typical values for aluminum salt slag (Huang et al., 2014) |
|---|---|---|---|
| Al | 13 (5.4) | 1.7 (0.21) | 14.2 |
| Mg | 1.6 (0.17) | 0.73 (0.15) | 2.0 |
| Cu | 0.26 (0.07) | 0.14 (0.10) | 0.088 |
| Sb | <0.01 (-) | <0.01 (-) | - |
| Co | <0.01 (-) | <0.01 (-) | - |
| Cr | 0.03 (0.02) | 0.01 (0.004) | 0.033 |
| **Element (mg/kg)** | **Fine fractions <20 mm composite sample** | **Combustible fractions composite sample** | **Crustal abundance (US EPA, 2012)** |
| Er | <0.50 | <0.50 | 2.1 |
| Eu | <0.50 | <0.50 | 1.3 |
| Au | <0.50 | <0.50 | 0.003 |
| Pd | <0.50 | <0.50 | - |
| La | 5.6 | 4.5 | 30 |
| Y | 9.7 | 2.2 | 24 |
| Pt | <0.50 | <0.50 | - |
| Ce | 11 | 8.4 | 60 |
| Nd | 5.0 | 3.8 | 27 |
| Ru | <0.50 | <0.50 | - |
| Pr | 1.2 | 1.0 | - |
| Sm | 0.81 | 0.75 | 5.3 |
| Gd | 0.72 | 0.68 | 4.0 |
| Tb | <0.50 | <0.50 | 0.7 |

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# A DECISION SUPPORT TOOL FOR ENHANCED LANDFILL MINING

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**ABSTRACT**

Enhanced Landfill Mining has great potential to reduce the negative effects of landfills on both the environment and human health, to reclaim valuable land and provide a new source of raw materials. However, uncertainties in economic feasibility and environmental and social outcomes act as a bottleneck to its widespread uptake. Here, we present a decision support tool (DST) which aims to reduce these uncertainties by assisting site operators in assessing the economic, environmental and social consequences of a proposed project, while also evaluating the best technology train to use and the amount of rare earth elements (REE) present. Such a tool is the first of its kind and we propose its use as an initial assessment aid prior to more complex modelling of project feasibility in order to increase the uptake of enhanced landfill mining practices in the field of sustainable waste management.

## 1. INTRODUCTION

Enhanced Landfill Mining (ELFM) refers to the process of excavating waste materials that have previously been disposed of from landfills and valorising these historic waste streams as both materials (waste-to-material, WtM) and energy (waste-to-energy, WtE) (Jones et al., 2013).

Europe has an estimated 150,000 to 500,000 landfill sites with a predicted 90% pre-dating the EU Landfill Directive of 1999 (Jones et al., 2013). These older landfill often lack environmental protection technology and will soon require expensive remediation measures in order to avoid harm to human health and the environment. In addition, from previously following a linear economy model of take-make-dispose, "non-sanitary" landfills currently store an abundance of valuable materials as waste, including secondary raw materials (SRM), critical raw materials (CRM) and rare earth elements (REE), and therefore represent a huge untapped resource (Gutierrez-Gutierrez et al., 2015; Laner et al., 2016). Such resources are finite and under increasing demand due to the emergence of new economies (Wante and Umans, 2010). These resources are currently sourced primarily from outside of Europe, making Europe vulnerable to price fluctuations as a result of global demand (Lapko et al., 2016).

ELFM has the potential to drastically reduce remediation costs, provide new resources in the shape of SRM, CRM and REE from within the EU and reclaim valuable land (Gutierrez-Gutierrez et al., 2015; Laner et al., 2016). As a result, there has recently been an increased interest in the application of ELFM and the European Parliament has recently taken the decision to add ELFM to the Landfill Directive (EURELCO, 2017). However, due to uncertainty regarding the economic feasibility and social and environmental consequences of ELFM (Frändegård et al., 2013a; Danthurebandara et al., 2015), this concept is not presently broadly implemented by operators. To date, there is no tool available to assess clearly the economic feasibility of a proposed ELFM project, to evaluate the environmental and social impacts or to identify the best process to use.

This paper aims to present a Decision Support Tool (DST) that uses a step-wise approach to assess the best ELFM process, the expected economic output and the social and environmental impacts of ELFM. The DST was developed to deal with municipal solid waste (MSW) and commercial and industrial waste (C&I) and provides 5 waste composition mixes as default parameters taken from a literature review. The tool also gives the users the option to input their own waste composition, along with other input parameters. Furthermore, 9 processing scenarios were developed, considering the following technologies: soil flushing, excavation, screening, shredding, air separation, ballistic separation, magnetic separation, Eddy-current separation, and Advanced Thermal Treatment (ATT). For each scenario, on-site/off-site was considered and 3 options are proposed as follows: (i) all the treatments are done on-site (no transportation), (ii) the sorting is done on site but the refuse-derived fuel (RDF) is transported to an off-site Waste to Energy facility (transport WtE only), (iii) the excavated waste is only

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screened on site, then transported to a Waste Treatment Facility (WTF) for sorting, and the RDF transported again to a WtE facility for recovery (Transport WTF + WtE). The tool determines environmental, social and economic indicators for each scenario using multi-criteria analysis and the best scenario approach from a sustainability standpoint for landfill mining is identified. The tool also estimates the amount of REE present in landfill, determined by literature review.

## 2. METHODS

### 2.1 Determination of Typical Waste Composition

The typical waste fractions and composition of municipal solid waste (MSW) and construction and industrial waste (C&I) were determined based on six published case studies across Europe (Table 1).

A weighted average approach was then used to define the 5 waste compositions scenarios considered by default by the DST (Table 2). However, results obtained with this simple model approach should be taken with caution because the waste composition and fractions at site can vary significantly and either over or underestimate of the percentage of certain fractions. The DST is based on values taken from literature which will need to be updated over time as new data will become available to better reflect landfill waste composition processes.

### 2.2 Determination of ELFM Scenarios

The ELFM scenarios and technologies were based on a critical review of published articles and industry references. In our case, ELFM begins either by in-situ leaching/soil flushing and metal recovery or directly by waste excavation. The waste is then sorted with various techniques and the calorific fraction is recovered as RDF by ATT (Table 3).

**TABLE 1:** Typical waste composition of MSW and C&I landfill sites.

| Case study | Waste Composition | | | | | | | C&I | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | Average | 5 | 6 | 2 (site 4) | 2 (site 5) | Average |
| Location | (site 4) | 2 | Belgium | Belgium | Germany | Belgium | | Germany | Belgium | Belgium | Belgium | |
| <10 mm soil type | (site 5) | Average | 44% | 43% | 46% | 43% | 46 | 27% | 62% | 70% | 58% | 60% |
| Plastic | 8% | 25% | 17% | 12% | 9% | 33% | 17 | 33% | 19% | 7% | 4% | 15% |
| Paper/card | 8% | 14% | 8% | 2% | 5% | | 7 | - | - | 1% | 3% | 2% |
| Wood | 7% | 4% | 7% | 9% | 10% | | 7 | - | - | 2% | 12% | 8% |
| Textile | 3% | 3% | 7% | 4% | 3% | | 4 | - | - | 2% | 2% | 2% |
| Glass | - | - | - | - | 2% | - | 2 | - | - | - | - | - |
| Stones, inert | 10% | 2.5% | 15% | 10% | 25% | 10% | 2 | 8% | 8% | 11% | 10% | 10% |
| Ferrous metals | 4% | 2% | 3% | 3% | 3% | 3% | 3 | 2% | 2% | 2% | 4% | 3% |
| Non-ferrous metals (REE) | 0.8% | | | - | | | 0.8% | - | - | - | - | - |
| Hazardous | 0.2% | - | - | - | - | - | 0.2% | - | | | | - |
| Organic waste | 3% | - | - | - | - | - | 3 | - | - | - | - | - |

[1] *Frändegård et al. (2013b)*, [2] *Quaghebeur et al (2013)*, [3] *Jones et al (2013)*, [4] *Danthurebandara et al (2015)*, [5] *Wanka et al (2016)*, [6] *Spooren et al (2012)*

**TABLE 2:** Waste composition used in this study.

| | MSW | | | | |
|---|---|---|---|---|---|
| **Waste Fraction** | **100% MSW** | **75% MSW, 25% C&I** | **50% MSW, 50% C&I** | **25% MSW, 75% C&I** | **100% C&I** |
| <10mm Soil Type | 45.9% | 49.5% | 53.1% | 56.8% | 60.4% |
| Plastics | 17.2% | 6.6% | 15.9% | 15.3% | 14.6% |
| Paper/Cardboard | 7.1% | 5.9% | 4.6% | 3.4% | 2.2% |
| Wood | 7.4% | 7.4% | 7.5% | 7.5% | 7.5% |
| Textiles | 4.0% | 3.5% | 3.0% | 2.5% | 2.0% |
| Glass | 1.6% | 1.2% | 0.8% | 0.4% | 0.1% |
| Stones (inert) | 10.4% | 10.3% | 10.3% | 10.2% | 10.2% |
| Ferrous Metals | 2.8% | 2.8% | 2.9% | 3.0% | 3.1% |
| Non-ferrous Metals (REE) | 0.8% | 0.6% | 0.4% | 0.2% | 0.0% |
| Hazardous | 0.2% | 0.2% | 0.1% | 0.1% | 0.0% |
| Organic | 2.7% | 2.0% | 1.3% | 0.7% | 0.0% |
| TOTAL | 100% | 100% | 100% | 100% | 100% |

**TABLE 3:** ELFM scenarios.

| S | Technologies train process | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| 1 | Soil flushing | - | - | - | - | - | - | - | - |
| 2 | Soil flushing | Excavation | Screening | Shredding | Ballistic separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) | - |
| 3 | Soil flushing | Excavation | Screening | Fines Ferrous metal separation | Shredding | Ballistic separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) |
| 4 | Soil flushing | Excavation | Screening | Shredding | Air separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) | - |
| 5 | Soil flushing | Excavation | Screening | Fines Ferrous metal separation | Shredding | Air separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) |
| 6 | Excavation | Screening | Shredding | Ballistic separation | Ferrous metal separation | Non-ferrous metal separation | ATT gasification) | - | - |
| 7 | Excavation | Screening | Fines Ferrous metal separation | Shredding | Ballistic separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) | - |
| 8 | Excavation | Screening | Shredding | Air separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) | - | - |
| 9 | Excavation | Screening | Fines Ferrous metal separation | Shredding | Air separation | Ferrous metal separation | Non-ferrous metal separation | ATT (gasification) | - |

*S = Scenario considered*

For each scenario, 3 transportation options are considered (Figure 1):

1. No transportation (all treatments occur on-site)
2. Transportation for WtE only (on-site sorting, RDF transported to WtE facility)
3. Transportation for WTF and WtE (excavated waste screened on-site only, transportation to WtF for sorting and RDF transported to WtE facility)

## 2.3 Model Outputs

The DST assesses the impacts of the landfill mining scenarios based on three criteria: Environment, Society and Economy. The indicator set used were adapted from the SuRF-UK indicator set for sustainable remediation assessment (Table 4, CLAIRE, 2011). It is important to note that, at the outset of the assessment, an equal number of indicators (five) were identified under each of the environ-

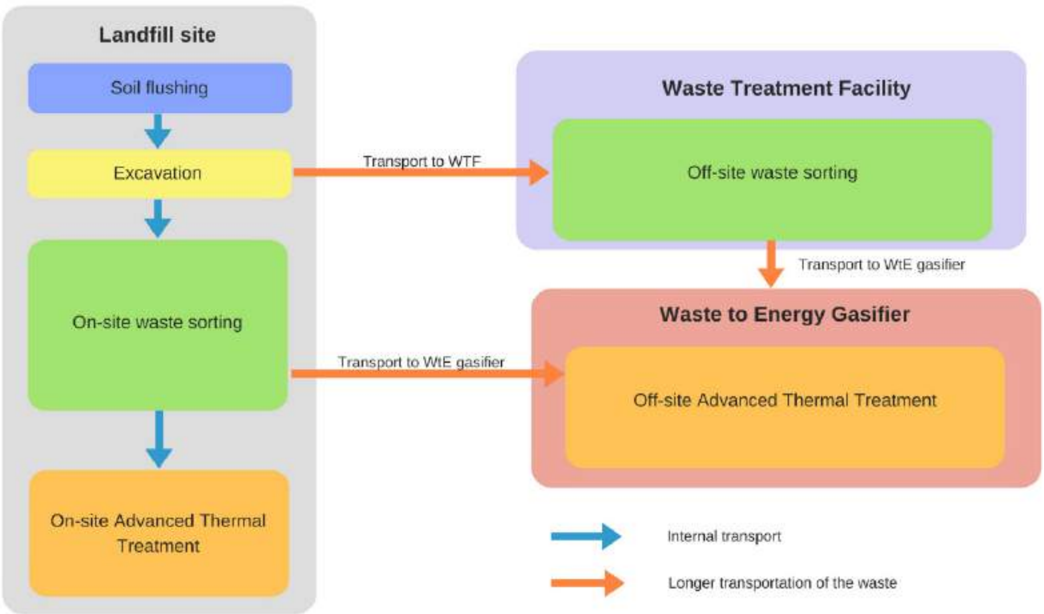



**FIGURE 1:** Transportation options.

**TABLE 4:** Sustainability indicator categories (adapted from CLAIRE, 2011).

| Indicator | Environment | Social | Economic |
|---|---|---|---|
| 1 | Emission to air | Human health & Safety | Direct economic costs and benefits |
| 2 | Soil and ground conditions | Ethics & Equity | Indirect economic costs & benefits |
| 3 | Groundwater and surface water | Neighbourhoods & locality | Employment and employment capital |
| 4 | Ecology | Communities & community involvement | Induced economic costs & benefits |
| 5 | Resource Use and Waste Generation | Uncertainty and evidence | Project lifespan & flexibility |

mental, social, and economic headlines (i.e., a total of 15 indicators across the sustainability assessment) (Table 4). This procedure ensured that, in the absence of individual indicator weighting, the three sustainability pillars were given equal weight (i.e., the outcome is not automatically biased by a disproportionate number of indicators in a single sustainability pillar). At higher levels of assessment, where stakeholder engagement and participatory processes seek to establish consensus on the relative weighting of the three components or their constituent indicators, it may be appropriate to apply weightings to the individual indicators to reflect the relative importance of different indicators to the stakeholders (CLAIRE, 2010).

The sustainability appraisal of the landfill mining options was carried out in a stepwise manner, starting with a simple qualitative assessment, followed by a semi-quantitaive muticriteria analysis (MCA) and a monetized cost-benefit assessment (CBA). The MCA approach was adopted using a spreadsheet tool (the DST). The benefits and impacts of undertaking LFM options were assessed based on the 15 SuRF-UK indicator categories (Table 4) and the relative importance of the five different indicator categories listed under each pillar of sustainability were weighted based on own judgement. Care was taken to ensure the total weights applied across the indicators under each of the environmental, social, and economic headings were equal, such that there was a balanced appraisal of the environmental, social, and economic factors.

### *2.3.1 Environmental Assessment*

The assessment criteria used for the environmental indicators are summarised in Table 5.

Due to the difficulty to find quantitative information on the environmental impacts of the selected technologies, these were scored by comparing them with each other rather than by giving them absolute values. Thus, -3 was assigned to the technology with the highest positive impact and +3 was assigned to the technology with the highest negative impact (Table 6). Each technology was also compared to a do-nothing scenario and to each other (pairwise comparison approach).

The given score captures the impact of the technology in the worst possible case. For example, when assessing the impact of soil flushing on water contamination, Sapsford et al. (2016) mentioned an environmental concern of this technique regarding the fate of the extractant, being able to contaminate groundwater in case of poor conditions or management. In that regard, to capture the risk of pollution of groundwater, and as the impact of soil flushing on water contamination is believed to be the worst of all technologies, a score of +3 was given for soil flushing on the water contamination indicator. However, if the landfill has a liner, the risk of water contamination is lower, so the score should be reduced. Therefore, the tool applies a correction factor to take into account action or technology that can mitigate the negative impacts of landfill mining. The indicators that have a correction factor include GHG, NOx, SOx and water contamination. The correction factor on GHG and NOx-SOx describes the influence of the distance of transportation on the impacts of the scenario and was set as follows: 0.8 if less than 10 km; (0.005 x distance km) + 0.75 if between 10 and 50 km and 1 if over 50 km.

**TABLE 5:** Assessment criteria for the environmental impacts.

| Assessment criteria for environmental Impact | | Definition |
|---|---|---|
| Air | GHG | Release of greenhouse gas emissions; closely linked with energy consumption |
| | PM | Production and release of particulate matter into the air |
| | Odour | Production of odour |
| | NOx SOx | Production and release of nitrous and sulphurous oxides into the air |
| | VOCs | Production and release of volatile organic compounds into the atmosphere |
| Water | Water Contamination | Impact on contamination levels in water |
| Soil | Soil Contamination | Impact on contamination levels in soil |
| Ecology | Biota | Intrusion (e.g. light level changes, landscape changes, visual changes) on surrounding biota |
| | Noise | Amount of noise generated |
| Resource Use and Waste Generation | Waste Production | Amount of waste produced |
| | Metal Recovery | Amount of metal recovered |
| | Combustible Recovery | Amount of RDF recovered |

**TABLE 6:** Scoring scale.

| Score | Definition |
|---|---|
| 3 | High negative impact |
| 2 | Moderate negative impact |
| 1 | Low negative impact |
| 0 | No impact |
| -1 | Low positive impact |
| -2 | Moderate positive impact |
| -3 | High positive impact |

The water contamination correction factor depends on the presence or not of a membrane liner in the landfill; if the user selects "Yes", then the correction factor applied is 0.2, otherwise it is 1. It should be noted that the lack of information supporting the sensitivity of impact with the changes of input data prevented a fine definition of the correction factors. However, to provide user with a starting point, the use of these correction factors based on own judgment are provided to describe the reduction of impact as realistically as practicable. The performance of the 9 scenarios and their 3 options are calculated by adding the scores of the technologies they involved. As scenario 4 and 5 involve 10 technologies, the scale of the performance for each indicator ranges from -30 to +30. A score of -30 represents the highest beneficial impact on the indicator in comparison with the "do-nothing" scenario, while a score of +30 represents the highest negative impact.

### *2.3.2 Economic Assessment*

Due to the difficulty in obtaining current estimates of costs for landfill mining from published case studies, some assumptions were made when choosing the economic indicators (Table 7). The costs and efficiencies of each technology were determined from a literature review (Sapsford et al., 2016, Danthurebandra et al., 2015, Ford et al., 2013, Wolfsberger et al., 2016) and were multiplied by the amount of input waste to calculate the amount of waste processed by each technology. Table 8 provides an overview of the capital cost and operating costs considered.

The efficiency of each technology considered is summarised in Table 9. The DST was developed to calculate an estimate of the revenue from mining a landfill, by taking into consideration the revenues produced by the sales of the recovered metals and the produced electricity, as well as the use of the remediated land. The prices for the recovery of materials and electricity considered in the DST are summarised in Table 10.

All revenues were summed to calculate the total revenue. The revenues were calculated as follows:

$$\mathrm{RMR} = \text{revenue from ferrous metal} + \text{revenue from non-ferrous metal}$$

where: RMR is revenue from material recovery (WtM)
Revenue from ferrous metal = amount (tonne) * price (€/tonne) and revenue from non-ferrous metal = amount (tonne) * price (€/tonne). Prices obtained from Letsrecycle.com, 2017.

**TABLE 7:** Economic indicators and their associated assumptions.

| Indicator | Definition | Assumptions |
|---|---|---|
| Net Income | Difference between revenues and costs | - |
| Revenue | Income from sale of recovered materials, sale of produced electricity and sale of reclaimed land | Heavy and hazardous fraction resulting from sorting and ATT have a net income of zero. Adjusted for inflation using inflation rates from 2005-December 2016 (OECD, 2017; Eurostat, 2017). All obtained currencies converted to Euros (€). Conversion rate for GBP (£) to Euros (€) and US Dollars ($) to Euros (€) 1.17 and 0.9416 respectively |
| Costs | Operational and capital costs of soil flushing, excavation, separation, sorting techniques and ATT. Also considers transportation costs | Adjusted for inflation using inflation rates from 2005-December 2016 (OECD, 2017; Eurostat, 2017). All obtained currencies converted to Euros (€). Conversion rate for GBP (£) to Euros (€) and US Dollars ($) to Euros (€) 1.17 and 0.9416 respectively |

**TABLE 8:** Costs estimates used by the DST.

| Technology | Capital cost (€/item) | Low operating costs (€/tonne) | High operating costs (€/tonne) | Reference |
|---|---|---|---|---|
| Soil flushing | - | 10 | 228 | Sapsford et al (2016) |
| Excavation | 2.10 €/tonne | - | 3.94 | Danthurebandra et al 2015) |
| Visual separation | - | - | 0.8 | Ford (2013) |
| Ballistic separation | 150,000 | - | 6.80 | Wolfsberger 2016 |
| Screening | 200,000 | - | 2.91 | Ford (2013) |
| Shredding | 325,000 | - | 9.71 | Ford (2013) |
| Air separation | 292,500 | - | 14.56 | Ford (2013) |
| Ferrous metal separation | 45,000 | - | 2.91 | Ford (2013) |
| Non-ferrous metal separation | 65,000 | - | 5.83 | Ford (2013) |
| Transportation | - | - | 0.2 | Schade et al., (2006) |
| ATT (gasification) | 50 €/tonne | - | 67 | Danthurebandra et al (2015) |

**TABLE 9:** Efficiency used for the DST for each technology considered.

| % of material that goes to following processes | <10mm soil type | Plastic | Paper/Card-board | Wood | Textile | Glass | Stones, inert | Ferrous metals | Non-ferrous metals | Hazard-ous | Organic waste | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Excavation | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | |
| Ballistic separation Heavy fraction | 5% | 50% | 20% | 60% | 20% | 50 | 35% | 95% | 75% | 80% | 50% | Wolfsberger et al (2016) |
| Fine fraction | 90% | 10% | 10% | 20% | 5% | 50% | 60% | 3% | 20% | 15% | 50% | Wolfsberger et al (2016) |
| Calorific fraction | 5% | 40% | 70% | 20% | 75% | 50% | 5% | 2% | 5% | 5% | 0% | Wolfsberger et al (2016) |
| Screening | 10% | 80% | 70% | 75% | 90% | 10% | 90% | 50% | 40% | 15% | 5% | Wolfsberger et al (2016) |
| Shredding | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | Assumption |
| Air separation | 1% | 99% | 99% | 1% | 99% | 1% | 1% | 1% | 1% | 1% | 1% | http://www.ni-hot.co.uk/prod-ucts/drum-sepa-rators/ |
| Ferrous metal separation | 98% | 95% | 100% | 98% | 99% | 100% | 98% | 20% | 100% | 00% | 100% | Wolfsberger et al (2016) |
| Non-ferrous metal separation | 97% | 98% | 98% | 98% | 99% | 100% | 100% | 97% | 20% | 100% | 100% | Wolfsberger et al (2016) |
| Fines Ferrous metal separation | 98% | 95% | 100% | 98% | 99% | 100% | 98% | 20% | 100% | 100% | 100% | Assumption |
| Soil flushing | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 70% | 70% | 100% | 100% | Assumption |

**TABLE 10:** Revenues from metals and energy recovery and from land reclamation.

| Revenue | Worst case | Best case | References |
|---|---|---|---|
| Electricity production | 80 €/MWh | 135 €/MWh | http://ec.europa.eu/eurostat/statistics-explained/index.php/File:Electricity_prices_for_household_consumers,_second_half_2015 |
| Ferrous metal recovery | 130 €/ton | 142 €/ton | Prices of scraps www.letsrecycle.com/prices/metals |
| Non-ferrous metal recovery | 1827 €/ton | 1913 €/ton | Prices of scraps www.letsrecycle.com/prices/metals |
| Land reclamation:<br>Residential<br>Industrial<br>Agricultural<br>Nature | <br>155 €<br>80 €<br>10 €<br>3 € | | Danthurebandara et. al (2015) |

$$\mathrm{RER} = \text{amount of RDF (tonne)} * \text{calorific value (MJ/tonne)} * \text{ATT efficiency} * \text{conversion factor (MWh/MJ)} * \text{price of electricity (€/MWh)}$$

where: RER is revenues from energy recovery (WtE)
Prices obtained from http://ec.europa.eu/eurostat/statistics-explained/index.php/.

$$\mathrm{RLR} = \text{land area (ha)} * \text{land value (€/ha)}$$

where: RLR is revenues from land reclamation. Land values obtained from Danthurebandara et al., 2015 for 4 different uses; residential, industrial, agricultural and nature. A land use for more landfill space was given a value of 0 €/ha, but can be changed by the users to consider avoided costs.

*2.3.3 Social Assessment*

The social indicators used were adapted from the SURF UK indicator set for sustainable remediation assessment (CLAIRE, 2011) and summarised in Table 4. A brief explanation of how they were considered in the tool is provided below.

- Community Involvement: measures the community involvement and acceptance of the project. This involvement depends on the consequences created that directly affect their life.
- Human Health: measures the impacts on the health of the site-workers and community members caused by the incidence of VOCs, noise, odour, dust and bioaerosols.
- Ethical considerations: measures the possibility of creating ethical disputes. For example, groundwater gets contaminated and a population is served from this source.
- Nuisance on neighbourhoods: measures the occurrence of nuisance factors (e.g. noise, light pollution, smells, litter and debris off site).
- Evidence of Sustainability and Level of Uncertainty:

measures the degree of environmental sustainability, as well as the levels of uncertainty related to the outcomes.

The social performance of the scenarios is calculated the same way as the environmental performance described previously. A correction factor was also applied for the human health and nuisance on neighbourhood indicators to take into account the influence of the number of close residents. Given the absence of supporting data, the correction factor were developed based on own judgement (Table 11).

### 2.4 Calculation of REE

REE estimates were calculated using a linear regression between percentage MSW and amount of REE. The data were taken from Morf et al. (2013) and Gutierrez-Gutierrez et al., (2015) (Table 12). REE value was calculated by multiplying the amount of REE with the market value from January 2017 to March 2017 (London Metal Exchange, 2017; Metalary, 2017).

## 3. MODEL OUTPUTS AND USER INTERFACE

The DST was created in Microsoft Excel. The following sections will explain the tool interface and outputs of the model.

### 3.1 Scenario Inputs

The scenario input tab is displayed in Figure 2. This tab is used to enter the input parameters. Users have the option to select either default values for waste composition, value of the remediated land or enter their own values (custom composition). The user can also select the best scenario calculation according to either best financial, environment or social outcomes.

### 3.2 Best Scenario Results and Scenario Comparison

The best scenario results tab displays the economic, environmental and social assessment results for the best scenario according to the user's criteria for selection (Figure 3). For the environmental and social assessments, the impact indicators are also displayed as a bar chart, where the baseline is the do-nothing scenario (Figure 4).

The scenario comparison tab compares the performance of all 9 ELFM scenarios against each other. This is displayed visually with radar charts, created by fixing the worst performer in each category (economic, environmental and social) to zero and the best performer in each category to 100; each scenario is placed along this scale of 0 to 100 (worst to best) by regressing the scale against actual values. Illustrative example is shown in Figure 5.

**TABLE 11:** Correction factor values used for the human health and nuisance on neighbourhood.

| Number of residents living at less than 1km from the boundaries | Correction factor values |
|---|---|
| < 200 | 0.1 |
| 200 - 400 | 0.2 |
| 400 - 600 | 0.4 |
| 600 - 800 | 0.6 |
| 800 - 1000 | 0.8 |
| > 1000 | 1 |

**TABLE 12:** Estimates of REE amount in landfill sites.

| | Concentration mg/kg | Gutierrez et al (2015) 75% MSW 25% C&I | Morf et al (2013) 57,2% MSW 42,8% C&I |
|---|---|---|---|
| REEs | Sc | 3.46 | 0.96 |
| | Y | 6.42 | 7.85 |
| | La | 9.36 | 9 |
| | Ce | 21.38 | 20.5 |
| | Pr | 2.39 | 1.9 |
| | Nd | 11.75 | 7.26 |
| | Sm | 2.06 | |
| | Eu | 0.59 | |
| | Gd | 2.07 | 0.75 |
| | Tb | 0.24 | |
| | Dy | 1.44 | |
| | Ho | 0.21 | |
| | Er | 0.65 | |
| | Tm | 0.08 | |
| | Yb | 0.52 | |
| | Lu | 0.07 | |
| PGMs | Pt | 0.02 | 0.059 |
| | Pd | 0.77 | 0.5 |
| | Ru | 21.90 | 0.0005 |
| Other critical | Li | 0.10 | 9 |
| | In | 7.71 | 0.29 |
| | Sb | 14.14 | |
| | Co | 1076.00 | |
| Others | Cu | 1076 | 2230 |
| | Ag | 2.26 | 5.3 |
| | Au | 0.18 | 0.4 |
| | Al | 17274 | 17000 |

## 4. MODEL TESTING AND VALIDATION

In order to test the tool and to further understand which factors affect landfill mining feasibility, 10 scenarios were simulated. The characteristics of each scenario are shown in Table 13. In all 10 cases, the same amount of waste and size of landfill were used. Five variations of the waste composition were considered to understand which materials make landfill mining more profitable. In addition, with each type of waste composition, parameters such as the presence of a geomembrane, number of residents and the distances to the energy and sorting facilities were varied in order to understand the variations of the social and environmental indicators. Given that the only parameter affecting the amount of REEs in the tool is the percentage of MSW, REEs calculation was only done for cases 1, 3, 5, 7 and 9. In this illustrative example the criteria for selection of the best scenario was "the highest best case net income".

The economic results are shown in Table 14. For all the different types of landfills used, the best suggested process approach is Scenario 8. This scenario allows the highest net income due to the high efficiency (99%) of the air separation process. This efficiency increases the amount of plastics, textiles and wood sorted, thus increasing the

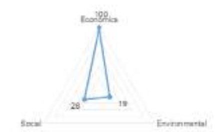
| Input | Default value | Custom value | Units |
|---|---|---|---|
| Total amount of waste in landfill: | 100,000 | | tonnes |
| Total landfill area: | 10 | | ha |
| **Waste Composition (% of weight) :** | **100% MSW** | **Custom composition** | |
| <10mm soil type | 45.89% | | w/w |
| Plastic | 17.22% | | w/w |
| Paper/Cardboard | 7.13% | | w/w |
| Wood | 7.38% | | w/w |
| Textile | 3.97% | | w/w |
| Glass | 1.59% | | w/w |
| Stones, inert | 10.38% | | w/w |
| Ferrous metals | 2.76% | | w/w |
| Non-ferrous metals (REEs) | 0.79% | | w/w |
| Hazardous | 0.20% | | w/w |
| Organic waste | 2.68% | | w/w |
| Current sum : | | 0% | |
| Is there a liner in the landfill? | No | | |
| Number of residents within 1 km radius | > 1000 | | PE |
| **Value of remediated land:** | | | |
| Residential | 155 | | €/m² |
| Industrial | 80 | | €/m² |
| Agricultural | 10 | | €/m² |
| Nature | 3 | | €/m² |
| Landfill | 0 | | €/m² |
| Distance from the landfill to Waste treatment facility | 50 | | km |
| Distance from waste treatment facility to waste to en | 80 | | km |
| Distance from the landfill to Waste to Energy plant (g | 10 | | km |
| Criteria for selection of the best scenario | The highest best case net income | | |

Show me the results !

**FIGURE 2:** Scenario inputs display.

ECONOMIC ASSESSMENT

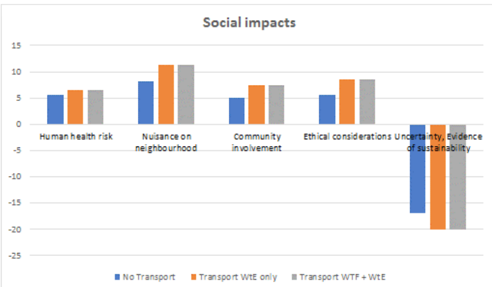
| NET INCOME | Worst case | Best case | Unit |
|---|---|---|---|
| | -3 778 824 | 17 212 579 | € |




| REVENUES | Worst case | Best case | Unit |
|---|---|---|---|
| | 3 262 716 | 20 626 730 | € |
| **Revenue from WtM** | 602 092 | 636 927 | € |
| Amount of ferrous metals | 1 094 | 1 094 | t |
| Amount of non-ferrous metals | 252 | 252 | t |
| Revenue ferrous | 142 353 | 155 372 | € |
| Revenue non-ferrous | 459 739 | 481 555 | € |
| **Revenue from WtE** | 2 660 624 | 4 489 808 | € |
| Amount of RDF | 22 172 | 22 172 | t |
| **Revenue from land** | | | |
| Residential | 15 500 000 | 15 500 000 | € |
| Industrial | 8 000 000 | 8 000 000 | € |
| Agricultural | 1 000 000 | 1 000 000 | € |
| Nature | 300 000 | 300 000 | € |
| Landfill | 0 | 0 | € |

| COSTS | Worst case | Best case | Unit |
|---|---|---|---|
| | 7 041 540 | 3 414 151 | € |
| OPEX excavation & sorting | 1 928 635 | 1 928 635 | € |
| OPEX WtE | 1 485 515 | 1 485 515 | € |
| OPEX Transport WtE only | 45 180 | | € |
| OPEX Transport WTF + WtE | 1 373 032 | | € |
| CAPEX excavation & sorting | 1 137 788 | | € |
| CAPEX WtE | 1 116 569 | | € |

**FIGURE 3:** Best scenario results.



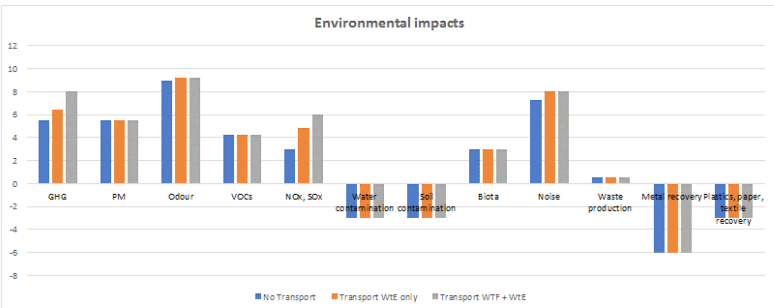




**FIGURE 4:** Bar charts showing social and environmental impacts.

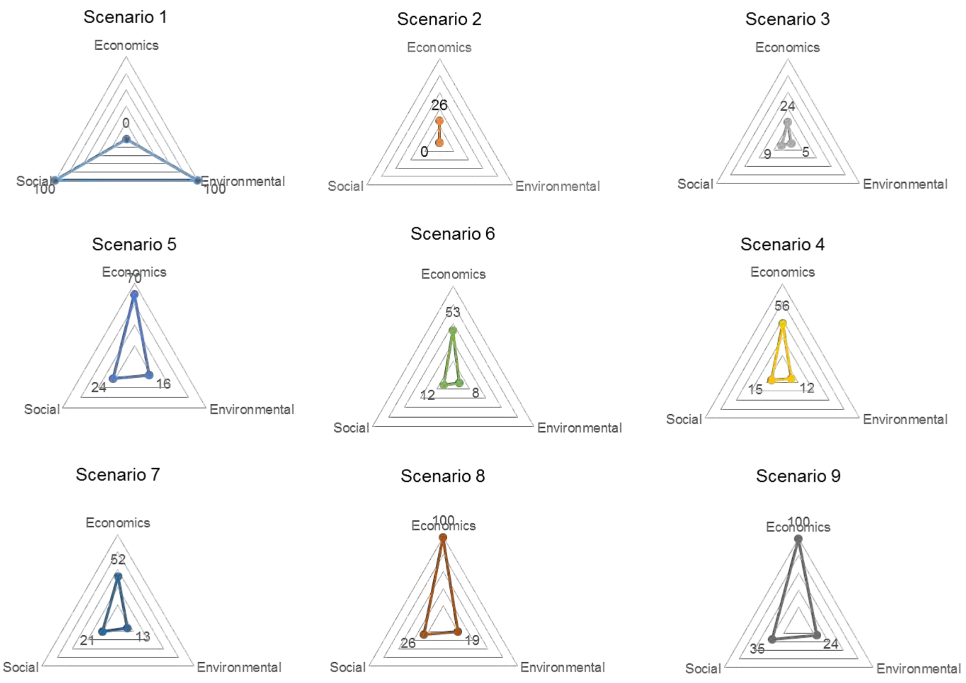



**FIGURE 5:** Radar charts for scenario comparison.

amount of electricity produced and sold. Even though the recovery of ferrous metals from the fines increases the total recovered amount of these materials, the extra cost of this process outweighs its revenues, thus decreasing the net income of the whole mining process. From these simulations, it can be seen that the higher the percentage of MSW in the landfill, the higher the net income. This is due to the fact that the higher the percentage of MSW, the higher the amount of non-ferrous metals, paper/cardboard and textile.

Social impacts such as human health risk and nuisance on neighbourhood increase with the number of people living in the areas surrounding the landfill. The presence of a geomembrane in the landfill increases the positive impact of landfill mining on the water contamination indicator. Also, the use of transportation increases the negative impacts on GHG emissions, SOx and NOx. For all the evaluated cases, the potential revenues from REEs are high, passing the 1.2 billion € threshold (data not shown). The elements with the highest values are: Sc, Pd, Au, Al and Cu. Even though the amount of Sc, Pd and Al is reduced as the percentage of MSW decreases, the quantity of Ag, Au, Cu and other elements increase, thus maintaining the high revenues. The high variations in the amount of REEs present in the different landfills can also be attributed to the fact that the correlation between REEs and percentage of MSW was done considering only two landfill sites.

To validate the model and the DST outputs, the DST was run for the REMO landfill as a case study. Input data

**TABLE 13:** Overview of the scenarios considered.

| Case | Waste composition | Is there a liner in the landfill? | Number of residents within 1 km radius (PE) | Distance from the landfill to Waste treatment facility (km) | Distance from waste treatment facility to waste to energy plant (km) | Distance from the landfill to Waste to Energy plant (gasifier) (km) | Criteria for selection of the best scenario |
|---|---|---|---|---|---|---|---|
| 1 | 100% MSW | Yes | 0 | 0 | 0 | 0 | Best net Income |
| 2 | 100% MSW | No | 1000 | 10 | 10 | 0 | Best net Income |
| 3 | 75% MSW 25% C&I | Yes | 0 | 0 | 0 | 0 | Best net Income |
| 4 | 75% MSW 25% C&I | No | 1000 | 10 | 10 | 0 | Best net Income |
| 5 | 50% MSW 50% C&I | Yes | 0 | 0 | 0 | 0 | Best net Income |
| 6 | 50% MSW 50% C&I | No | 1000 | 10 | 10 | 0 | Best net Income |
| 7 | 25% MSW 75% C&I | Yes | 0 | 0 | 0 | 0 | Best net Income |
| 8 | 25% MSW 75% C&I | No | 1000 | 10 | 10 | 0 | Best net Income |
| 9 | 100% C&I | Yes | 0 | 0 | 0 | 0 | Best net Income |
| 10 | 100% C&I | No | 1000 | 10 | 10 | 0 | Best net Income |

**TABLE 14:** Overview of the operating and capital costs for the 10 scenarios.

| Scenarios | | Costs (€) | | OPEX excavation & sorting (€) | | OPEX WtE (€) | | OPEX Transport WTF + WtE (€) | | CAPEX excavation & sorting (€) | | CAPEX WtE (€) | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Case | Waste composition | Worst scenario | Best scenario | Worst scenario | Best scenario | Worst scenario | Best scenario | Worst scenario | Best scenario | Worst scenario | Best scenario | Worst scenario | Best scenario |
| 1 | 100% MSW | 716,819,611 | 515,536,732 | 291,223,938 | 291,223,938 | 224,312,794 | 224,312,794 | - | - | 32,680,988 | - | 168,601,891 | - |
| 2 | 100% MSW | 754,191,849 | 515,536,732 | 291,223,938 | 291,223,938 | 224,312,794 | 224,312,794 | 37,044,569 | - | 32,387,030 | - | 168,563,393 | |
| 3 | 75% MSW & 25% I&W | 679,076,429 | 491,499,857 | 285,573,737 | 285,573,737 | 205,926,120 | 205,926,120 | - | - | 32,680,988 | - | 154,895,584 | - |
| 4 | 75% MSW & 25% I&W | 715,894,061 | 491,499,857 | 285,573,737 | 285,573,737 | 205,926,120 | 205,926,120 | 36,817,632 | - | 32,680,988 | - | 154,895,584 | - |
| 5 | 50% MSW & 50% I&W | 641,333,247 | 467,462,983 | 279,923,536 | 279,923,536 | 187,539,446 | 187,539,446 | - | - | 32,680,988 | - | 141,189,276 | - |
| 6 | 50% MSW & 50% I&W | 677,596,273 | 467,462,983 | 279,923,536 | 279,923,536 | 187,539,446 | 187,539,446 | 36,263,026 | - | 32,680,988 | - | 141,189,276 | - |
| 7 | 25% MSW & 75% I&W | 603,590,065 | 443,426,108 | 274,273,336 | 274,273,336 | 169,152,772 | 169,152,772 | - | - | 32,680,988 | - | 127,482,969 | - |
| 8 | 25% MSW & 75% I&W | 639,298,485 | 443,426,108 | 274,273,336 | 274,273,336 | 169,152,772 | 169,152,772 | 35,708,419 | - | 32,680,988 | - | 127,482,969 | - |
| 9 | 100% I&W | 565,846,883 | 419,389,233 | 268,623,135 | 268,623,135 | 150,766,098 | 150,766,098 | - | - | 32,680,988 | - | 113,776,662 | - |
| 10 | 100% I&W | 601,000,696 | 419,389,233 | 268,623,135 | 268,623,135 | 150,766,098 | 150,766,098 | 35,153,813 | - | 32,680,988 | - | 113,776,662 | - |

was taken from (Van Passel et al., 2013) and a 50% MSW, 50% C&I waste scenario was used, congruent with waste sources described in the literature. The DST economic outputs are displayed in Table 15 alongside the economic assessment results from Van Passel et al. (2013).

While net income, WtE revenues and excavation, sorting and pre-treatment costs show little variance between the DST outputs and those published, values for WtM revenues and incineration costs vary greatly. This does not invalidate the DST outputs, but may be a product of model assumptions; for example (Van Passel et al., 2013) considered a scenario where a high capacity ATT plant is built, whereas the DST ATT plant is flexible in capacity depending on the amount of waste to be processed. In addition, some differences are likely to be a result of the economic models used; (Van Passel et al., 2013) used the more complex Net Present Value (NPV) model which considers monetary value change over time whereas the DST does not account for this.

## 5. CAUTIONARY NOTES

The DST provides a framework for the assessment of landfill mining projects. The DST is based on values taken from literature which need to be updated over time as new data will become available to better reflect landfill waste composition processes. No formal sensitivity analysis has been performed to test for interactions among the indicators due to a lack of field-scale data available whilst existing influences are present and are mentioned previously. Therefore, caution must be taken with different model settings as the results may not directly be comparable and recommendations for landfill mining scenarios are not necessarily supported by the authors. Several limitations such as error margin on the waste composition, technology process efficiency, technology cost, land value, weighting approach used which is based on a small panel of two academics and two professionals should be consider by the user when reviewing the tool results. Conservative estimates must be used in order to not overestimate or underestimate the sustainability criteria.

## 6. CONCLUSIONS

The DST is able to predict the economic, environmental and social outcomes of a ELFM project, along with an estimate of the amount of REE present and the best ELFM process and technology train to use. The DST has been designed to allow user input parameters, or for the user to select input parameters provided by the tool from literature review. The user is able to select the criteria for the best scenario and compare across different ELFM process scenarios. Overall, the DST is the first of its kind and acts as an initial assessment tool prior to more complex assessments and modelling. The DST will facilitate the uptake of ELFM practices by providing a feasibility assessment that is user-friendly for site operators.

The model validation shows that the DST modelling is congruent with literature in some economic outputs and differs in others. From this, we recommend future research into designing a similar DST that incorporates more complex models that account for value changes over time, such as NPV and full Life Cycle Analysis (LCA). In addition, parameters that are time dependent, such as market values, could, in future, be linked to continually updated databases to improve validity. Overall, the DST is an innovative and progressive tool that is successful in serving as a starting point for site operators to assess the feasibility of a proposed ELFM. The DST aids in reducing the uncertainty regarding the economic feasibility and social and environmental consequences of ELFM and will therefore encourage the uptake of ELFM projects.

**TABLE 15:** REMO simulation results.

| Output | DST Value (€) | Van Passel et al. (2013) Value (€) | Variation |
|---|---|---|---|
| Net Income | 2,123,650 | 1,933,825 | 10% |
| WtM Revenues | 2,637,355 | 736,003 | 258% |
| WtE Revenues | 8,785,195 | 6,831,834 | 29% |
| Excavation, Sorting and Pre-treatment Costs | 2,875,956 | 3,373,913 | -15% |
| Incineration Costs | 6,304,862 | 2,260,409 | 179% |

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# RECYCLING OF WASTE PLASTICS DISPOSED OF IN LANDFILLS: THE EFFECT OF WASHING TREATMENT

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**ABSTRACT**

In recent years, environmental pollution in the vicinity of final disposal sites for inert waste has resulted in problems such as deterioration of water quality and increases in leachates and odours. Accordingly, the number of sites that no longer accept these wastes has risen. One of the main causes of these problems is represented by organic matter adhering to waste plastics, accounting for the majority of waste deposited at final disposal sites for inert waste. Furthermore, the amount of waste plastics undergoing thermal recycling has increased owing to recent energy demands. This study examines the recycling of waste plastics into solid fuel, known as refuse paper, and plastic fuel (RPF), by focusing on the washing method. The first washing effect at liquid to solid ratio (L/S) = 1 was confirmed to be effective in the rough removal of adherent organic matter and was also found to be suitable for pretreatment for the second washing for thermal recycling. As compared with the first washing, although effectiveness of the second washing with regard to chemical oxygen demand (COD) was not observed, the effectiveness of the second washing versus total nitrogen (T-N) at low speed and L/S = 1 was comparable to that of the first washing at L/S = 10. These results indicate that a combination of first washing at L/S = 1 and second washing at low speed and L/S = 5 constituted an effective pretreatment for refuse paper and plastic fuel (RPF) production. Furthermore, a mixed antichlor was produced by combining a commercial antichlor and waste having a dechlorination effect as countermeasures against hydrogen chloride gas emitted when RPF derived from the burning of waste plastics. We tested its performance in reducing Cl gas generation by mixing it into the RPF. As the antichlor was added to RPF for low Cl gas emission, 36% of the CaO–oyster shells added as antichlor against Cl content was found to be effective in reducing Cl gas emission. Thus, it was concluded that using an appropriate percentage of antichlor, the combination of first and second washing processes (under some conditions) was effective in the pretreatment of waste plastic destined to be recycled as RPF.

## 1. INTRODUCTION

In recent years, environmental pollution in the vicinity of final disposal sites for inert waste has resulted in problems such as deterioration of water quality and increases in leachates and odours. Accordingly, the number of sites that no longer accept these wastes has risen. Although organic matter adhering to waste plastics, accounting for the majority of waste deposited at final disposal sites for inert wastes, is considered to be a cause of these problems, the amount of incoming wastes with these characteristics is controlled simply by visual inspection, as required by law (Act on the Technology Standards for final disposal of municipal solid waste and industrial solid waste, (article 2.2). Following implementation of the revised Waste Management Act, enforced on June 17, 1998, the installation of new facilities including final disposal sites for inert waste has advanced smoothly due to the prescribing of structural reinforcement and a complicated application procedure. Therefore, the number of newly installed final disposal sites for inert wastes has decreased since 1999 (Ministry of the Environment, 1998-2012). Furthermore, the

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amount of waste plastics recycled increased by 2.3 × 105 tons from FY2011 to FY2013. As a result, the amount finally disposed of decreased by about 3.1 × 105 tons from 10.5 × 105 tons in FY2011 to 7.4 × 105 tons in FY2013. Of the many recycling categories, only thermal recycling, including refuse-derived fuel, raw material for cement production, and fuel used in cement kilns, increased from FY2011 to FY2013, at 5.3 × 105 tons (Plastic Waste Management Institute, 2012, 2015). The use of waste plastics as an alternative fuel has therefore increased in the light of recent energy demands. Moreover, prior to enforcement of the Law for the Promotion of Sorted Collection and Recycling of Containers and Packaging in April 2000, good-quality waste plastics, which should be recycled, were deposited in final disposal sites for inert wastes. These good quality waste plastics are considered to be suitable candidates for thermal recycling or material recycling, deemed to be important resource repositories for resource-poor Japan. However, currently enforced laws and the manifest system make it impossible to exhume inert wastes and to recover waste plastics for recycling from the final disposal sites. Therefore, an amendment to these laws to exhume inert wastes and to recover waste plastics for recycling from the final disposal sites is mandatory.

EU laws establish that recycling of all waste plastics by the year 2030, with residues being landfilled directly at the final disposal site. The current state of the art for recycling of waste plastics in Asia is still significantly lacking.

This paper examines the recycling of waste plastics deposited in final disposal sites for inert wastes prior to April 2000 and waste plastics disposed of in landfills. Waste plastics that are not recycled but disposed of in landfills may result in foul odours during transportation and storage due to the presence of adherent organic matter. Similar odours also occur at final disposal sites for inert wastes. Focusing on an effective washing method as a pretreatment technique for incineration residue (Tameda et al., 2007), we developed a technique for use in the efficient recycling of waste plastics. Our washing method includes two steps. The first washing relates to on-site agitation in water tanks to remove adherent organic matter and resolve the issue of odours during transportation and storage at final disposal sites for inert wastes. The second washing is performed by means of high-speed rotation to thoroughly remove adherent organic matter and enable the use of waste plastics for thermal recycling.

In the recycling of waste plastics, a solid fuel known as refuse paper and plastic fuel (RPF) was used, thus facilitating its subsequent use as an alternative fuel. It should be noted that hydrogen chloride (HCl) gas is emitted when RPF is burned. The content of Cl in RPF, a cause of gas emission, is regulated by Japanese Industrial Standard (JIS) Z7311:2010 as a mass fraction of all Cl content. However, some types of RPF do not meet this standard, and countermeasures against Cl gas are required. A mixed antichlor was therefore produced by combining a commercial antichlor and waste having a dechlorination effect, and its performance in reducing Cl gas generation was tested by mixing it into the RPF.

## 2. MATERIALS AND METHODS

### 2.1 Experiment on the effect of first washing to roughly remove adherent organic matter

We conducted an experiment on the effects of the first washing to roughly remove adherent organic matter.

#### *2.1.1 Overview of the experiment*

For the first washing experiment, 40-foot containers were modified to allow agitation from above using a heavy machine (Figure 1). Two water tanks were installed, including the first and second washing tanks. 15 $m^3$ of water was poured into the first and second tanks, respectively, for a total of 30 $m^3$. The liquid–solid ratio (L/S) was determined as the volume of water against the volume of waste to be washed in a 1:1 ratio. For this experiment, waste containing waste plastics that would be deposited in final disposal sites for inert waste was used; 10 batches of 3 $m^3$ waste were continuously washed. The time required for washing once (at the first washing tank + the second washing tank) corresponded to approx. 30 minutes and it did not affect the subsequent capitalization process. To assess washing efficiency, waste plastics were extracted from the waste for analysis. We analysed the compositions of the waste to be washed and performed chemical oxygen demand (COD) and total nitrogen (T-N) on the waste plastics before and after washing (Table 1).

#### *2.1.2 Results and Discussion*

Ten cases in which inappropriate matter for disposal was contained in waste were selected by visual inspection and the compositions analysed on the basis of weight percentage; the results are shown in Table 2. Overall, waste plastics accounted for 63% on average, and inappropriate matter for disposal other than the five types of stable wastes such as waste plastics, rubber scrap, metal scrap, demolition wastes and waste glass and ceramics accounted for 14% on average. The details of Case 9, used for the first washing experiment, are shown in Figure 2. Waste plastics constituted the highest percentage at 35%; inappropriate matter was second at 31%; and residue of 75 mm or smaller were third, at 24%. Polyvinyl chloride (PVC), an undesirable compound in thermal recycling, was as low as 2%. Organic pollutants such as wood, paper, and food residues contained in the inappropriate matter, likewise undesirable for landfills, corresponded to 11.8%.

Figure 3 shows the results of a leachate test for the COD of waste plastics samples collected before and after the first washing experiment. The value obtained before first washing was 170 mg/L. After washing, values of 27 mg/L (L/S = 10; removal efficiency = 83%), 29 mg/L (L/S = 3; removal efficiency = 83%), 35 mg/L (L/S = 1.5; removal efficiency = 79%), and 39 mg/L (L/S = 1; removal efficiency = 77%) were obtained. These results indicate that although COD removal efficiency decreased as L/S decreased, adequate washing efficiency was obtained even at L/S = 1.

Figure 4 shows the results of a leachate test for T-N of waste plastics samples collected before and after the first washing experiment. The value obtained prior to testing was 16 mg/L. Subsequently, values of 2.1 mg/L (L/S

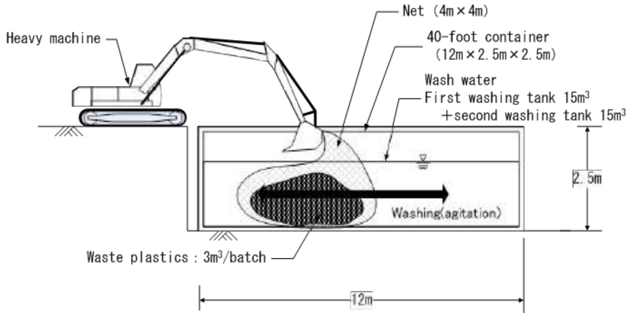


**FIGURE 1:** Outline of first washing procedure.

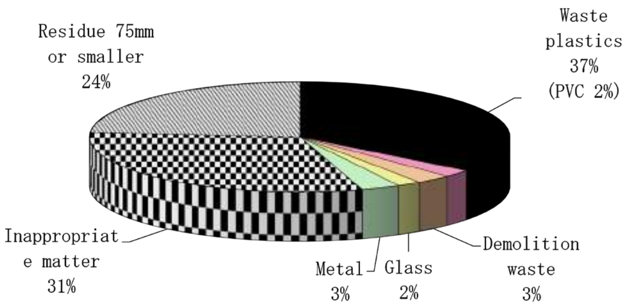


**FIGURE 2:** Detailed composition analysis of Case 9.

= 10; removal efficiency = 87%), 4 mg/L (L/S = 3; removal efficiency = 75%), 3.5 mg/L (L/S = 1.5; removal efficiency = 78%), and 3.6 mg/L (L/S = 1; removal efficiency = 78%) were obtained. These results indicate that although T-N removal efficiency decreased as L/S decreased, adequate washing efficiency was obtained even at L/S = 1. After washing ten times, the residual water COD-Mn was 120 mg/L and T-N was 19mg/L. Depending on its quality the residual water was discharged or recycled.

## 2.2 Experiment on the effect of second washing to produce RPF

The recycling of waste plastics into RPF was investigated. To further remove adherent organic matter on waste plastics roughly washed by first washing, we adopted a second washing process as pretreatment for RPF production and conducted an experiment on the effects of the second washing process.

**TABLE 1:** Plan of first washing experiment.

| | |
|---|---|
| Sampling interval of first washing tank | After 1st, 3rd, 5th, 7th and 10th washing |
| Sampling interval of second washing tank | After 1st, 3rd, 5th, 7th and 10th washing |
| Leachate test items for waste plastics before and after washing | COD-Mn T-N the standard methods = Ministry of the Environment Notification No. 46 |
| | The amount of the sample = solid (1): liquid (10) |
| | The number of replica = 3 replicas |
| Waste to be washed | Composition analysis |

**TABLE 2:** Results of composition analysis of waste to be washed.

| | Waste plastics | | Demolition waste | Ceramics waste | Metals | Inappropriate matter | Residues | | Total | Measured weight | A-B |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Soft | Non soft | | | | | (2 mm~ 75 mm) | (2 mm or smaller) | • • • A | • • • B | |
| Case 1 | 101.4<br>25.1% | 249.6<br>61.8% | 0.1<br>0.0% | - | 2.6<br>0.6% | 16.0<br>4.0% | 30.8<br>7.6% | 3.2<br>0.8% | 403.7<br>100.0% | 440 | -36.3<br>91.75% |
| Case 2 | 47.2<br>11.2% | 304.0<br>72.4% | - | - | 4.0<br>1.0% | 46.4<br>11.0% | 17.6<br>4.2% | 0.8<br>0.2% | 420.0<br>100.0% | 420 | 0.0<br>100.00% |
| Case 3 | 64.0<br>9.5% | 379.2<br>56.3% | 9.6<br>1.4% | - | 32.0<br>4.7% | 77.2<br>11.5% | 105.6<br>15.7% | 6.4<br>0.9% | 674.0<br>100.0% | 960 | -286.0<br>70.21% |
| Case 4 | 68.0<br>23.9% | 66.4<br>23.3% | 20.8<br>7.3% | - | 12.0<br>4.2% | 36.8<br>12.9% | 47.2<br>16.6% | 33.6<br>11.8% | 284.8<br>100.0% | 290 | -5.2<br>98.21% |
| Case 5 | 26.4<br>15.0% | 62.0<br>35.2% | 0.8<br>0.5% | - | 1.2<br>0.7% | 44.4<br>25.2% | 36.8<br>20.9% | 4.4<br>2.5% | 176.0<br>100.0% | 190 | -14.0<br>92.63% |
| Case 6 | 36.8<br>7.2% | 218.4<br>42.9% | 19.2<br>3.8% | 15.2<br>3.0% | 23.2<br>4.6% | 124.0<br>24.3% | 64.8<br>12.7% | 8.0<br>1.6% | 509.6<br>100.0% | 620 | -110.4<br>82.19% |
| Case 7 | 222.4<br>20.7% | 700.8<br>65.4% | 12.8<br>1.2% | - | 30.4<br>2.8% | 25.6<br>2.4% | 75.2<br>7.0% | 4.8<br>0.4% | 1,072.0<br>100.0% | 1290 | -218.0<br>83.10% |
| Case 8 | 101.6<br>10.9% | 357.6<br>38.5% | 106.4<br>11.5% | 35.2<br>3.8% | 32.0<br>3.4% | 52.4<br>5.6% | 226.4<br>24.4% | 17.6<br>1.9% | 929.2<br>100.0% | 830 | 99.2<br>111.95% |
| Case 9 | 59.2<br>6.7% | 272.8<br>31.0% | 24.0<br>2.7% | 16.0<br>1.8% | 26.4<br>3.0% | 273.6<br>31.1% | 168.0<br>19.1% | 40.8<br>4.6% | 880.8<br>100.0% | 1030 | -149.2<br>85.51% |
| Case 10 | 100.8<br>17.6% | 326.4<br>56.9% | 4.0<br>0.7% | - | 11.2<br>2.0% | 50.4<br>8.8% | 79.2<br>13.8% | 1.6<br>0.3% | 573.6<br>100.0% | 610 | -36.4<br>94.03% |

*Unit: Kg*

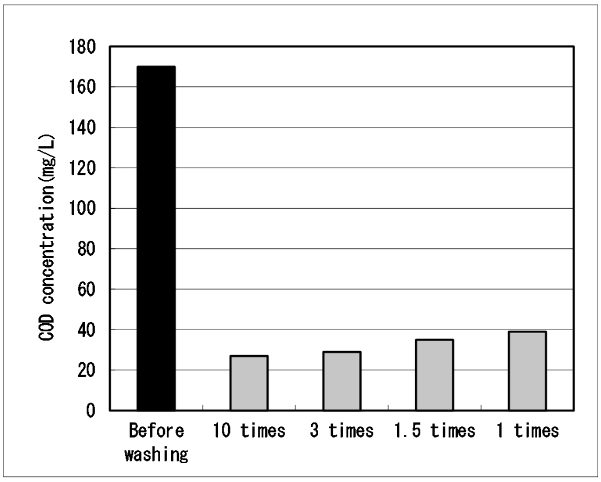


**FIGURE 3:** Results of chemical oxygen demand (COD) - leachate test before and after washing.

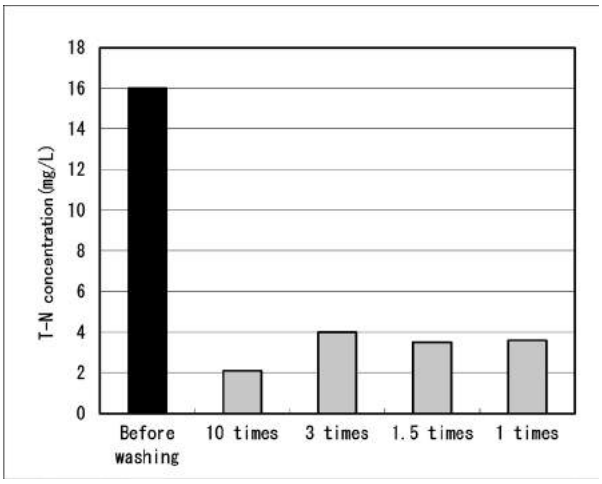


**FIGURE 4:** Results of total nitrogen (T-N) - leachate test before and after washing.

*2.2.1 Overview of the experiment*

For the second washing experiment, a high-speed rotary washer (Photo 1) was used and performed leachate tests of waste plastics collected before and after the second washing to check the effectiveness of the second washing process under several washing conditions. The washing conditions used in this experiment are shown in Table 3. COD value before the first washing was 170 mg/L. After washing, values of 39 mg/L (L/S = 1; removal efficiency = 77%) were obtained. Thus, to check the effects of removal in second washing process, organic matter adhering to waste plastics that were inappropriate for disposal in landfills and were unwashed before the second washing experiment were used as samples.

*2.2.2 Results and Discussion*

Table 4 shows the removal efficiency of biochemical oxygen demand (BOD), COD, and T-N for waste plastics after the second washing. The influence of L/S on BOD removal efficiency was inconsistent. Although BOD removal efficiency tended to be higher as L/S became high at low speeds, it increased in the order of L/S = 3, 1, and 5 at medium speed and L/S = 5, 3, and 1 at high speed. The influence of number of revolutions (speed) on BOD removal efficiency was such that although BOD removal efficiency was highest at low speed at L/S = 3 and 5, it was also high at medium speed at L/S = 1. The influence of L/S on COD removal efficiency was likewise inconsistent. Although COD removal efficiency tended to be higher as L/S became high at low speed, it increased in the order of L/S = 3, 1, and 5 at medium speed. At high speed, it was highest at L/S = 1, whereas COD removal efficiency when L/S = 3 was equal to that obtained at L/S = 5. The influence of the number of revolutions (speed) on COD removal efficiency was such that COD removal efficiency was highest at low speed. The influence of L/S on T-N removal efficiency was negligible; the highest T-N removal efficiency was observed at low speed.

As an overall tendency observed in the results described above, washing efficiency achieved highest values when the number of revolutions was low and L/S = 5. As compared with the first washing, although the effectiveness of second washing on COD was not observed, the influence produced on T-N at low speed and L/S = 1 was comparable to the effectiveness of first washing at L/S = 10. These results indicate that a combination of first washing with L/S = 1 and second washing at low speed and L/S = 5 represents an effective pretreatment for RPF production.

**PHOTO 1:** Rotary washer for second washing.

## 2.3 Experiment on antichlors

An experiment was conducted to examine the antichlors of reducing HCl gas generated in burning of RPF produced from waste plastics subjected to the second washing. A standard concentration was established for HCl gas present in gas emitted from RPF of JIS grade A, in which Cl content was 0.3% or less. The final goal was to meet the standard while using an RPF Cl content of 0.5%. This value corresponds to an RPF of JIS grade B, in which Cl content exceeds 0.3% but lower than 0.6%. To observe the dechlorination effect, we used PVC, which has a high conversion ratio to HCl, rather than RPF (Kawamoto et al., 2011).

**TABLE 3:** Conditions of second washing experiment.

| | |
|---|---|
| Number of revolutions of high-speed rotary washer | 700 rpm (low speed), 880 rpm (medium speed), 1250 rpm (high speed) |
| Particle size of waste plastics to be washed | 40 mm |
| Liquid - solid ratio | 1 time, 3 times, 5 times |
| Matter to be analyzed | Waste plastics to be washed regarded as inappropriate for final disposal at such sites for inert waste.<br>Waste plastics before and after washing; wastewater after washing. |
| Items analyzed (waste plastics before and after washing) | pH, EC, Cl, COD, BOD, T-N, T-C, TOC, IC |
| Items analyzed (wastewater after washing) | pH, COD, BOD, T-N, T-S, $S^{2-}$, SS, T-P |

**TABLE 4:** Removal efficiency of biological oxygen demand (BOD), chemical oxygen demand (COD) and total nitrogen (T-N) by second washing.

| **Liquid-solid ratio** | | **1 time** | **3 times** | **5 times** | **1 time** | **3 times** | **5 times** | **1 time** | **3 times** | **5 times** |
|---|---|---|---|---|---|---|---|---|---|---|
| **Number of revolutions** | | | **Low speed** | | | **Medium speed** | | | **High speed** | |
| BOD | Before washing (mg/L) | | | | | 56 | | | | |
| | After washing (mg/L) | 13 | 8.8 | 6.6 | 9.2 | 13 | 7.9 | 14 | 13 | 17 |
| | Removal efficiency (%) | 77% | 84% | 88% | 84% | 77% | 86% | 75% | 77% | 70% |
| COD | Before washing (mg/L) | | | | | 110 | | | | |
| | After washing (mg/L) | 37 | 25 | 23 | 33 | 47 | 26 | 49 | 56 | 56 |
| | Removal efficiency (%) | 66% | 77% | 79% | 70% | 57% | 76% | 55% | 49% | 49% |
| T-N | Before washing (mg/L) | | | | | 64 | | | | |
| | After washing (mg/L) | 2.3 | 2.1 | 2.3 | 4.8 | 8.4 | 3.1 | 8.2 | 8.9 | 8.3 |
| | Removal efficiency (%) | 96% | 97% | 96% | 93% | 87% | 95% | 87% | 86% | 87% |

*2.3.1 Overview of the experiment*

Nine agents and seven types of waste (Table 5) were selected as antichlors, and their dechlorination effects when used alone were investigated as single-antichlor experiments. Several agents and wastes featuring high dechlorination efficiency were selected to create mixed antichlors by combining an agent and a type of waste, and their dechlorination effects were investigated to determine the best antichlor to use in mixed-antichlor experiments.

The experiments were performed using the equipment illustrated in Figure 5. PVC and an antichlor in a porcelain dish were placed in an electric furnace. Air was introduced into the furnace by an air pump, and the emitted HCl gas was trapped in a flask. HCl gas concentration was measured by redox titration in an HCl-trapping experiment. The burning conditions of the experiment are shown in Table 6. Other properties of the emission gas were also investigated by a simplified method in an experiment conducted to assess emission gas properties.

*2.3.2 Results and Discussion*

*2.3.2.1 HCl-trapping experiment.* To develop new antichlors, the amount of HCl gas emitted and removal efficiency of HCl gas were obtained by using single antichlors, as listed in Table 5; the results are shown in Figures 6 and 7 and in Table 7. Agents meeting the standard included CaO, $NaHCO_3$, and ZnO. CaO displayed the highest removal efficiency at about 88%. Although none of the waste types met the standard, calcined lime cake displayed the highest removal

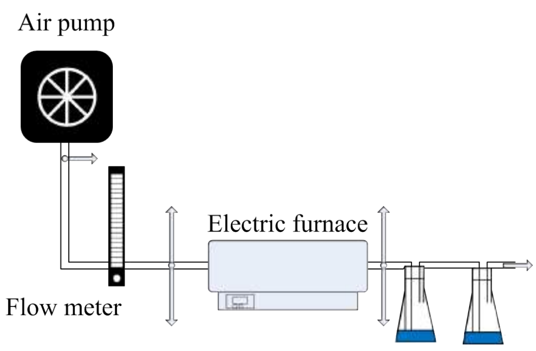


**FIGURE 5:** Schematic illustration of combustion experiment equipment.

**TABLE 5:** Selected antichlors.

| | |
|---|---|
| Agents | Calcium carbonate, magnesium oxide, magnesium hydroxide, calcium oxide, calcium phosphate, calcium hydrogen phosphate, sodium hydrogen carbonate, zinc oxide, magnesium hydrogen carbonate |
| Waste | Lime, oyster shells, zeolite, scallop shells, neutral solidifying agent, lime cake, calcined lime cake |

**TABLE 6:** Conditions of HCl-gas-trapping experiment.

| | |
|---|---|
| Temperature | 800 °C |
| Combustion retention time | 10 min |
| Supplied air flow | 700mL/min |
| Air flow time | 10 min |
| Analysis method | Redox titration |

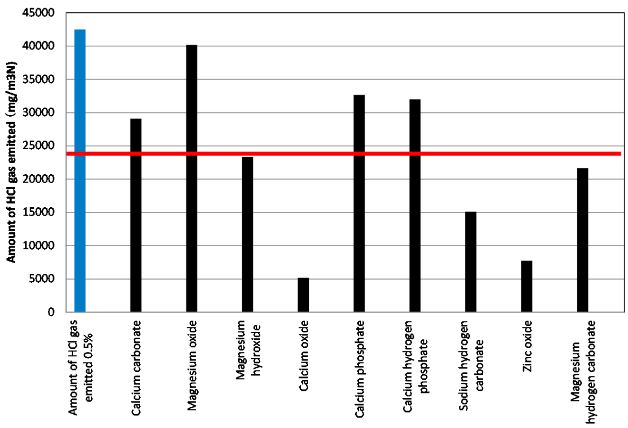


**FIGURE 6:** Amount of HCl gas emitted with single antichlors (agents).

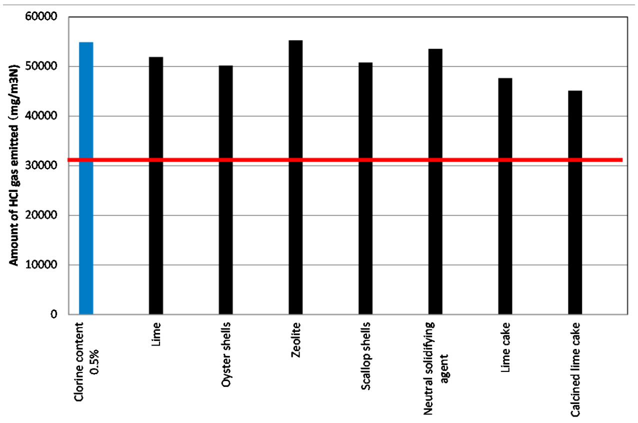


**FIGURE 7:** Amount of HCl gas emitted with single antichlors (waste).

efficiency of about 18%, followed by lime cake 13%, oyster shells 9%, and scallop shells 7%.

Subsequently, the amount of antichlor to be added was examined to enhance removal efficiency for HCl gas. However, it proved difficult to increase the quantity of the agent due to economic reasons. Thus, only the four waste-derived antichlors featuring the highest removal efficiency, as previously described, were examined, and the amounts were set at 10%, 20%, 30%, and 40% of Cl content. It should be noted that the amount of calcinated lime cake was only 40% owing to amount limitations. The results, shown in Figure 8 and Table 8, indicated that oyster shells (40%) displayed the highest removal efficiency at approx. 60%.

On the basis of these results, we selected CaO from the agents and oyster shells from the waste for the mixed antichlor, and set the agent–waste ratios at 0.5%–3.6%, 1%–1.8%, and 1%–36% considering the synergistic effect. The results of the combustion experiment of mixed antichlors are shown in Figure 9 and Table 9. As a result, only the antichlor of CaO–oyster shells with 1% and 36% added against the Cl content, respectively, met the standard, displaying a removal efficiency of approx. 71%.

**TABLE 7:** Removal efficiency of HCl gas with single antichlors (agents or waste).

| Single antichlor (agents) | Removal efficiency | Single antichlor (agents) | Removal efficiency |
|---|---|---|---|
| Calcium carbonate | 31.50% | Magnesium hydrogen carbonate | 49.05% |
| Magnesium oxide | 5.49% | Lime | 5.45% |
| Magnesium hydroxide | 45.11% | Oyster shells | 8.59% |
| Calcium oxide | 87.83% | Zeolite | -0.65% |
| Calcium phosphate | 23.15% | Scallop shells | 7.48% |
| Calcium hydrogen phosphate | 24.70% | Neutral solidifying agent | 2.49% |
| Sodium hydrogen carbonate | 64.44% | Lime cake | 13.20% |
| Zinc oxide | 81.74% | Calcined lime cake | 17.82% |

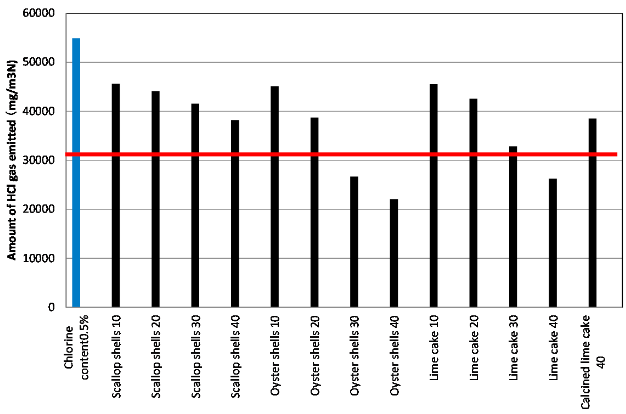


**FIGURE 8:** Amount of HCl gas emitted by addition ratio of single antichlors (waste).

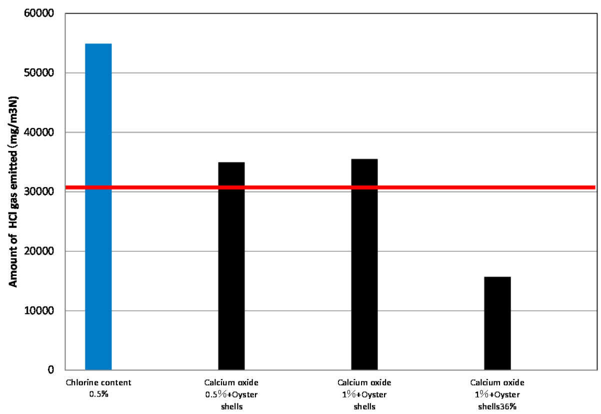


**FIGURE 9:** Amount of HCl gas emitted with mixed antichlors.

**TABLE 8:** Removal efficiency of HCl gas by addition of single antichlors (waste).

| Single antichlor (waste) | Removal efficiency |
|---|---|
| Scallop shells 10 | 16.90% |
| Scallop shells 20 | 19.67% |
| Scallop shells 30 | 24.28% |
| Scallop shells 40 | 30.38% |
| Oyster shells 10 | 17.82% |
| Oyster shells 20 | 29.46% |
| Oyster shells 30 | 51.43% |
| Oyster shells 40 | 59.74% |
| Lime cake 10 | 17.08% |
| Lime cake 20 | 22.44% |
| Lime cake 30 | 40.17% |
| Lime cake 40 | 52.17% |
| Calcined lime cake 40 | 29.82% |

**TABLE 9:** Removal efficiency of HCL gas with mixed antichlors.

| Mixed antichlor | Removal efficiency |
|---|---|
| Calcium oxide 0.5% + Oyster shells 3.6% | 36.29% |
| Calcium oxide 1% + Oyster shells 1.8% | 35.34% |
| Calcium oxide 1% + Oyster shells 36% | 71.38% |

*2.3.2.2 Experiment on emission gas properties.* Using the selected antichlor, the properties of the emission gas were analysed by means of a simplified method; the results are shown in Table 10. Although HCl gas concentration of a blank sample with a Cl content of 0.5% was 3,300 mg/m³N, the concentration detected in a sample with the mixed antichlor was as low as 260 mg/m³N, with a calculated removal efficiency of 92%. Indeed, the antichlor featuring a combination of 1% CaO to 36% oyster shells added against Cl content was found to act as an effective mixed antichlor, and the amount of antichlor used to treat Cl gas in emission gas was thus reduced.

## 3. CONCLUSIONS

The results of this study are summarized in the following points:

1. The effect of the first washing at L/S = 1 was confirmed, as was the effect of first washing to roughly remove adherent organic matter. The first washing was also found to be suitable for use as pretreatment prior to second washing for thermal recycling.
2. As an overall tendency observed in waste plastics after the second washing, washing efficiency was found to be highest when the number of revolutions was low (700 rpm) and L/S was 5. With regard to the influence of L/S at each number of revolutions on removal efficiency, a higher L/S related to a higher removal efficiency of organic pollutants.
3. As compared with first washing, although the effectiveness of the second washing against COD was not observed, the effectiveness of second washing against T-N at low speed and L/S = 1 was comparable to that obtained with the first washing at L/S = 10. These results indicate that a combination of first washing with L/S = 1 and second washing at low speed and L/S = 5 constitutes an effective pretreatment for RPF production. Moreover, to lengthen the second contact time of washing water and waste plastics in the high-speed rotary washer, an increased cleansing efficiency may be expected due to a "gradual change in the rotational speed at a low-speed side" and by "soaking washing".
4. As an antichlor to be added to RPF with low Cl gas emission, the addition of 1% CaO and 36% oyster shells against Cl content was found to be effective, and the amount of antichlor used to treat Cl gas in the emission gas was thus reduced.

**TABLE 10:** Analysis results of emission gas properties.

| Analysis | Items | | Units | Sample name: Blank sample | Sample name: Sample with mixed antichlor |
|---|---|---|---|---|---|
| Emission gas analysis | Average temperature of gas in furnace | Measured value | °C | 808 | 810 |
| | Moisture content | Measured value | % | 9.3 | 10.5 |
| | Dust concentration | Measured value | $g/m^3_N$ | 1.7 | 1.8 |
| | | $O_2$ - reduced value | $g/m^3_N$ • 12% | 5.5 | 5.4 |
| | CO concentration | Measured value | ppm | 1100 | 1400 |
| | | $O_2$ - reduced value | ppm • 12% | 3500 | 4200 |
| | NOx concentration | Measured value | ppm | 10 | 14 |
| | | $O_2$ - reduced value | ppm • 12% | 32 | 42 |
| | SOx concentration | Measured value | ppm | 21 | 15 |
| | | Amount emitted | $m^3_N$ / h | 0.0000013 | 0.0000009 |
| | HCl concentration | Measured value | $mg/m^3_N$ | 3300 | 260 |
| | | $O_2$ - reduced value | $mg/m^3_N$ • 12% | 11000 | 780 |

Temperature of gas in the furnace; concentrations of dust, CO, NOx, SOx, and HCl measured 20 min after sample insertion.
Moisture content measured 5 min after sample insertion.
SOx emission calculated on the basis of 0.06 m3N/h of the amount of emitted gas.

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# OLD MUNICIPAL AND INDUSTRIAL WASTE LANDFILLS: EXAMPLES OF POSSIBLE APPLICATION OF GEOPHYSICAL SURVEY TECHNIQUES FOR ASSESSMENT PRIOR TO RECLAMATION

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**ABSTRACT**

Further to the implicit environmental risk, abandoned municipal and industrial waste landfills frequently represent a serious problem, particularly as adequate information relating to their depth and lateral extension is either very limited or completely lacking. Moreover, knowledge of waste consistency, presence and quality of saturating fluids, metals and so on, should not be neglected when designing reclamation procedures. With the purpose of overcoming -at least partially- the abovementioned lack of information, the potential role of geophysical techniques including gravity, electrical and seismic methods, and their convenience of use, is proposed and examples provided. The first example relates to a combined application of gravity and shallow reflection geophysical methods in a mine tailings basin; in this case, the geophysical results obtained allowed the geometry and depth to the bottom of the landfill, the presence of fractures affecting the bottom and the density of disposed materials to be estimated; the second example related to application of seismic refraction tomography by means of which the structure of an old municipal waste landfill was derived; in the third example an old disposal site hosting demolition rubble was explored using the electrical resistivity tomography technique. On the whole, the convenience of applying geophysical exploration techniques for the pre-reclamation assessment of old waste landfills has been demonstrated.

## 1. INTRODUCTION

In the majority of cases, old waste landfills represent a serious environmental problem, and site reclamation interventions should be carefully designed and estimated prior to realizing the reclamation works. Therefore, an accurate knowledge of the landfills is strictly necessary while, in many cases, the overall information relating to the old landfills is scarce, with even details of their horizontal extent and depth lacking.

Direct investigation methods that provide information on the surface and subsurface extent and composition of wastes include excavating shallow test pits, using direct-push exploration techniques and drilling boreholes. However, drilling into or through the waste and into the underlying soils and/or bedrock should be performed only if necessary, and only if the driller is experienced in the methods used to prevent cross-contamination. Additional health and safety concerns (especially exposure to methane gas) should be addressed in the health and safety plan when borings are located in the waste; it should also be mentioned here that information obtained from direct investigation is punctual and generally expensive. On the contrary, surface geophysical methods (e.g. see Dobrin and Savit 1988; Sharma 1997) are prevalently non-invasive and may play an important role in delineating and characterizing the waste dump (Balia and Littarru, 2010; Cardarelli and Di Filippo, 2004; De Iaco et al., 2000; Johansson et al. 2007).

The following sections will attempt to demonstrate how geophysical techniques can contribute to the pre-reclamation assessment of old landfills; namely, applications of gravity, seismic -both reflection and refraction- and electrical resistivity techniques will be shown.

## 2. A COMBINED APPLICATION OF GRAVITY MEASUREMENTS WITH SHALLOW REFLECTION SEISMOLOGY

In this case, the gravity and the shallow reflection methods have been used.

As known - and widely illustrated in several textbooks of applied geophysics (e.g. Sharma 1997)- the physical parameter at the base of the gravity method is the density

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of the underground materials. Gravity measurements are normally taken at the surface by means of gravity meters, which measure relative values of the gravity field. Field measurements should be compensated for the effects of the elevation of measuring points, for the latitude and mass distribution in the surroundings, to enable the residual point-to-point differences to be attributed exclusively to underground density variations. Gravity surveys are often carried out along profiles conveniently placed at the surface. Therefore, in the case of waste disposal sites, if the density of wastes differs from the density of the land that hosts the landfill, a gravity anomaly can be measured and processed.

Figure 1 shows the mine tailings basin of San Giorgio situated in the most relevant mining district of Sardinia, Italy where the gravity survey concerned was carried out.

Gravity measurements were made along a line (see Figure 1) at 10 m intervals by means of a Lacoste&Romberg model G gravity meter. The associated anomaly profile (relative gravity after compensation) is shown in Figure 2, clearly depicting a gravity depression correlated to waste thickness and density.

However, full interpretation of the gravity anomaly was a problem featuring two unknown elements, i.e. 1) the depth to the bottom of the basin (or thickness of the waste) and 2) the density of the waste. Since the direct measurement of mean density of the waste was a difficult task, a decision was made to identify the bottom of the basin by means of a shallow reflection seismic survey partially superimposed to the gravity profile. It is perhaps pointless to underline that the physical property underlying seismic methods is the velocity of propagation of the elastic - or seismic- waves. As well known, the seismic reflection method is very likely the best-developed method in applied geophysics, particularly due to its role in geophysical prospecting for oil and gas (Dobrin, 1976; Dobrin and Savit, 1988; Yilmaz, 1987). Since the 1990s, reflection seismology has been progressively adapted for use with shallow and ultrashallow targets (e.g. Balia and Gavaudò, 2003 and references therein). In the present case, the shallow reflection profile was achieved using a 48 channel Abem MK6 seismograph, geophone interval 1 m, single geophones natural frequency 40 Hz undamped, shot interval 1 m, energy source 8 kg sledge-hammer, maximum CMP fold 2400%, recording length 0.250s,

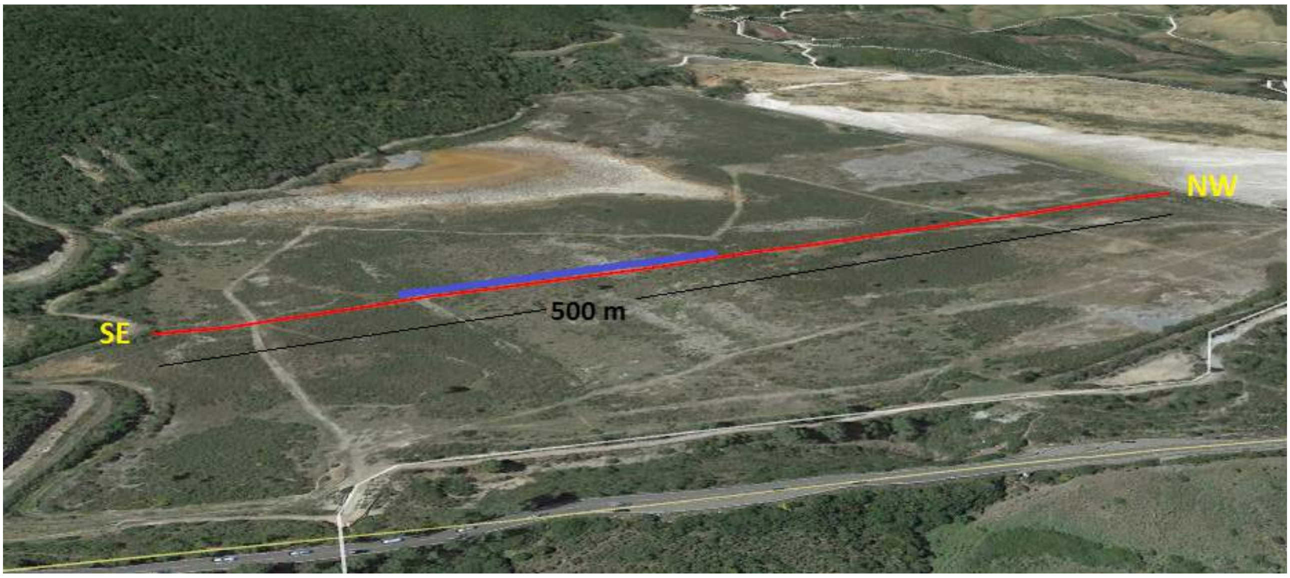


**FIGURE 1:** The San Giorgio mine tailings basin (southwestern Sardinia – Italy). The red and blue lines indicate, respectively, the position of the gravity and the shallow reflection profiles.

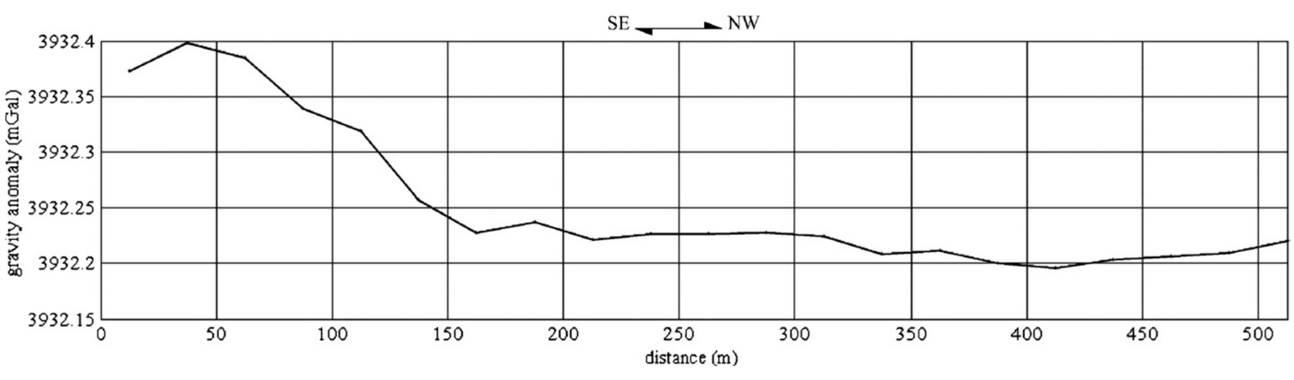


**FIGURE 2:** Gravity anomaly measured along the profile crossing the mine tailings basin of San Giorgio as shown in Figure 1.

and sampling interval 0.00025 s (Balia and Littarru, 2010). The seismic reflection section time-to-depth converted -also by means of one calibration borehole- is shown in Figure 3: this clearly shows the original ground surface, i.e. the bottom of the tailings basin: this way the only unknown of the problem was represented by the density of waste. In the end, in order to estimate the above said density, the gravity anomaly curve was interpreted using an iterative modeling procedure consisting in the introduction into the calculation software of different values of density to obtain the best match between the calculated and measured anomaly, assuming for the landfill a 2.5D mass distribution model. The result of this operation is shown in Figure 4: the best fit between measured and computed anomaly was obtained with a density of 2140 kg/m$^3$ for the waste, while the density of the hosting rock, largely known, was 2500 kg/m$^3$. Subsequently, depth and shape of the bottom, bottom fracturing conditions and waste density were estimated.

In total, the fieldwork carried out involved two days plus five days for data processing and results display. The cost of the entire geophysical work was in the order of costs required for the single calibration hole.

## 3. AN EXAMPLE OF APPLICATION OF SEISMIC REFRACTION TOMOGRAPHY

Refraction tomography (e.g. Azwin et al. 2013, Osypov 2001 and references therein) is a relatively recent technique that has largely replaced the classical technique of seismic refraction method -based on travel time analysis and interpretation- and the ultrashallow reflection technique (Balia 2013 and references therein). In the seismic tomography technique the acquisition procedure is no different from that used in traditional refraction surveying, although in this case the amount of data is significantly greater and data processing is somewhat different.

The location at which data for this experiment were acquired is shown in Figure 5 as it is today. The landfill was active until the end of the 1970s, when it was simply covered with plant land and covered by a grassy mantle, and was therefore not discernible as a site hosting a large volume of waste. Figure 6 shows the seismic tomography section along the profile in Figure 5. In detail, a Seistronix Abem RAS seismograph and 36 geophones at 2 m interval were used, with 22 on-line shots realized by means of a

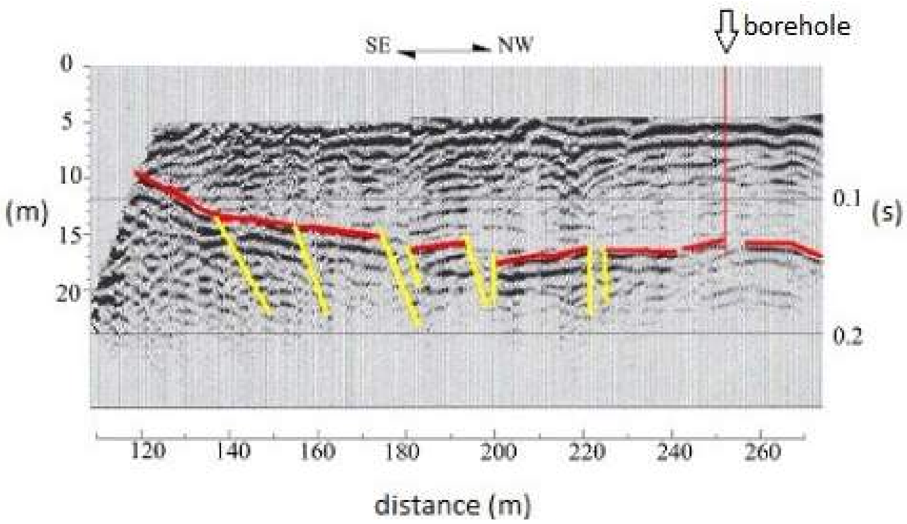


**FIGURE 3:** Depth-converted seismic section partially superimposed to the gravity profile as shown in Figure 1. The bottom of the basin, as well as several faults affecting the latter, are clearly identifiable.

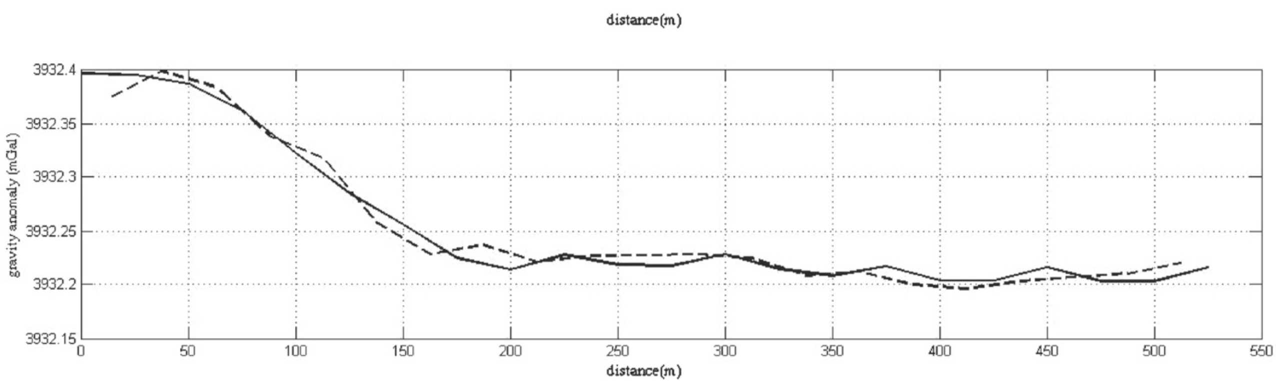


**FIGURE 4:** Best fit between the measured (solid line) and computed gravity anomaly (dashed line).This result was obtained for a density contrast of - 360 kg/m$^3$ between the waste and the host rock.


**FIGURE 5:** The location of the old municipal waste dump (southcentral Sardinia -Italy). The red line indicates the position of the seismic spread for the tomography shown in Figure 6.

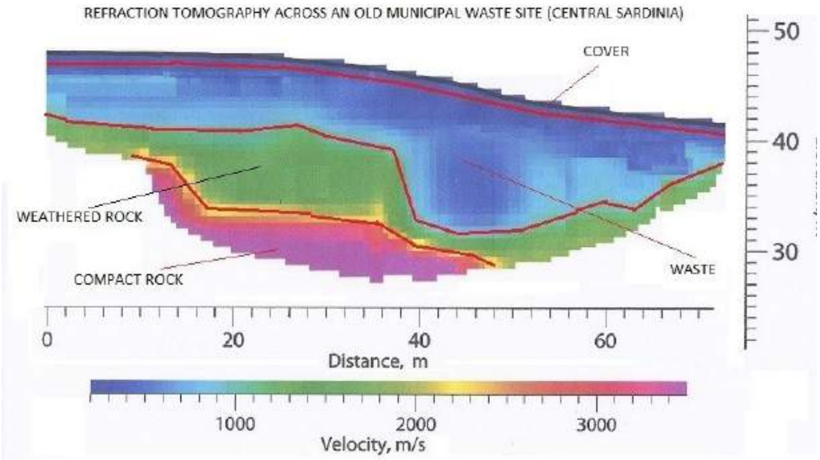


**FIGURE 6:** Refraction tomography (compressional waves) along a profile crossing the old municipal waste site located as shown in Figure 5.

simple hand-hammer to cover the section with (36 x 22) = 792 seismic rays. As mentioned, the processing software is quite sophisticated: briefly, it uses only the acquired first-arrival times, planimetric coordinates and elevations of both geophones and shots to derive the distribution of seismic wave velocity in the subsoil. In detail, the software (Optim- SeisOptPro V5.0) employs a non-linear optimization technique known as ASA -Adaptive Simulated Annealing (e.g. Ingber 1996).

As shown in Figure 6, the quality of information provided by the tomography is very good: the typical levels of the dump are clearly depicted and, from top to bottom, can be interpreted as follows: 1) ground cover approx. 1 m thick with P-wave velocity of 200-300 m/s; 2) waste body 3-12 m thick with P-wave velocity of 400-800 m/s; 3) upper portion of the bottom made up of weathered Miocene marl with a thickness of 2-6 m and P-wave velocity of 1700-2000 m/s; 4) compact Miocene marl with P-wave velocity of 3000-3500 m/s. Since the Miocene marl is known to be impermeable, it therefore ensues that at least the deep aquifers are protected from pollution. The above interpretation has been validated by means of a shallow excavation on the right end of the tomography. The field-work took place over half a day, plus two days for data processing, thus yielding a highly advantageous cost-benefit ratio for seismic tomography.

## 4. AN EXAMPLE OF APPLICATION OF ELECTRICAL RESISTIVITY TOMOGRAPHY (ERT)

As widely known, underground resistivity is a highly relevant property since it depends on base properties and characteristics such as mineralogical composition, porosity, ground water and its salinity and/or contamination, clay content, compaction degree, soil lithology, fracture zones, etc. Electrical resistivity is usually estimated by transmitting into the underground an electrical current using two current electrodes and measuring the electrical potential difference -induced by the above said current- between two other electrodes. Then, based on current intensity and induced potential difference and taking into account the position of the four electrodes, i.e. the geometry of the quadrupole, the underground resistivity can be estimated applying Ohm's law for a half-space (Sharma 1997, Parasnis 1996). A series of different electrode array configurations are available, but all configurations are aimed at gathering data to be used in estimating lateral and vertical variations in ground resistivity values; among these configurations the famous "Schlumberger", "Wenner" and "dipole-dipole" should be cited (e.g. see Parasnis 1996). With regard to electrical resistivity tomography (ERT) data acquisition is normally carried out using the "Wenner" or/and the "dipole-dipole" electrode configuration; the basic differences of ERT compared to traditional procedures (vertical soundings, resistivity profiles, resistivity pseudo-sections) are represented by the larger quantity of acquired data and the data processing method (e.g. Chambers et al. 2006; Sing et al. 2010; Tripp et al. 1984 and references therein).

Figure 7 shows the results obtained from a set of electrical resistivity data referring to a relatively old disposal site hosting demolition rubble, situated in south Sardinia. Data acquisition was carried out at the beginning of spring after a relatively rainy winter, and therefore the subsoil was significantly wet. The cover of the wastes was made of moderately clayey soil, which was not particularly wet due to sun and wind. Data acquisition and processing were performed using the electrical resistivity tomography (ERT) technique, by means of an IRIS Instruments Syscal Pro apparatus. In the ERT technique the acquisition of data involves the provision of a large number of electrodes along the measuring profile, all connected to the measuring apparatus by a multipolar cable. The measuring software automatically selects all possible quadrupole configurations and, for each of these, measures the apparent electrical resistivity of the subsoil. In the case at hand, 126 electrodes at 1 m intervals were arranged and the Wenner configuration was selected.

Once the measurements are completed, the system processes these to provide the true-resistivity section, or ERT tomography. In addition to the measurement of resistivity, the use of modern equipment also facilitates the obtaining of induced polarization (IP) measures (Parasnis 1996) . However, the acquisition and processing times for induced polarization measures may be significantly extended, implying in the majority of cases that resistivity measurements alone are executed, with IP measurements being made only when strictly necessary.

Interpretation of the resistivity tomography is clear: it shows a 1-2 m thick top layer with an electrical resistivity of 20-90 ohm-m covering the waste. The underlying wastes are not homogeneous and their variable resistivity suggests the presence of varying degrees of humidity: in fact very low, medium and relatively high resistivity can be observed.

The lowest resistivity (1-14 ohm-m, blue color) very likely corresponds to waste moistened with water salinized by dissolution of the salts contained in rubble and to the presence of concrete reinforcing iron; medium resistivity (30-80 ohm-m, green color) is associated with moderately wet rubbles; highest resistivity (up to 300 ohm-m and more) may be interpreted as dry rubble, very likely with voids.

This interpretation was supported by the findings of three very shallow excavations carried out in the center and close to both sides of the tomography. Data acquisition and processing required a full day's work, yielding a satisfactory level of information. The tomography does not show the bottom of the dump.

## 5. CONCLUSIONS

The examples shown in the previous sections do not cover the complete spectrum for the wide potential of geophysical methods in the assessment of old waste disposal sites. However, it has been demonstrated how relevant information for use in the proper planning of reclamation interventions, namely geometry of the dump, physical properties of the waste and hydrogeological and geotechnical conditions of the site, may be obtained by means of appropriate well designed and well executed geophysical surveys.

From a strategic point of view, gravimetric surveys may be conveniently employed, particularly when the external borders of an old landfill need to be identified; seismic techniques yield information on the thickness of the waste body, the degree of compaction of the waste and the conditions of the bottom; electrical techniques provide information on hydrogeological conditions and leachate presence and distribution. Moreover, it should not be overlooked that the cost/benefit ratio of geophysical techniques is considerably advantageous compared to that of direct survey techniques, the use of which following a previous geophysical study, may be conveniently optimized.

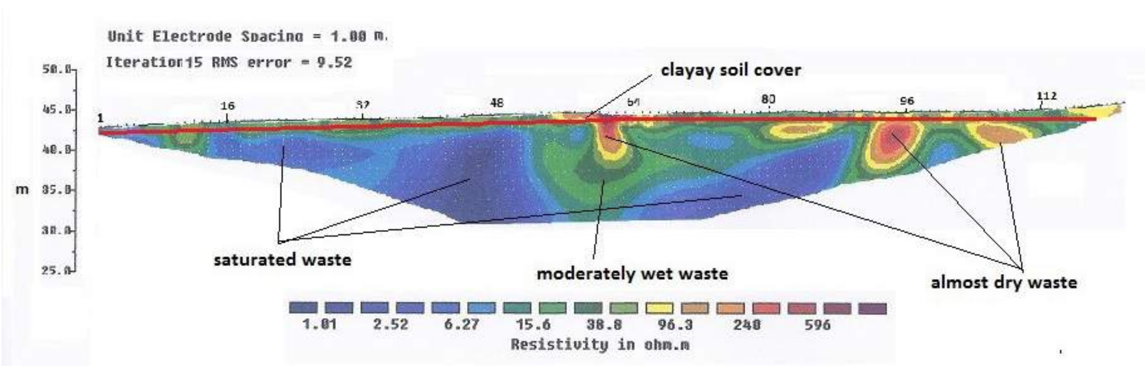


**FIGURE 7:** Electrical resistivity tomography on an old disposal site hosting demolition rubble in southern Sardinia - Italy.

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# ROBUST AND RELIABLE TREATMENT OF LEACHATE AT A CLOSED LANDFILL SITE IN SUSSEX, UK

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**ABSTRACT**

As fewer new landfill sites are opened and operated, increasingly the management of older sites containing large masses of domestic wastes is becoming increasingly important. Safe treatment and disposal of leachates is generally a key issue, and at many older unlined sites, the ingress of rainfall or groundwater is a significant issue needing consideration. Such leachate can typically be relatively weak, but is characterised by large seasonal variations in generation rate, in response to winter rainfall. Results have wide application at many closed landfill sites, which are often located far from access into the public sewer, where on-site supervisory staff are no longer available, and where wide seasonal variations in leachate generation rates pose a particular challenge. By a combination of the robustness of the SBR treatment process, and incorporation of automated SCADA controls, with remote access, such plants can operate reliably with minimal operator inputs.

## 1. INTRODUCTION

This paper will describe a case study at the closed Small Dole Landfill Site in Southern England, where leachate quality is strongly methanogenic, but year-round contains typically between 100 and 150 mg/l of ammoniacal-N. In spite of this, leachate flow rates have varied between 100 and 700 $m^3$/d since 2010, when a full-scale leachate treatment plant was designed and constructed, by substantial refurbishment and reconstruction of an existing treatment system.

The paper will describe the problems faced, the solutions adopted, and will present seven years of detailed operational data. 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. 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.

Final effluent is retained within a Treated Leachate Balance Tank, adjacent to the watercourse, and programming with tidal information allows for discharges of treated leachate to be made in accordance with tidal flows, although this is not a licence requirement.

Process design of the new leachate treatment plant included detailed laboratory treatability trials, and although (based on available leachate generation data) the plant was not originally designed for flows as high as 700 $m^3$/d, treatment has been so effective that the process was readily enhanced and extended to allow this to be carried out reliably.

### 1.1 Implementation of SBR treatment systems

Numerous recent papers, including Robinson T.H., 2013, 2015, and Robinson H.D et al., 2013, emphasise the advantages of the modified Sequencing Batch Reactor (SBR) system for treatment of landfill leachates, whereby regular volumes of leachates containing high concentrations of ammoniacal-N (toxic to bacteria that nitrify them to nitrate), can first be diluted within a large treatment reactor volume essentially containing treated leachate, such that bacteria are not inhibited (Ehrig and Robinson H.D., 2010). After a 20-hour extended aeration period, aeration of the reactor stops, suspended solids settle, and clarified treated leachate is decanted, before the cycle starts again. The modified SBR process forms the basis for hundreds of successful leachate treatment plants in the UK (e.g. Robinson H.D., et.al, 2005).

Robinson H.D. and Olufsen J., 2007, presented a similar case study to that of Small Dole LTP, where an SBR system at Efford Landfill treated leachates biologically, prior to passage through a horizontal flow reedbed for

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polishing, before effluents are discharged into a similarly sensitive watercourse, the Avon Water. As at Efford, this paper demonstrates that a well-designed, constructed and operated SBR, with a reedbed effluent polishing system, is able to operate consistently, reliably, and cost-effectively, to meet stringent surface water effluent discharge standards at all times.

### 1.2 The use of reedbeds

The benefits of reedbeds for incorporation into the treatment of landfill leachates have long been recognized by many authors. Most commonly, reed bed systems have been used successfully both for the complete treatment of relatively weak leachates from old, closed landfills (e.g. see Robinson H.D., 1999; Robinson H.D. et.al, 1999), and also for the polishing of leachates that have been treated biologically, in order to enable effluents to be discharged safely into surface watercourses (e.g. see Robinson H.D., 1993, 1999; Robinson H.D. et.al, 2003; 2008; Robinson H.D. and Olufsen, 2007; Strachan et.al, 2007; Novella et.al, 2004). In almost all circumstances, greatest success has been achieved where concentrations of ammoniacal-N in liquids entering the reed bed do not exceed 20 mg/l, whether beds are operated as vertical or horizontal flow systems.

Robinson H.D. et al., 2015 and 2017 successfully demonstrated that reed beds have great potential to provide an environment in which biological oxidation and degradation of methane dissolved in leachates from closed landfill sites can readily, effectively, reliably and cheaply be reduced to concentrations acceptable for discharge into public sewers, but few other case studies have been reported. In addition to methane removal, those papers demonstrated that seasonal removal of low levels of ammoniacal-N (15 mg/l) was also taking place, however this had not been part of the original design purpose of the bed. It was also evident that removal of iron was taking place within that specific reed bed, where slow accumulation of iron within the bed over the longer term was taking place.

Wilson et al., 2015 proved that as well as providing successful biological treatment of leachates through the refurbished SBR process, the installation of vertical and horizontal flow reed beds at Small Dole leachate treatment plant provided effective tertiary treatment, which included the removal of small amounts of suspended solids in biological effluent to very low levels (<2 mg/l) prior to discharge. Wilson also determined that complete removal of ammoniacal-N was consistently achieved, to well within the consented discharge level of 6.0 mg/l, primarily by nitrification but also to a minor extent by uptake in the reed beds.

Robinson T.H., 2017 summarised the successful operation of three specific reedbed treatment systems, providing effective treatment of various determinands. At the first site, removal of suspended solids and iron was noticed, 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 was also evident. The second site (commissioned during 2013) has continued to remove all methane from leachate entering it, whilst the third provided very successful removal of any residual levels of ammoniacal-N and $BOD_5$. The reed bed situated there also assists in significantly reducing levels of phosphate in final effluent.

## 2. SMALL DOLE LANDFILL SITE AND LEACHATE TREATMENT PLANT

Small Dole Landfill and Leachate Treatment Plant (LTP) is owned and managed by CEMEX UK Operations Limited and is situated along the banks of the River Adur, West Sussex, UK, 10 km inland of the South Coast of England. Due to the location of Small Dole Landfill, and the tidal nature of the River Adur, the site is environmentally sensitive, and discharges of treated leachate must be monitored and regulated very carefully. When waste deposit ceased in 1995 and the site was closed, 30 Hectares of land was restored to grassland pastures.

### 2.1 Leachate Treatment Plant Update, 2010

Previously, discharges of leachate from the landfill site were controlled by pumping and spray irrigation onto the restored landfill surface, under a waste disposal license, achieving good evapotranspiration rates during warmer summer months in Southern England. Subsequently, during the 1990s, a leachate treatment plant was designed and constructed by the former owners of the site, upon closure of the landfill.

The original treatment system was inherited by CEMEX on the acquisition of the RMC Group. Following experience of the failure of the system to comply with Environmental Permit conditions that became significantly more restrictive, CEMEX invested in a major upgrade and the treatment plant was redesigned, constructed and commissioned by Phoenix Engineering during 2010. The LTP now operates as a modified SBR system, utilising previously installed underground aeration tanks, whilst incorporating a new raw leachate balancing tank and a settlement tank (Plate 1). Phoenix Engineering also installed a site-specific, bespoke SCADA system, which enables complete automation of the treatment system, and allows remote operation of the plant.

### 2.2 The Small Dole SBR Treatment Process

At Small Dole, a modified Sequencing Batch Reactor

**PLATE 1:** Aerial view of the updated Leachate Treatment Plant, following modifications made by Phoenix Engineering in 2010.

(SBR) process has been adopted in order to treat large volumes of leachate as efficiently as possible, using two pre-existing aeration tanks and a large settlement tank. This arrangement enables small volumes of leachate typically containing from 100 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.

### 2.3 The Small Dole Reed Bed Polishing System

During the discharge of treated leachate from the Settlement Tank every 6 hours, biologically-treated effluent is fed through vertical and horizontal flow reedbeds in series (See Robinson, T.H., 2017) in a successful effluent polishing process. The reed beds were installed during the refurbishment to provide tertiary treatment and additional final treatment of the effluent. Effluent then drains into a treated leachate balance tank, which is designed to enable balancing of discharge flows into the Tidal River Adur, as and when required.

Plate 2 is an aerial photograph looking in a westerly direction, from above the location of the leachate treatment plant. The view shows the vertical flow reed bed (VFRB) to the right, and the two parallel horizontal flow reed beds (HFRB) to the left, with the River Adur visible in the distance. At the far western side of the site, the Treated Leachate Balance Tank (TLBT) controls the discharge of final effluent from the reedbeds, discharging consented volumes at time intervals determined by the tidal behaviour of the River Adur at this point. The daily consent for effluent discharge into the River Adur has been set at 600 $m^3$/d, however the site has occasionally been granted temporary higher discharge rates for fully treated leachate, when extreme weather conditions have been experienced (see later).

Many previous papers have highlighted the success of combining both biological treatment of leachates with effluent polishing provided by reedbed systems (e.g. Novella et al., 2004).

**PLATE 2:** Aerial view of the vertical flow reed bed, and the two parallel horizontal flow reed beds, following construction by Phoenix Engineering in 2010.

## 3. SMALL DOLE LEACHATE FLOW RATES AND QUALITY

Although the leachate within the south east of the Small Dole Landfill is relatively well contained within underlying gault clay horizons, the base of the western side of the landfill is made up of sandstone of the Folkstone beds, which are thought to be in direct hydraulic continuity with the surrounding groundwater. Because of the geological situation at Small Dole, and the proximity of the sensitive watercourse, a series of abstraction boreholes surround the perimeter of the Small Dole site, in which pneumatic pumps control the inflow of leachate from across the site, into a raw leachate balancing tank located at the leachate treatment plant.

### 3.1 Leachate Flow Rates

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 the summer periods are below 100 $m^3$/day, whilst mean daily flows over winter are approximately 400 $m^3$/day (the winters of 2014 and 2016 were particularly wet). Figure 1 presents detailed daily flow data for the volumes of leachate being collected within the Raw Leachate Balance Tank (RLBT).

The River Adur flows from north to south past the western boundary of the landfill, and has a peak flow reported at around 115 cubic metres per second (CUMECs) and the West Adur at around 120 CUMECs (Environment Agency, 2008). Peak flow past the landfill has been predicted as being more than 235 $m^3$/sec.

### 3.2 Rates of Leachate Treatment

Originally the new treatment plant was only designed to treat previous measured peak flows of leachate from the landfill site, of up to 280 $m^3$/day. However, following a review of leachate generation rates experienced by the refurbished plant, it was concluded that the Small Dole treatment plant would in future be required to deal successfully with two challenges related to leachate generation:

1. High variations in flow rates; from typical summer leachate flows of 100 $m^3$ per day to rates as high as 700 $m^3$ per day during winter months.
2. Rapid responses to sudden rainfall events. For example, Winter 2013/14, when more than 77,300 $m^3$ of leachate needed treating during the six-month period (Nov-April), with a peak flow of 17,995 $m^3$ during March 2014 (average 622 $m^3$/d).

Records of the flows of leachate into the Small Dole LTP between 2011 and 2017 have enabled the following mean seasonal values for leachate generation to be calculated:

- Spring / Summer: (May to October) = 125 $m^3$/day
- Autumn / Winter: (November to April) = 280 $m^3$/day

Figure 2 shows that since the upgrade to the treatment plant during 2010, the system has consistently managed to treat the highly variable volumes of leachate with relative ease.

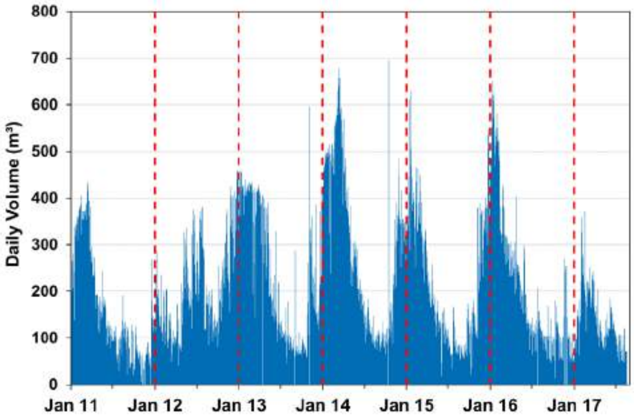


**FIGURE 1:** Daily Raw Leachate Flows at Small Dole from January 2011 to August 2017 ($m^3$).

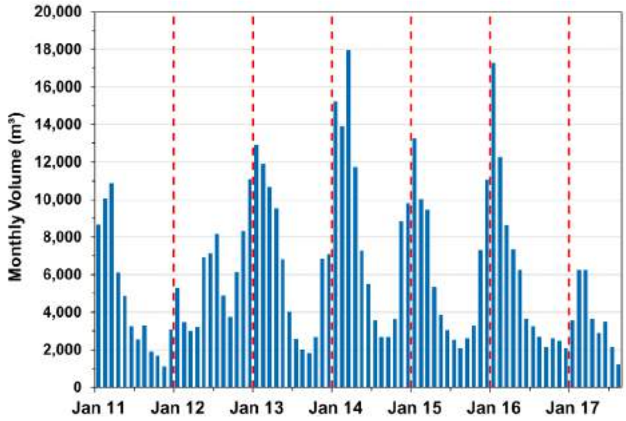


**FIGURE 2:** Monthly treated leachate volumes January 2011 to August 2017 ($m^3$).

## 3. LEACHATE QUALITY

Because of the seasonal variation in leachate generation rates, the strength of incoming leachates from the Small Dole Landfill Site depends heavily on the time of year. Table 1 highlights the differences in mean leachate strength between summer and winter periods.

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 (increases of 252% and 244% respectively), when compared to winter. Similarly, leachates produced during the summer contain 51% more ammoniacal-N than those generated during the winter periods.

Table 1 demonstrates that regardless of the time of year, or the resultant leachate strengths or volumes; leachates are consistently treated by the LTP process. COD, $BOD_5$, and ammoniacal-N are all treated to very low levels during both summer and winter periods.

The fact that there is little change in chloride concentrations between effluents and raw leachates during treatment in both summer and winter periods, is to demonstrate that dilution effects during the treatment process are not significant, in terms of changes in quality through the treatment system.

## 4. LEACHATE STRENGTHS AND SEASONAL LOADING

As discussed previously, the generation of leachates at Small Dole varies drastically, depending on seasonal variations in rainfall. Six-month periods during Autumn/Winter (November to April) and Spring/Summer (May to October) show an obvious change in leachate generation rates, where winter mean flow (280 $m^3$/day) is more than double that of summer months (125 $m^3$/day).

Although resultant strengths of leachate are much lower during winter months, it has been observed that the overall loading of contaminant concentrations on the treatment plant are significantly higher during these periods. Despite the 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.

Using chloride as an indicator for the level of dilution within the leachate being collected; Figure 3 demonstrates that although chloride concentrations are generally lower than 350 mg/l during winter months and greater than 600 mg/l during summer months, the mean daily load for chloride during winter is consistently higher than 125 kg/day, compared to mean daily loads of below 50 kg/day during the summer. The same is true for all other significant parameters, so in terms of contaminant load, the plant must provide greatest treatment during colder winter months.

Figure 4 therefore presents data for ammoniacal-N concentrations and loading results, showing a very similar trend to chloride. Although concentrations of up to 150 mg/l are reached during summer months, mean daily loads are much higher during winter periods, exceeding 20 kg/day of ammoniacal-N during every winter period; and reaching 40 kg/day during the winter of 2013/14.

## 5. RESULTS AND DISCUSSION

Although dilution of the leachate being generated within the landfill site is occurring seasonally, dilution plays no role in the treatment process itself (as highlighted by chloride concentrations in Table 1). Therefore, although concentrations of contaminants such as ammoniacal-N vary drastically on a seasonal basis, the treatment plant must treat the variable strengths of inflowing leachate feeds.

**TABLE 1:** Variations in strength of Leachate produced at Small Dole.

| 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 |

Figure 5 presents results for the concentrations of ammoniacal-N within the leachate at Small Dole, compared to the concentrations of nitrate-N in the final effluent, prior to discharge to the River Adur.

Because the points for ammoniacal-N coming in to the plant, and nitrate-N exiting the system match so well, this shows that all ammoniacal nitrogen is being effectively fully nitrified and converted into nitrate nitrogen. This, combined with the trace levels of ammoniacal-N in final effluent (presented in Table 1), demonstrates the success of the system at achieving complete nitrification treatment, as required by the discharge consent.

## 6. CONCLUSIONS

On behalf of CEMEX UK Operations Limited, during 2010 Phoenix Engineering designed, constructed and commissioned a refurbished leachate treatment system at Small Dole Landfill Site in Sussex, UK, which both automated the operation and substantially improved its performance; resulting in increased robustness, reliability, and enhanced treatment capability. Following this refurbishment, the Small Dole treatment plant has consistently been able to treat all leachates generated by the landfill, in spite of large seasonal variations in leachate volumes and leachate strengths.

During summer months, when generation rates are lower and leachate strength is stronger (ammoniacal-N greater than 100 mg/l), the treatment plant has been successful in continuously removing all contaminants down to below the required discharge consent. Although the treatment plant must deal with stronger leachates during summer periods, far greater flows of leachate generated during winter months mean that within these periods the overall mass loading of contaminants is significantly greater.

The addition of a dedicated external settlement tank to the two, parallel, pre-existing buried aeration tanks, has been successful in not only improving the overall performance of the plant but also in greatly increasing its flexibility in treatment capacity. This has enabled more than twice the originally-predicted volumes of leachate to be treated (more than 700 m$^3$), than were first envisaged (280 m$^3$/day).

As part of the refurbishment, vertical flow and horizontal flow reed beds were installed to provide successful tertiary treatment, including the removal of small amounts of suspended solids and any residual ammoniacal-N, prior to discharge into the River Adur.

The refurbished plant has performed extremely well, always achieving discharges that are compliant with the sites Environmental Permit. This provides a good example of a modified SBR leachate treatment plant, that is likely to have widespread applications at similar closed landfill sites around the world.

Results obtained at the Small Dole leachate treatment plant demonstrate how effectively SBR and reedbed 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. Using results obtained at Small Dole, future treatment plants can be designed confidently, on a similar basis, where treatment of leachates with similar loading rates, at similar volumes, is required.

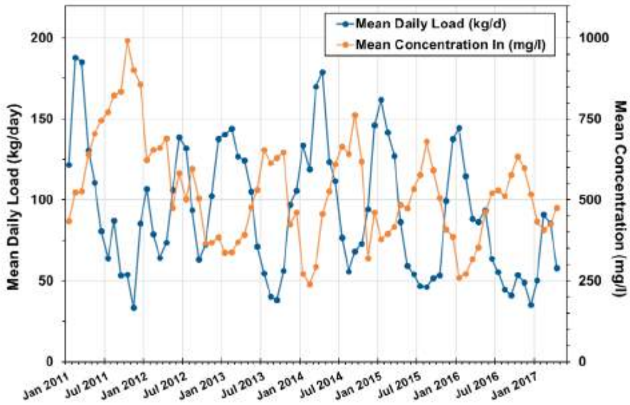


**FIGURE 3:** Chloride mean concentration (mg/l) and mean daily load (kg/day).

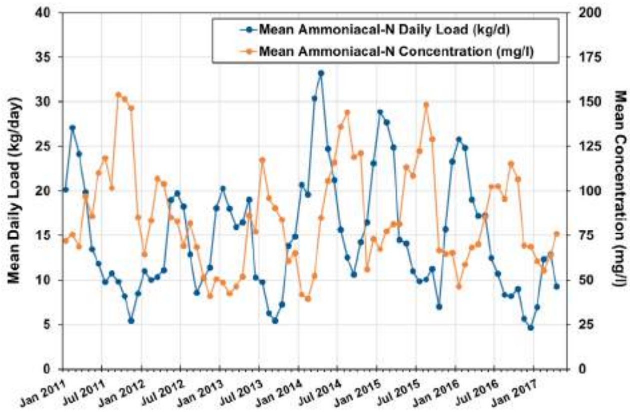


**FIGURE 4:** Ammoniacal-N mean concentration (mg/l) and mean daily load (kg/day).

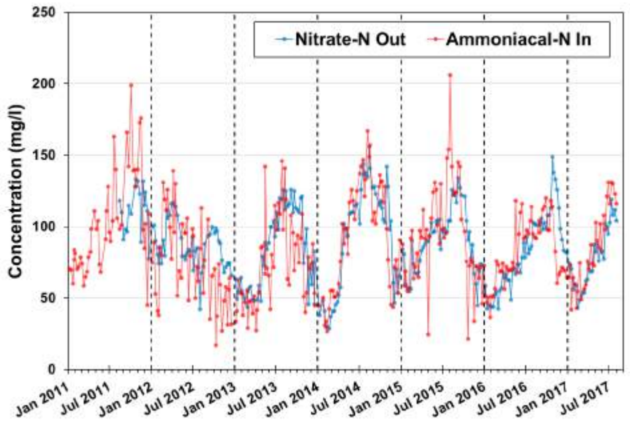


**FIGURE 5:** Concentrations of ammoniacal-N within raw leachate and Nitrate-N within final effluent at Small Dole, January 2011 to August 2017 (mg/l).


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# LAND APPLICATION OF MUNICIPAL BIOSOLIDS: MANAGING THE FATE AND TRANSPORT OF CONTAMINANTS OF EMERGING CONCERN

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**ABSTRACT**

Municipal biosolids provide organic matter to soil, and nutrients essential for crop growth. Some contaminants of standing and emerging concern such as pharmaceutical and personal care products (PPCPs), hormones, brominated flame retardants, and highly persistent perfluoroalkyl acids are not fully removed in the waste treatment process; and thus, they are often found in resultant biosolids applied to land. This paper provides an overview of selected research led by Agriculture and Agri-Food Canada on monitoring and predicting the fate and transport of these noted contaminants in field soil-hydrological environments following land applications of dewatered and liquid municipal biosolids. Different land application practices were examined in the context of their potential to reduce environmental exposure. Field studies of liquid/slurry municipal biosolids demonstrated that in macroporous soils contaminants of all types can rapidly reach shallow groundwater and tile drainage systems. Nevertheless, loads of contaminants in subsurface (tile) drainage can be significantly reduced if an aeration-based pre-tillage is employed. For dewatered municipal biosolids, directly injecting biosolids into subsoil had an indifferent effect upon water contamination, when compared with traditional surface application methods. For very high single applications of dewatered municipal biosolids to land, compounds such as the antifungal miconazole, the PBDE congener BDE 209, and perfluorooctanoic acids, for example, can persist in biosolid aggregates. Yet, for modestly macroporous soils, most of these compounds will not enter critical subsurface water receptors.

## 1. INTRODUCTION

The municipal sewage treatment process does not fully eliminate contaminants of emerging concern (CEC) (Clarke and Smith, 2011; Verlicchi and Zambello, 2015) such as pharmaceutical and personal care products (PPCPs), hormones, polybrominated diphenyl ethers, and perfuoroalkyl acids in biosolid residuals and wastewaters (Hernando et al., 2006; McLellan and Halden, 2010; Nieto et al., 2010); Venkatesan and Halden, 2014; Alder and van der Voet, 2015. In many countries and more localized jurisdictions, land application of municipal biosolids (treated sewage sludge) is routinely conducted (Joshua et al., 1998; USEPA, 1999; European Commission, 2001; Mantovi et al., 2005; Schut, 2005; Kelessidis and Stasinakis, 2012) as a means to provide nutrients and organic matter for crop growth and reduce disposal burdens. In Canada, the amounts of municipal biosolid mass that can be applied to land is often governed by heavy metal limits; limit criteria that can vary among provincial jurisdictions (CCME, 2010). CECs resulting from and application of municipal biosolids to agricultural field soils have been detected in groundwater and subsurface drainage networks (Barnes et al., 2008; Lapen et al., 2008, Edwards et al., 2009) and in surface runoff (Pedersen et al., 2005; Topp et al., 2008; Sabourin et al, 2009); also many CECs have been shown to persist in soil (Kinney et al., 2006; Gottschall et al., 2012; 2013; 2017). There is also evidence of CECs being absorbed and translocated in plants (Boxall et al., 2006; Stahl et al., 2009; Wu et al., 2010; Picó et al., 2017).

This paper summarizes a suite of studies conducted in Ontario Canada by Agriculture and Agri-Food Canada, that examine CEC persistence in soil and hydrological exposure pathways associated with land application of municipal biosolids; with consideration of: biosolid type (LMB, liquid municipal biosolids; DMB, dewatered municipal biosolids), land application rates, and methods of land application.

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## 2. METHODS AND MATERIALS

### 2.1 Study sites

Table 1 documents for selected studies, the field sites and generalized experimental details, including type/amount of municipal biosolids applied, land application methods, environmental endpoints monitored, and study duration. The Winchester study site consisted of six treatment plots and two control plots, each 100 m in length x 15 m in width. All plots were tile drained (artificial subsurface drainage). The Ottawa study site was located on an experimental agricultural field. The field consisted of four independently tile drained plots, approximately 3 ha each, of which only two plots were used for the experiment (one treatment and one control/reference). The London, Ontario study consisted of 25-30 plots of 2 $m^2$. For all study sites, no previous biosolids applications had taken place before the experiments summarized here. Table 2 lists the various types of CECs monitored for each study with selected compounds identified.

## 3. RESULTS AND DISCUSSION

### 3.1 LMB applications

*3.1.1 Winchester*

Selected PPCPs monitored were: triclosan (antibacterial), sulfapyridine, sulfamethoxazole (antimicrobials), cotinine (nicotine metabolite), atenolol (beta blocker), carbamazepine (anticonvulsant), fluoxetine (antidepressant), acetaminophen, naproxen, ibuprofen (analgesics), gemfibrozil (lipid regulator). PPCPs moved rapidly (within minutes) to tile drains (~0.8m depth) following LMB ap-

**TABLE 1:** Summary of selected municipal biosolid CEC studies. All municipal biosolids that were surface applied, with the exception of the AerWay® approach, were subjected to soil incorporation via light tillage implement within 24hrs post application.

| Study Site | Biosolid Type and Application Rate* | Application Method(s) | Environmental Monitoring | Duration of Study | References |
|---|---|---|---|---|---|
| Winchester | LMB, 93,500 L $ha^{-1}$ | Surface apply over aerated (AerWay® SSD) soil vs. surface apply on no-till soil followed by incorporation (to ~0.10 m) | Subsurface drainage (tile) | 40 days post-application | Lapen et al., (2008); Gottschall et al., (2010) |
| Winchester | DMB, 8 Mg dry weight dw $ha^{-1}$ | Direct DMB injection (to ~0.11 m) using the Terratec Environ. Ltd. direct injection system vs. surface apply on no-till soil followed by incorporation (to ~0.10 m) | Subsurface drainage (tile), groundwater, soil | ~6 months post-application | Edwards et al., (2009) |
| Ottawa | DMB, 22 Mg dw $ha^{-1}$ | Surface apply on no-till soil followed by incorporation (to ~0.10 m) | Subsurface drainage (tile), groundwater, soil, wheat grain | ~1 year post-application | Gottschall et al., (2012; 2013, 2017) |
| London | LMB, 93,500 L $ha^{-1}$ | Injection (to ~0.10 m) vs. surface apply followed by incorporation (to ~0.15 m) | Surface runoff | ~9 months post-application | Topp et al., (2008) |
| London | DMB, 8 Mg dw $ha^{-1}$ | Surface apply on no-till soil followed by incorporation (to ~0.15 m) | Surface runoff | ~1 month post-application | Sabourin et al., (2009) |

**LMB = liquid municipal biosolids, DMB = dewatered municipal biosolids*

**TABLE 2:** Selected CEC (classes) and measurement targets.

| Contaminant Class | CEC | Measurement Targets | Study Sites; References |
|---|---|---|---|
| Polybrominated Diphenyl Ethers (PBDEs)/Other Brominated Flame retardants (BFRs) | BDE-47<br>BDE -99<br>BDE-153<br>BDE-154<br>BDE-183<br>BDE-209<br>Decabromodiphenyl ethane (DBDPE)<br>1,2-Bis(2,4,6-tribromophenoxy)ethane (BTBPE) | Subsurface drainage (tile), groundwater, soil, biosolid residues, wheat grain | Winchester, Ottawa; Gottschall et al., (2010; 2017) |
| Perfluoroalkyl Acids (PFAAs) | Perfluorooctanoic acid (PFOA)<br>Perfluorooctane sulfonate (PFOS) | Subsurface drainage (tile), groundwater, soil, biosolid residues, wheat grain | Winchester, Ottawa; Gottschall et al., (2010; 2017) |
| Hormones and Fecal Sterols | Estrone<br>Testosterone Desogestrel<br>Androsterone Progesterone Coprostanol<br>Cholesterol<br>Cholestanol | Subsurface drainage (tile), groundwater,soil, biosolid residues, wheat grain | Ottawa; Gottschall et al., (2013) |
| PPCPs:<br>Antidepressants<br>Analgesics<br>Lipid regulators<br>Antimicrobials<br>Beta blockers | Fluoxetine<br>Ibuprofen<br>Gemfibrozil<br>Sulfamethoxazole<br>Atenolol | Subsurface drainage (tile), groundwater, soil, biosolid residues, wheat grain | Winchester, Ottawa, London; Lapen et al., 2008); Gottschall et al., (2012); Topp et al. (2008); Sabourin et al., ( 2009) |

plication, with surface spreading of LMB resulting in significantly higher ($p<0.05$) tile loads of PPCPs than surface spreading of LMB immediately preceded by aeration-based tillage. Maximum concentrations of PPCPs were detected exclusively where LMB was surface spread over no-till soil, ranging from 267 ng $L^{-1}$ for atenolol, to 4117 ng $L^{-1}$ for ibuprofen; sulfapyridine was only detected once above limits of quantitation (22.4 ng $L^{-1}$, at a surface spread over no-till plot). By aerating the soil using the AerWay® SSD system, application induced loads of PPCPs to tile drains via soil preferential flow paths (macropores) were critically reduced. For some of the more persistent PPCPs, there may be more parity in loading among the two land application methods over the longer term. But high concentration PPCP pulses during and immediately following land application were clearly dampened by the soil aeration methods deployed in the study. Figure 1 shows mass export of selected PPCPs.

Major polybrominated diphenyl ethers (PBDEs) monitored for this experiment were: BDE-47, -99, -153, -154, -183, and -209; Perfluoroalkyl acids (PFAAs) monitored included perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA). These compounds have human and environmental health implications (Thibodeaux et al., 2003; Costa and Giordano, 2007). Maximum concentrations in tile drainage ranged from 6-320 ng $L^{-1}$ following LMB application, and all maximum values, like the PPCPs, were observed for the surface spread over no-tilled plots. Mass loads for PBDEs were significantly higher ($p<0.05$) for surface spreading vs. control/reference plots (where no amendment was applied), but there were no significant analyte load differences between surface spreading (over no-till) and aeration tilled plots. For PFAAs, only PFOS and PFOA were found above detectable limits in subsurface tile drainage, with maximum concentrations of 17 and 12 ng $L^{-1}$, respectively, on a surface spread and aeration plot, respectively.

#### *3.1.2 London*

The selected PPCP compounds monitored were: atenolol, carbamazepine, cotinine, gemfibrozil, naproxen, ibuprofen, acetaminophen, sulfamethoxazole and triclosan. Surface runoff (generated by rainfall simulator) for plots where LMB was injected (~0.1 m depth in soil), rarely had concentrations of PPCPs above limits of quantitation, while runoff from the surface spread plots ranged from 70-1477 ng $L^{-1}$ (atenolol (70), carbamazepine (221), cotinine (83), gemfibrozil (597), acetaminophen (114), ibuprofen(1477), naproxen (509), triclosan (258)) 1 day post-application, generally declining thereafter following first order kinetics, with K ($d^{-1}$) values ranging from 0.023 for triclosan to 0.346 for sulfamethoxazole. Carbamazepine and triclosan were still detected from runoff events 266 days post-application. Results show that injection of biosolids prevents surface runoff of PPCPs, and that concentrations of selected compounds in runoff from surface applied amendment (followed by 'soil incorporation') could produce concentrations in toxicologically important ranges; notwithstanding cumulative inputs to downstream receptors. However, in terms of reducing inputs to subsurface drainage, injection into discrete furrows may augment loads of contaminants to subsurface tile drains in relation to application to surface application on tilled soil (Akhand et al., 2008). Figure 1 shows mass export of selected PPCPs.

### 3.2 DMB applications

#### *3.2.1 Winchester*

PPCPs monitored that were selected for discussion herein included: acetaminophen, fluoxetine, ibuprofen, gemfibrozil, naproxen, carbamazepine, atenolol, sulfamethoxazole, cotinine, triclosan, and triclocarban. There were no significant differences ($p>0.05$) in PPCPs loads in tile drainage among surface spread and directly injected DMB (0.05 m diam. injected continuously to a depth of ~ 0.11 m in soil) plots, although late study period (>100 days post-application) average loads were consistently higher from tiles of injected plots, but they were not different significantly ($p>0.05$). This was likely due to PPCPs in the injected DMB being more protected from photodegradation, higher soil temperatures, oxygen, and the more biologically active surface soils than the surface spread DMB which was more exposed to environmental elements and microorganisms in the surface soils. Hence, there may have been more persistence in PPCPs in the injected biosolids, in relation to those in biosolids spread on surface and lightly incorporated into soil. Maximum PPCP concentrations in subsurface tile drainage did not exceed many literature based aquatic toxicity thresholds except for one triclosan tile water sample from the surface spread plot. Surface spreading and direct injection of DMB resulted in lower concentrations of PPCPs in subsurface drainage than application of LMB, although PPCPs appeared to be more persistent in soil, especially for directly injected materials. However, by directly injecting DMB, problems associated with vector attraction and odour are minimized; without, as we have shown, increased liability of transport to subusurface hydrological receptors. Figure 1 shows mass export of selected PPCPs.

#### *3.2.2 London*

Selected compounds monitored were atenolol, carbamazepine, cotinine, caffeine, gemfibrozil, naproxen, ibuprofen, acetaminophen, sulfamethoxazole, triclosan and triclocarban. Maximum concentrations in surface runoff generated by rainfall simulator ranged from undetectable (gemfibrozil) to 110 ng $L^{-1}$ (triclosan), and time to reach maximum concentration varied from 1 to 36 days post-application. The compounds with the least mass exported (<1% of that applied) had log Kow (octanol-water partition coefficient) values of ≥3.18 (triclocarban, triclosan, sulfamethoxazole, ibuprofen, naproxen and gemfibrozil), while those with >1% mass exported had log $K_{ow}$ values of ≤2.45 (acetaminophen, carbamazepine, caffeine, cotinine, atenolol), indicating log $K_{ow}$ may be a factor in determining runoff potential of these PPCPs. Figure 1 shows mass export of selected PPCPs.

#### *3.2.3 Ottawa*

The biosolids applied to the Ottawa site represented high instantaneous surface application rates of 22 Mg dw

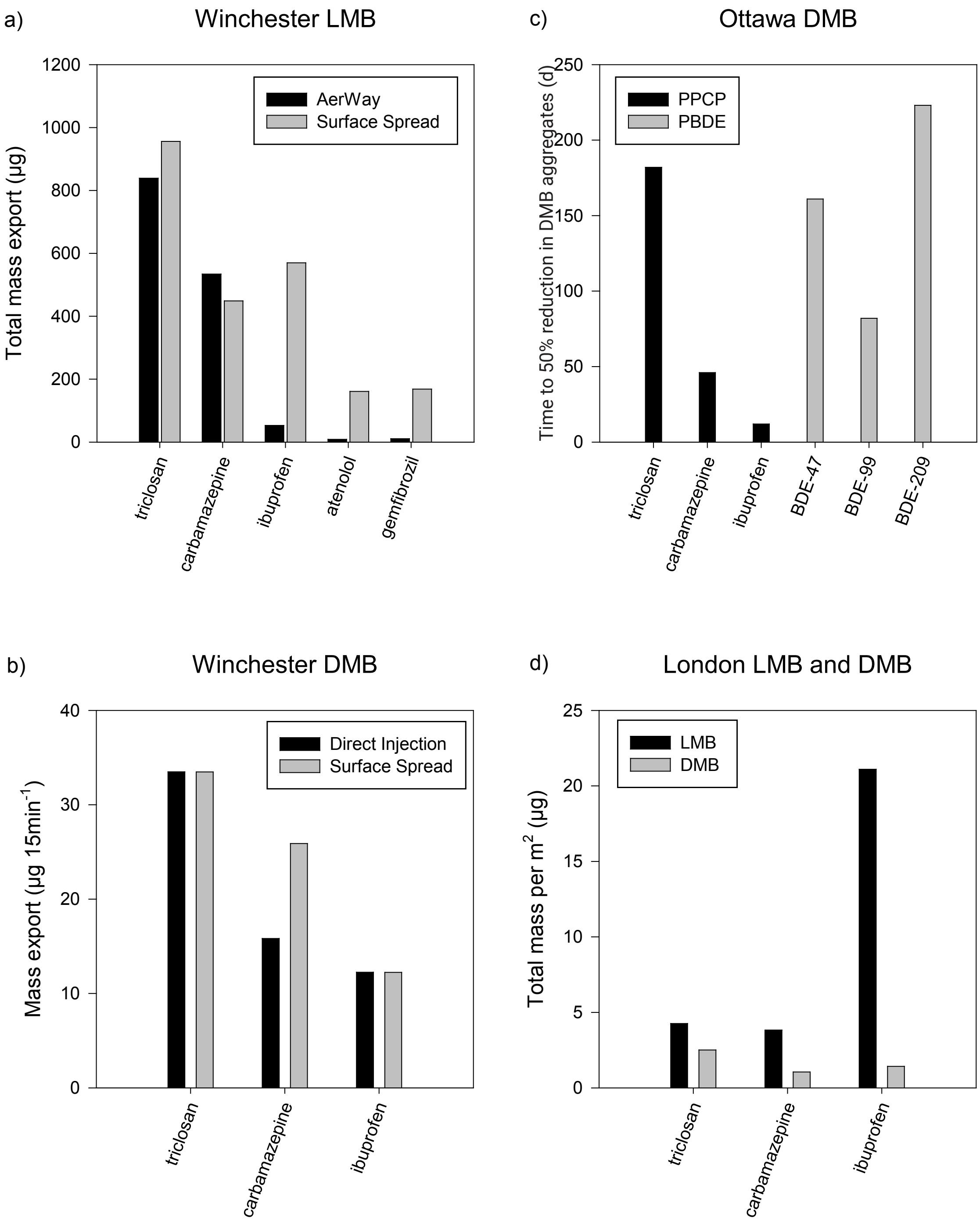


**FIGURE 1:** Selected results from biosolid studies: a) study period (40 d) PPCP mass load estimates (µg) in subsurface tile drainage associated with AerWay® LMB and surface+incorporate LMB application, b) maximum mass export (µg per 15 minutes) in subsurface tile drainage for selected PPCPs associated with DMB direct injection and DMB surface application, c) time to 50% reduction in concentration of CEC in DMB aggregates that were in soil following land application and, d) total mass per m2 (µg) of selected PPCPs in surface runoff from LMB (surface apply) and DMB (surface apply) applications (note: no PPCP concentrations for LMB injection surface runoff were above detectable limits).

$ha^{-1}$. Hormones (androsterone, desogestrel, estrone) were only detected on two occasions, up to ~2 months post-application in tile drainage (2-34 ng $L^{-1}$), but were not detected in groundwater (2 m depth). Sterols were detected up to ~1 yr post-application in tile drainage and sterol ratios were indicative of biosolid-borne contamination. The limited transport of hormones and sterols to subsurface tile drainage networks may be attributed to a combination of the hydrophobicity of these compounds and more limited macroporosity of the field soil (in relation to the Winchester study soils). The transitory contamination from hormones and sterols appears unlikely to result in any significant pulse exposure risk in subsurface drainage and groundwater, even at the high application rates examined.

Over 80 PPCPs were monitored, but only carbamazepine, ibuprofen, acetaminophen, triclosan, triclocarban, venlafaxine, and citalopram were detected in subsurface drainage (with concentrations ranging from 5–74 ng $L^{-1}$). No PPCPs were detected in groundwater >2m and those detected at 2m depth (ibuprofen, triclosan, triclocarban, venlafaxine) were only detected on one occasion within one month after DMB application. PPCPs persisted in DMB aggregates (intact DMB within the soil) up to ~1 yr post-application, however (Figure 1). But these persistent PPCPs were not critically detected in subsurface tile drainage and groundwater receptors. No PPCP was detected in wheat (grain) grown following land application.

PBDEs, other BFRs, and PFAAs were detected in subsurface drainage and 2m groundwater for up to ~1 yr post-application. Several compounds in subsurface drainage were detected at significantly higher ($p<0.05$) concentrations than reference plot (no DMB applied)/pre-application (DMB plot) concentrations (BDE-47, -100, and -153). PBDEs and PFAAs persisted up to ~1 yr post-application in DMB aggregates within the soil as well. Several PBDEs in DMB aggregates had concentration reductions >90% after 1 yr post-application, following an exponential decay pattern (Figure 1). No PBDEs or other BFRs were found in wheat grain. Although a considerable PBDE and PFAA load was applied at time of biosolid application (22 Mg dw $ha^{-1}$), only subsurface drainage showed significant increases of PBDEs relative to pre-application levels, and detection of PBDEs and PFAAs in subsurface drainage, groundwater, and soil indicated that atmospheric deposition was likely an important source of these compounds. In addition, post-application levels of PBDEs and PFAAs in the soil remained largely within background soil levels derived from the literature.

## 4. CONCLUSIONS

Transport of CECs from biosolids (liquid and solid) was tempered by limited macroporosity of soil, as well as application techniques that disrupted preferential flow paths to subsurface water resource receptors. Pre-tillage is crucial in this regard, and in cases where transport to subsurface drainage was rapid, as with the LMB application at Winchester, CEC concentrations peaked only briefly, and rarely exceeded concentrations typical of effluents associated with many waste water treatment plants. Further, although many CECs were detected in soil and water for extended periods following, in particular, dewatered biosolid applications, concentrations did not typically exceed many documented acute or chronic toxicity thresholds, and compounds were shown to dissipate considerably over time (i.e PBDEs, PPCPs). Nevertheless, cumulative effects and impacts of transformation products and metabolites needs better experimental documentation (Mompelat et al., 2009) in regard to the fate and transport pathways associated with land applied biosolids. Further, the implication of nonextractable residues on dissipation kinetics (Boxall et al., 2012) needs to be more succinctly examined, since dissipation could be very strongly linked to, among many things, the nature and mode of land application method.

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# MARINE PLASTIC LITTER - A MASSIVE WASTE PROBLEM

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**ABSTRACT**

The pollution of the oceans with anthropogenic and especially plastic litter is acknowledged as one of the major environmental stressors. Life cycle assessments of plastic products carried out to date have failed to take into account the fact that the oceans represent a final sink for plastics. At present, around 800 species have been shown to have negative interactions with marine litter, the majority relating to entanglement in and ingestion of plastic items. Additionally, marine litter causes socio-economic costs and may impact the wellbeing of society at large. The causes and sources are manifold and include insufficient producer responsibility, lack of awareness of the consequences of littering as well as poor sewage and waste management. Beside large items such as bags and bottles, the presence of microplastic particles sized 5 millimeter and smaller has also been verified in water bodies, sediments and marine organisms throughout the oceans of the world. Large garbage patches, where litter accumulates due to prevailing flow regimes have been verified in all large ocean currents. Plastics degrade very slowly in the marine environment due to physical, chemical and biological processes, and when they settle in sediments they may persist for centuries. The presence of marine litter is largely based on society´s prevailing production and consumption patterns. Meanwhile, this issue has gained increasing recognition in international and regional fora, as exemplified by the European Strategy for Plastics in a Circular Economy, various resolutions by the United Nations and Action Plans adopted inter alia by Regional Seas Conventions and the G7/G20. The challenge remains to take advantage of the current political momentum to effectively implement these Action Plans and further develop tailor-made solutions. Change can only be triggered by compiling solutions together with experts of important sectors such as from waste prevention and management and by spreading the knowledge through education at all levels and age groups.

## 1. INTRODUCTION

In addition to other key issues such as climate change, the pollution of the environment, and particularly oceans, with anthropogenic litter, is acknowledged as one of the major environmental stressors, causing detrimental impacts on marine biodiversity as reported over the last four decades (Sutherland et al., 2010). The term marine litter comprises any solid material which has been deliberately discarded, or unintentionally lost on beaches, on shores or at sea, including materials transported into the marine environment from land by means of rivers, drainage or sewage systems or winds. It includes any persistent, manufactured or processed solid material and originates from a series of sea- and land-based sources (UNEP, 2005).

Although marine litter consists of a wide range of materials including metal, wood, rubber, glass and paper, there is clear evidence that plastic litter is by far the most abundant type of material. On average, 75% of marine litter collected from European beaches is represented by various forms of plastics, with a similar predominance of plastics being reported from sampling on the seabed and in biota (Barnes et al., 2009). Due to the light weight of these products, plastics can be transported by ocean currents over long distances and are pervasive throughout our oceans from the poles to the equator, from the sea surface to the deep sea and from rivers to lakes and coastal areas.

The mass production of plastics started in the middle of the twentieth century, and traces confirming this advent are present in the Earth sediments (Zalasiewicz et al., 2016).

In addition to large items such as plastic bags or bottles, the presence of microplastic particles has also been verified in water bodies, sediments and marine organisms throughout the oceans of the world.

## 2. WHERE DOES THE LITTER COME FROM?

Plastics production increased rapidly from the 1950s, with global production reaching approx. 311 million tons in

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2014. Plastics have replaced the use of traditional materials in numerous sectors, including construction, transportation, household goods and packaging, and are also used for a series of novel applications, including in the medical field. Many different varieties of polymer are produced, although in terms of volume the market is dominated by several major types: polyethylene (PE, high and low density), polyethylene terephthalate (PET), polypropylene (PP), polyvinyl chloride (PVC), polystyrene (PS, including expanded EPS) and polyurethane (PUR) (UNEP, 2016).

Due to insufficient producer responsibility, lack of awareness over the consequences of littering, the short life cycle of many products and high consumption rates, the durability of plastics, poor sewage and waste management (see Figure 1) including badly operated and illegal landfills and untreated stormwater as well as and maritime use, particularly in the shipping and fishing sectors, a significant portion of the plastics produced worldwide enters into and persists in marine ecosystems (Lebreton et al., 2017 and Werner et al., 2017). For the year 2010 a modelling exercise was carried out for 192 coastal countries, which led to an estimated 3.5 billion metric tons of solid waste being produced, of which 275 million tons were plastics. Mismanagement of plastic litter in these countries alone led to an estimated input of eight million metric tons plastic waste entering the bordering seas, including the Mediterranean and the Black Sea in 2010 (Jambeck et al., 2015). Another study indicated that between 1.15 and 2.41 million tons of plastic waste currently enters the ocean every year from rivers, with over 74% of emissions occurring between May and October (Lebreton et al., 2017).

## 3. MICROPLASTICS

Plastic particles sized 5 millimeter and smaller are defined as microplastics. Primary microplastics are those originally manufactured with small dimensions; secondary microplastics are those resulting from the breakdown and use of larger items. An assessment of the land-based sources and emissions of microplastics released into the marine environment, has recently been carried out in the North-East Atlantic, showing that the major sources are preproduction pellets, cosmetics, abrasive cleaning agents, rubber infill from artificial sports fields, road runoff from car tyre wear, laundry fibres, and paints (Verschoor et al., 2017). The assessment also focused on the emission of larger land-based litter such as bottles and packaging, which subsequently break down in the sea to microplastic particles. The estimated source emissions are shown in Figure 2.


**FIGURE 1:** Plastic input from municipal solid waste and wastewater (source: GRID-Arendal and Maphoto/Riccardo Pravettoni: https://www.grida.no/resources/6931).

Estimated microplastic emissions in OSPAR catchments (tonnes / year)

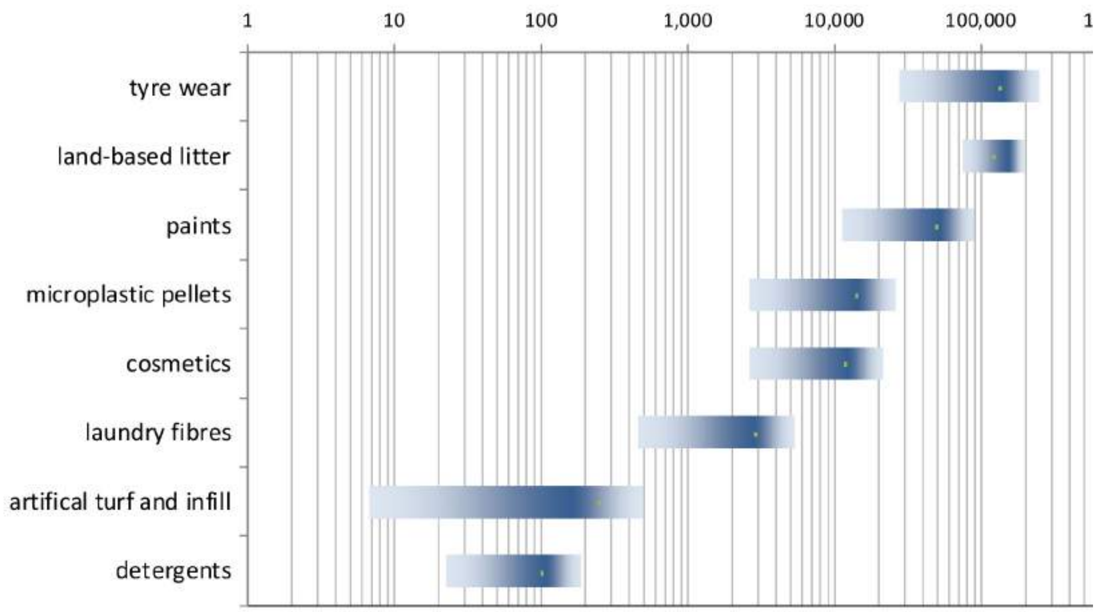


**FIGURE 2:** Estimated emissions in OSPAR catchment area (source: Verschoor et al., 2017).

## 4. TRASH GYRES IN MANY PLACES

Large garbage patches where litter accumulates due to prevailing flow regimes have raised particular concern. Although the exact size, content, and location is difficult to accurately predict, the so-called Great Pacific Garbage Patch is known to be the biggest, as it spans waters from the West Coast of North America to Japan (NOAA, 2017). Meanwhile these kinds of patches have been verified in all large ocean currents. According to a global estimation, 5.25 trillion particles with a total weight of circa 270.000 tons are floating on the surface of the oceans (Eriksen et al., 2014 - see also Figure 3). Compared to the inputs these numbers appear low, although it should be kept in mind that litter washed ashore, ingested by biota and that has sunk to the seafloor is not included here. Available data hypothesize that 70 percent of marine litter sinks to the seafloor. As an example, a time series of litter caught in fishing nets in the North Atlantic identified plastics in 62% of the trawls conducted, with densities of litter on the seabed calculated to be up to 580,000 particles per square kilometer.

The Northern fulmar, a small relative of the albatross, feeds exclusively on the open sea and regularly confuses litter particles with food. Data from approx. 1,000 dead fulmars concluded that this sea bird species ingests and egests circa six tons of plastic litter per year in the North Sea region alone. On a global scale, marine species process and redistribute hundreds of tons of plastics in a similar vein every year (Van Franeker, 2011). Data quantifying the biological degradation of synthetic polymers by microorganisms are currently still lacking.

## 5. BEST-BEFORE DATE: 2618

Tourists mainly perceive litter pollution at holiday destinations as an aesthetic disturbance alone, but for many marine species this pressure is a serious threat to their health and often even to survival. Plastics degrade very slowly in the oceans due to physical, chemical and biological processes. Given that plastic items are often buoyant, an increasing load of plastic litter is being dispersed over long distances, and when they settle in sediments they may persist for centuries (Derraik, 2002). Macroplastics fragment into millions of meso- (2.5 cm down to 5 mm) and microplastic particles (smaller than 5 mm), making them accessible to a wide range of marine biota, from primary producers to higher trophic-level organisms potentially infiltrating the entire marine food web. For example, the WWF determined for the Baltic Sea in the year 2011 that 5,000 – 10,000 gill nets were lost or discarded (WWF Po-

**FIGURE 3:** Floating marine litter in Cretan waters (source: Stefanie Werner, 2016).

land, 2011). According to scientific research the remaining fishing capacity of so-called ghost nets varies from 6-20% of their initial fishing capacity. Gill nets are made of nylon, which takes up to 600 years to degrade (Ten Brink et al., 2009). Whereas abandoned fishing nets may initially cause entanglement and strangulation of marine species for substantial timeframes, they sooner or later degrade to increasingly smaller particles. The smaller they get the wider the range of species that might ingest them.

## 6. MARINE LIFE SUFFERS

A recent literature review updated the total number of marine species known to be negatively affected by marine litter to 817 (CBD, 2016). In more than half of these studies entanglement in and ingestion of marine litter items have been documented. Circa 17% of these species are red-listed by the International Union for Conservation of Nature (IUCN) or classified as threatened or endangered. Although the interactions pose a threat to the entire population in only a few cases, it is inevitable that the known impacts will cause a deterioration of the physical condition of affected individuals, with a far greater number of organisms being affected by as yet undocumented sublethal effects. Each year, millions of animals that live in the oceans are debilitated, mutilated and killed by marine litter. In particular, packaging such as strings and sheets, as well as litter related to fishing activities, constitute a high risk for marine life. Rope and netting account for 57% of encounters of marine organisms with litter worldwide, followed by fragments (11%), packaging (10%), other fishing related litter (8%) and microplastics (6%).

An example is the Northern gannet, a seabird which collects litter items at sea for use in nest building. The breeding colony on the uninhabited island of Grassholm (Wales, United Kingdom) is the third largest worldwide with around 40,000 breeding pairs. A study investigated the use of plastics as nesting material. On average, the gannet nests contained 470 grams of plastics, which equates for the entire colony to 18.6 tons (Votiera et al., 2010). Remains of nests, ropes, strings and packaging were particularly abundant, and were also observed in the gannet colony on the German island of Helgoland, where 97% of nests contain plastics (see Figure 4). As a result, mortality due to entanglement and strangulation in litter items is two to five times higher than normally in this population (Dürselen et al., in publication).

Impacts of the ingestion of marine litter include starvation from a full stomach due to a continuous feeling of saturation, low storage of body fat, as well as injuries and blockages of the gastrointestinal tract. In addition, plastics often contain toxic or hormonal effective chemicals or absorb persistant organic pollutants from the seawater. Therefore, the ingestion of plastic particles may provide a pathway facilitating the transport of harmful chemicals to organisms. Deposit- and filter feeding fauna are particularly susceptible to the uptake or ingestion of microplastics, as well as planktonic invertebrates in oceanic gyre regions where microplastic concentrations are high. A study revealed an average ratio between microplastics and mesozooplancton weights of 0.5 in the North Western Mediterranean Sea, which might induce a potential confusion for zooplankton feeders (Collignon et al., 2012).

**FIGURE 4:** Entangled Northern Gannet on the island of Helgoland (source: Peter Hübner).

In addition, marine litter is known to act as a vector for the transport of biota, including invasive species, and to damage habitats by altering the assemblage of species, e.g. by providing artificial habitats or through smothering. As an underlying ethical aspect of the above-mentioned biological impacts, the issue of animal welfare should not be neglected. The major types of litter affecting marine animals may cause problems for a wide range of species. Impacts may result in poor animal welfare over a range of timeframes; acute impacts may produce suffering and distress for minutes, while chronic impacts may be cumulative, causing increasing suffering over periods as long as years.

Expanding on the socioeconomic perspective, the impacts are manifold. Amongst other things, marine litter may spoil the beauty of the sea and the coastal zone, interfere with fishing and damage fishing boats and gear, block cooling water intakes in power stations, contaminate beaches, commercial harbors and marinas, injure livestock and coastal grazing land, interfere with ships, causing accidents at sea, be a serious hazard to human health, particularly when composed of medical and sanitary waste, damage local economies by contaminating fish catches, driving away tourists and cost a significant amount to clean up. For example, the annual beach cleaning costs in European countries range from around 3,000 to up to 65,000 Euros per kilometer. A frequently raised concern relates to the issue of potential health threats due to the presence of microplastics and associated contaminants in seafood. However, on the basis of current evidence, the risk appears to be no more significant than via other routes of exposure (Werner et al., 2016).

## 7. TO COMBAT MARINE LITTER

The existance of marine litter is largely based on society's prevailing production and consumption patterns (OSPAR, 2014). The removal of litter from the marine environment may only succeed in capturing marginal amounts, is time- and cost-consuming, and implies additional ecological risks such as the by-catch of marine organisms and damage of habitats. Useful removal activities include passive Fishing-for-litter initiatives, where fishermen are provided with the required logistics to store litter they catch on board of their vessels and to dispose it free of costs in harbors.

However, general prevention programs should represent the major focus of our efforts to combat marine litter. The saving of resources, improving the life-cycle of products, implementing extended producer responsibility schemes, establishing adequate waste, sewage and storm water management, modifying and substituting products and raising awareness should be at the heart of resolute action to prevent further inputs of marine litter. In times of global markets, producers should pay attention to the availability of appropriate waste disposal structures at the point of destination of their goods, which includes a need to set up these structures when not available. The use of plastics should become more sustainable by applying smart product design and revising policies hampering the achievement of a reduced application of plastics. Other key issues relate to an extended durability and long service life of products, strong specifications to prevent technical obsolescence and the avoidance of single-use applications wherever possible. A fundamental aspect relates to the current widespread and non-transparent utilization of additives, which should be thoroughly revised if a true circular economy is to become a feasible possibility. A multiplicity of added substances such as softeners and flame retardants impede eco-effective recycling. Plastics turn to waste or cannot be reused in high-quality products, with only downcycling representing a possible option rather than recycling. Biodegradable polymers do not yet represent a viable option for the replacement of conventional plastics, as they only degrade faster under determined industrial conditions (e.g. constant high temperature), but not in the marine environment, and standards underlying certification of the latter are still lacking.

With regard to sea-based sources of marine litter, a 100 percent integration of waste disposal fees in the regular harbour fees is required to prevent illegal discharging from ships at sea. This No-Special-Fee system has already been widely introduced in Baltic Sea harbors. Inspections at sea and related sanctions should be intensified and the retrieval and recovery of ghost nets e.g. through gear marking should be supported on both an economic and organizational level. Last but not least, with the aim of raising awareness and preventing littering, the topic of marine litter should be comprised in all academic and professional curriculums and become a subject for general education.

Action Plans on Marine Litter focusing on a comprehensive set of actions aimed at targeting the major sea-based and land-based sources, together with the implementation of suitable removal and education activities, exist on an international (G7/G20, UNEP), regional (Regional Seas Conventions) and national (in Europe especially under the EU Marine Strategy Framework Directive) level. These instruments strive to obtain an efficient and effective horizontal multi stakeholder involvement. By considering the implications for the marine environment as an ultimate sink for litter they add weight to existing sectoral approaches of other regimes and legal frameworks.

In a European context, Regional Action Plans on Marine Litter exist for the North-East-Atlantic (OSPAR), the Mediterranean (MEDPOL) and the Baltic Sea (HELCOM). Actions related to waste management in these plans aim at identifying loopholes that result in the evolution of waste into marine litter, at raising awareness of the link between waste management and marine litter in the public opinion, the commercial sector and politics; clarifying the role of the waste management sector in preventing and reducing marine litter; identifying concrete examples for Best Practise in waste management to be promoted via the Regional Seas Conventions, and developing the latter on the basis of measures to be applied in the context of their implementation.

Close cooperation with waste management experts to achieve these goals is currently ongoing. Indeed, the author fervently hopes that events such as Sardinia 2017 will lead to a deeper connection and understanding between the "Waste" and "Marine litter" communities to achieve the vision of plastic-free seas for future generations.

## 8. CONCLUSIONS

Marine litter is regarded as one of the most threatening types of pollution to marine ecosystems. Life cycle assessments of plastic products carried out to date have failed to take into account the fact that the environment and especially oceans represent a final sink for plastics. A range of problems associated with marine litter render this a highly complex issue. Being bioavailable to many species, micro-plastic particles smaller than 5 millimeters in size, which originate from the breakdown and use of bigger items, as well as from their direct application in plastic products, are of particular concern. Plastics are highly persistent and often contain toxic or hormone-based chemicals or absorb persistent organic pollutants from seawater. At present, approx. 800 species have been found to have detrimental interactions with marine litter, the majority relating to entanglement in and ingestion of plastic litter items. Additionally, marine litter bears a high socio-economic burden and may impact the wellbeing of society at large.

Recently, this issue has gained increasing recognition in international and regional fora, as exemplified by the European Strategy for Plastics in a Circular Economy, various resolutions by the United Nations and Action Plans adopted inter alia by Regional Seas Conventions and the G7/G20. The challenge remains to take advantage of the current political momentum to effectively implement these Action Plans and further develop tailor-made solutions. Change can only be triggered by compiling solutions together with experts of important sectors such as from waste prevention and management and by spreading the knowledge through education at all levels and age groups.

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# APPLYING CIRCULAR ECONOMY THINKING TO INDUSTRY BY INTEGRATING EDUCATION AND RESEARCH ACTIVITIES

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Students

**ABSTRACT**

Collaboration between universities and external organisations offers opportunities for multiple and mutual benefits, including the development of employability skills in students. This paper outlines the educational approach taken and results achieved when under- and post-graduate students were tasked with working with a water supply and waste water treatment company (Southern Water; SW) with the aim of identifying opportunities to apply circular economy thinking to SW's operations at a waste water treatment plant (WWTP) in England. The students were presented with a "real-world" consultancy task to identify and evaluate the waste streams within the WWTP process and produce options for their reduction, recovery and reuse without hindering operational effectiveness. The mutual benefits of this collaborative venture were demonstrated via: i) the utility of students' recommendations and SW's desire to participate in and fund follow-up activities, including academic consultancy, MSc and PhD projects; ii) positive feedback from SW and the students; and iii) the quality of the exercise as a vehicle for academic learning and development of professional and employability skills. Academics can address the challenge of simultaneously needing to develop students' employability skills whilst covering core topics required by professional bodies by deliberately incorporating open-ended, real-world industrial activities into teaching and learning activities within assessed modules. Active learning approaches to education in waste and resource management incorporating consultancy-style work of this nature are strongly recommended.

## 1. INTRODUCTION

Increasing numbers of young people attending university, driven by government policies recognising the links to the growth of the economy (Glover et al., 2002), has resulted in many students struggling to find appropriate employment upon graduation. A degree qualification is no longer a guarantee of a job, and this is reflected in the attitudes of many students who report that they chose to study at university not solely for academic advancement but to make them more employable (Glover et al., 2002; Gedye et al., 2004).

Studies have shown that that students who opt for degrees that contain work placements are better placed for employment when they graduate compared with students who lack this experience (e.g. Bowes and Harvey, 2000). The most valued skills for employers have been reported as research skills; ability to work in teams; and production of professional reports (Kemp et al., 2008). However, it is not always possible to squeeze a work placement into the formal programme of some degree subjects, especially if they are accredited by a professional body that requires core topics to be covered in a degree's syllabus. Modern academics therefore have to think of other ways to provide students with opportunities to develop their employabiity skills. Employability is commonly considered as a set of personal qualities (e.g. self-confidence, efficacy, reflectiveness, flexibility, international outlook), accomplishments, practitioner skills and understanding that make individuals more likely to secure employment and to be successful in their chosen occupations. Employability skills are becoming a vital yardstick for career success (Carbery and Garavan, 2005).

Academics at the University of Southampton's Centre for Environmental Science (CES) have addressed the challenge of simultaneously needing to develop students' employability skills whilst covering all the core topics required by professional bodies by deliberately incorporating them into a range of non-work placement modules. In particular, academics at the CES have, over a long period, developed,

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initiated, and delivered educational activities focused on waste and resource management that involve collaboration between university students and staff with external organisations. These activities have multiple aims, including:

- Generating new knowledge relating to case studies that exemplify the implications and impacts of waste-related research across a range of spatial scales;
- Providing students with real-world, in situ experiences as a means to enhance their skills with regard to problem-solving, sustainability, team-working, consultancy and employability;
- Providing mutual benefits to external organisations, universities and students, through sharing of resources to extend their value and impact.

Jensen et al. (2015) and Strayer (2012) have demonstrated that this type of approach - the incorporation of open-ended, real-world industrial examples into teaching and leaning strategies - acts to motivate students to produce work of a higher quality and depth than would normally be expected.

This paper outlines the educational approach taken and results achieved when under- and post-graduate students were tasked with working with a waste water treatment company (Southern Water; SW) with the aim of identifying and applying circular economy thinking to SW's operations at a major waste water treatment plant (WWTP) in England during October-December 2016.

### 1.1 The circular economy

The global economy has been built almost exclusively on the foundations of a linear model of extraction, production, consumption and dispose of as waste. The negative effects caused by this model are threatening the welfare of natural ecosystems and affecting the stability of the global raw materials market (Ghisellini et al., 2016). The acceleration of resource use globally, with many countries becoming more industrialised and with ongoing development of innovative technologies, is starting to threaten raw materials depletion.

The circular economy (CE) is based on a natural ecosystem concept, having a closed loop of material flow. The CE is an expansion of the waste hierarchy, whereby conventional waste streams that were often a cost to an organisation are viewed as source of resources and revenue, whilst minimising or even reversing their environmental impact. Adopting a CE will not only bring environmental benefits but could save the UK up to £700 million annually (Ellen MacArthur Foundation, 2017). Recovering resources from previously used materials will replace the need to extract virgin resources via mining, practices that are expensive (Mueller et al., 2017) and are associated with environmental impacts. The subsequent resource efficiency and environmental benefits that would follow with the adoption of a CE economy are significant (Curran and Williams, 2012). The European Commission has adopted an ambitious circular economy package, which contains proposals for legislation on waste to foster Europe's transition towards a circular economy (European Commission, 2015).

The ongoing challenges of non-renewable energy depletion and subsequent environmental pollution is encouraging businesses to look at wastewater as a resource due to the large potential for energy generation, material extraction and reuse. Considerable amounts of materials including metals, pharmaceuticals and nutrients enter WWTP and are either removed or lost in the effluent to the environment. The substances and materials, particularly nutrients and metals could theoretically be harvested from the wastewater and sold for reuse. There is also large potential for energy recovery through the capture of heat from wastewater and generation of biogas from treated sewage sludge.

In fact, wastewater treatment facilities have been considered at as a critical area for the implementation of CE thinking on the international stage. Karmenu Vella, EU Commissioner for the Environment, Maritime Affairs and Fisheries, said that "the greatest potential in relation to the circular economy is in the reuse of municipal wastewater". He goes on to say that it is "an economic opportunity that European Union companies could take up even more (Brockett, 2015)." It is clear that with international recognition, projects to bring about CE thinking in wastewater treatment will be supported and viewed as necessary in future years.

In the United Kingdom, there has been some small-scale adoption of CE processes in the wastewater sector (Table 1). However, the rate of uptake is limited. This project provided a novel opportunity to combine education and research activities via a preliminary study that aimed to inform Southern Water about the range of realistic, cost-effective options available with respect to the application of CE thinking to its operations.

### 1.2 Study location and characteristics

Southern Water is a private waste water treatment company based in the South of England with a water supply and treatment area of over 10,530 km$^2$. The company currently has 365 waste water treatment facilities in Hampshire, Kent, Sussex and the Isle of Wight, treating and recycling 718 million litres of waste water daily. The UK Environment Agency is SW's environment regulator and ensures that both UK and EU environment standards are met.

The studied WWTP (Millbrook) is owned by SW and located within the Western Docks in Southampton. The WWTP has undergone a number upgrades including a £20 million renovation in 1997 consisting of the enhancement of the anaerobic digester (AD) and sludge treatment to provide secondary treatment. Millbrook WWTP currently treats a mixture of sludge and wastewater from 250,000 people, nearly half of which is brought in from the region's smaller WWTPs. The facility consists of largely traditional wastewater treatment components including preliminary screening, primary treatment, nutrient removal and secondary treatment. It is designed to treat a full flow of 850 l/s before discharging into the River Test estuary. Approximately 14,000 t of sludge is converted into 10,000 t of bio solids soil enhancer each year via mesophilic anaerobic digestion with a hydraulic retention time of 15 days, and then sold to a variety of outlets for beneficial land use. Figure 1

**TABLE 1:** Examples of the current circular economy practices in operation or planning within wastewater treatment works nationally and globally.

| Recovery method | Water company | Status of operation | Companies involved | Potential | Reference |
|---|---|---|---|---|---|
| Co-digestion food waste | Thames Water | Full scale | EcoGenR8 | 40,000 t $yr^{-1}$ food waste, 13,000 MWh to the grid. | EcoGenR8 (2013) |
| Co-digestion food waste | Wessex Water | Full scale/ pilot | GENeco | 35,000 t $yr^{-1}$ food waste, 8,300 homes equivalent of methane. | Wessex Water (2015) |
| Latent heat from sewage | Scottish Water | Pilot | SHARC Energy Systems | 95% of heat requirement of a large campus site. | Scottish Water (2017) |
| ANAMOX and phosphate recovery | Severn Trent Water | Full scale | NMC Nomenca | PE > 650,000 treated, reduced energy costs (aeration) phosphate and ammonia recovered as struvite. Reduced maintainance costs ~£70,000 and 2 tonnes of struvite fertiliser produced daily. | North Midland Construction (2016) |
| Difgen – fresh water hydro power recovery | Scottish water | Water turbine recovering energy from freshwater flow control | Zeropex | £800k investment, returning £147k annually by producing 600 MWh $yr^{-1}$. | Scottish Water (2017) |
| Micro-algal biofuel | Aqualia (Spain) | Pilot | FP7 consortium | Recovery of $CO_2$ (from biogas) nutraceuticals (from the microalgae) removal of nutrients and production of biomass. | Maga (2016) |
| Phosphorus recovery | Edmonton WWTP (Canada) | Struvite precipitation and recovery | Ostara, Pearl | 2,000 tonnes of struvite produced annually. Reduction in maintenance costs | Linderholm et al (2012) |
| Energy recovery (FOG) | Thames Water | FOG recovery and energy production | 2OC | 19MW facility in East London, using FOG from households, businesses and industry with Thames water collecting and delivering the fat. | Power Technology (2017) |
| Carbon capture | NA | Carbon capture and algal biomass growth | Boots Ltd and PML | Capture exhaust carbon and use a feedstock for algal cultivation. | Levidow et al., (2014) |

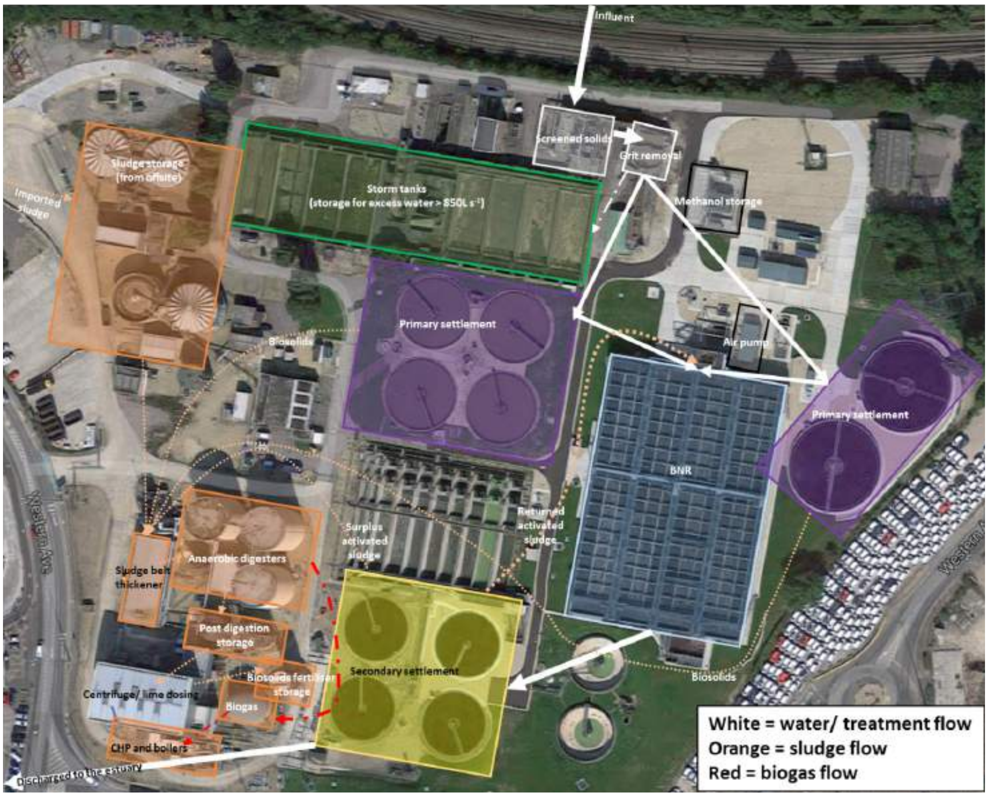


**FIGURE 1:** Aerial view of the Millbrook site detailing the flows of wastewater (white arrows), biosolids and activated sludge (orange dotted arrows), and biogas (red dotted arrow) between the labelled treatment areas.

provides a site map of the WWTP, highlighting:

- 6 primary settlement tanks
- 4 secondary settlement tanks
- 1 tertiary treatment flow system
- 3 anaerobic digesters
- 8 storm flow tanks
- 1 biogas storage silo.

Millbrook WWTP is typical of the UK evolution of wastewater treatment facilities and adjacent brownfield sites, with old, new and decommissioned infrastructure present. As this facility is located within one of the UK's busiest shipping ports there is limited room for expansion, and legacy piping for both the facility and dockyard reduce further the capacity for redevelopment.

## 2. METHODOLOGY

### 2.1 Students' Task

The task was set as part of a suite of assessments for the University of Southampton's module in "Sustainable Resource Management". This is an optional module available to students in the final year of a Bachelor's (BSc) degree and to students studying at Masters (MSc) level; these students are studying at levels 6 and 7, respectively, within the UK's Frameworks for Higher Education Qualifications (QAA, 2014).

SW tasked the university, through this module, to scope ideas to apply a CE approach to their entire wastewater operations. To maximise the impact and effectiveness of this research, the students were directed to work in parallel with the academics with focus on a single WWTP (section 1.2), thereby enabling students to join in with current research driven by industry. Prior to a site visit, students were extensively briefed on current wastewater technologies and operations, shown example data and a selection of CE concepts that had already been applied to SW operations. Examples of good research practice and a scaffolded schema were presented to the students to aide in their research development. A virtual initial tour of the site was delivered, along with a health and safety briefing. An extensive online repository of review papers, technical reports, images and commercial operational data was produced with students gaining access immediately. Students initially participated in an accompanied site visit to Millbrook WWTP to view the site and observe its operations in situ. They were expected to take their own notes during the site visit and were given an opportunity to discuss with SW representatives and ask questions; normal obligations associated with site visits and professional consultancy projects were followed. SW supplied several years of daily and hourly operational site data which was made available to the students. No further instructions, data or advice were directly given to the students unless explicitly asked, to which any outcomes were shared with all students promptly.

The students were tasked to identify and evaluate the major and minor waste streams within the WWTP process and produce options for their reduction, recovery and reuse without hindering the operational effectiveness of the site. They were subsequently required to produce a report that:

- Identified waste streams generated by the operations within and upstream of the Millbrook WWTP.
- Determined potential methods for reducing, recovering and/or processing selected waste materials using adaptions to the current systems deployed.
- Estimated the income/reduced costs of each recovery method.
- Identified and summarised processes and/or industrial networks that incorporate circular economy thinking within the WWTW setting that could be practically and realistically deployed by SW.
- Provided a priority list of 3 potential improvement projects, ranked by likely benefit (including economic, social, environmental, energy, efficiency, system, reputational, etc.).
- Provided a concise summary of how the findings could contribute to the adoption of circular economy operations throughout SW.

### 2.2 Collation of results and report generation

After submission, the students' work was marked with the methodologies, recovery techniques and relationships recorded. A comprehensive list was produced that complemented the work undertaken by staff at the university. Where CE ideas had already been identified by staff members, all new references, methodologies and equations were added. If ideas or techniques had not previously been identified or explored, then a more comprehensive investigation of the students' work was undertaken to check both the practicality and feasibility of the ideas. Regardless of their suitability, all CE ideas (once processed) were presented to SW, along with summaries of their suitability to allow SW to both approve ideas and act as a reference source for all potential CE approaches.

It should be noted that the emphasis of these student reports was to present all ideas, both "good" and "bad". The rationale for this was to explore an open-minded approach to the possibilities that could be available and are often overlooked by institutionalised experts. As the Masters students can come from a variety of backgrounds, reports were marked based on the student's research method and understanding of the direct/ indirect benefits of adopting a CE approach. Their direct understanding of the science and engineering principals in this instance was not directly marked, as this would give an unfair advantage to pure science and engineering graduates.

## 3. STUDENT REPORT OUTCOMES

In this section, examples of some of the key results generated for SW by this collaborative approach between staff and students are presented in order to illustrate the work undertaken. A confidential, commercial project report that incorporates some of these results was delivered to and approved by SW in early 2017. Within the reports submitted by the students, a broad range of ideas was generated. Cumulatively, the students produced an extensive list of ideas and the standard of work was high. The ideas produced included recovery of:

- Micro-plastics
- Nutraceuticals/ Pharmaceuticals
- Sewerage latent heat
- Fats, oils and grease (FOG)
- Metals
- Screened solids and grits
- Chemical nutrients (N, P, K)
- Organic materials
- $CO_2$
- Energy via anaerobic digestion
- Micro algae
- Exhaust heat
- Gravitational energy
- Waste water effluent
- Water effluent as a water source for crop production (aquaculture)
- Space for utilisation

In this section, four examples of some of the key results provided by a range of "good students", both from the environmental science pathway and engineering graduates are presented in order to illustrate the flow of work undertaken with the benefits to SW highlighted. The figures produced by the students are presented unedited, as is the text produced by the students, although mirror edits have been made to suit the required format for this publication. The results have been benchmarked with the literature and by SW with the exception of the economic costs, which are necessarily rough estimates.

### 3.1 Student A: Waste streams generated

The identification of waste streams and their current utilisation created a valuable starting point for further analysis of resource recovery. This also allowed SW to clearly identify all the resources that are currently underutilised within their operations. Student A reports a good example of the initial scoping of resources, determined by the site visit, discussions with SW staff and online resources:

"Table 2 provides a summary of all physical waste streams identified and indicates the presence of key contained resources which may prove a source of revenue generation through recovery and resale. Any current utilization of such materials is also detailed. As is evident from Table 2, Millbrook WWTP is currently operating significantly beneath its potential with respect to utilization of circular economy applications for the recovery of available resources. Though resource cycling is present, for example the recycling of sludge nutrients for use as soil enhancer, the variety of materials that are not currently utilized underlines the opportunity for further revenue generation through the application of circular economy principles".

**TABLE 2:** Summary of identified waste streams, their contained resources and current utilisation by Southern Water.

| Stage generated | Waste/resource stream | Contained resources | Current utilization |
|---|---|---|---|
| **Primary influent treatment** | Screenings | Plastics, textiles | Utilized - composted alongside grits in hot rot facility |
| | Grits | Minerals | Utilized - composted alongside screenings in hot rot facility |
| | | Organics | Utilized - composted alongside screenings in hot rot facility |
| | Sludge/biosolids | Nutrients: (N + P) | See anaerobic digestion -> digestate |
| | | Metals | See anaerobic digestion -> digestate |
| | | Organic C | Utilized - methanogen food source during anaerobic digestion |
| | FOG | Hydrocarbons | No dedicated utilization - co-digested with sludge |
| **Secondary influent treatment** | BNR products | Gaseous N | Not utilized - released to atmosphere |
| | Activated sludge | Denitrifying bacteria | Utilized - pumped upstream of BNR, recycling of bacteria |
| | | Nutrients: (N + P) | See anaerobic digestion -> digestate |
| | | Organic C | Utilized - methanogen food source during anaerobic digestion |
| **Anaerobic digestion** | Biogas | $CO_2$, $CH_4$ | Partly utilized:<br>• 90% utilized as fuel for CHP<br>• 0% wasted (flue) due to insufficient CHP capacity |
| | Digestate | Nutrients: (N + P) | See centrifugation -> De-watered digestate + reject water |
| | | Metals | See centrifugation -> De-watered digestate + reject water |
| **Centrifugation** | Reject water | Nutrients: (N + P) | Not utilized - re-enters WWT process |
| | | Dissolved metals | Not utilized - re-enters WWT process |
| | De-watered Digestate | Nutrients: (N + P) | Utilized - sold to agricultural industry as soil enhancer |
| | | Trace metals | Utilized - sold to agricultural industry as soil enhancer |
| **Post-treatment** | Treated effluent discharge | Dissolved metals | Not utilized - released to Solent (Strait of water on the UK South Coast) |
| | | Nutrients: (N + P) | Not utilized - released to Solent |
| | | Pharmaceuticals | Not utilized - released to Solent |
| | | Nutraceuticals | Not utilized - released to Solent |
| | | Microplastics | Not utilized - released to Solent |

### 3.2 Student B: Metal recovery

Metal recovery was identified by all students as a key area that could benefit from CE adoption with WWT. The breadth of recovery technologies explored was extensive, with both direct and indirect benefits explained, e.g. "the presence of toxic metals within biosolids soil enhancer and the benefits of their removal on both quality and health (Singh and Agrawal, 2008; Li et al., 2014; Nanchariah et al., 2015)". Table 3 highlights some of the key technologies explored by the students and synthesised further for delivery to SW. The example below is typical of the student's response, presenting the ideas available without the full technical understanding of where within the WWT process these technologies could be directly deployed. Students were not directly penalised, as it was the benefits of application with its potential and not the engineering that was assessed. The findings from Student B are outlined within Table 3.

### 3.3 Student C: Biogas for Transportation

Environmental impacts were a primary driver for the majority of students due to the nature of their degree pathways. Student C delivered a good example of a comparative analysis with estimations on both the environmental benefits and economic returns:

"To be suitable for use as a transport fuel, biogas must be upgraded to biomethane (Larsson et al., 2016), which involves removing trace gases and $CO_2$ offsite in an external production plant (Bates, 2015). Biomethane produced may be supplied as either Liquefied Biomethane (LBM) or Compressed Biomethane (CBM) (Bates, 2015); the fuel can be transported by road from the external production plant in pressurised containers to an onsite dispensing station at the WTTP. This provides the opportunity for onsite vehicles to be powered by on-site produced biogas.

An interesting adoption of biogas into transport is demonstrated by the 'POOBUS' operating in Bristol; it is a 40-seat Bio-bus which runs on biomethane generated from sewage and food waste (Geneco, 2015). It is estimated that a single passenger's annual food and sewage waste would fuel the bio-bus for 37miles and releases up to 30% less $CO_2$ (Geneco, 2015). This would provide an effective cascade system if bus fleets in Southampton were to adopt the same technology. It is reported that emissions associated with production, dispensing and use of biomethane in vehicles are 74% lower than conventional gaseous fuels as seen in Table 4.

Table 5 summarises the financial aspects of utilising biogas as a transport fuel.

### 3.4 Student D: Hydroelectric power generation

Below is an example of a student's fully explored adoption of a CE method into SW operations. Here the students have utilized the site visit, online resources and commercial data. Unlike the example above in 3.2, we have locations for potential sites with justifications. It should be noted that the identification of low head height energy recovery potential has now been raised within SW, with their own innovation team exploring this CE method further:

"The UK water treatment industry has voluntarily agreed a target of 20% renewable energy power consumption by 2020 (Environment Agency, 2009). The water flowing through large WWTWs provides a potential source of renewable energy that may be reclaimed and transformed

**TABLE 3:** Basic description of the process used in BES metal recovery (adapted from Wang and Ren, 2014)..

| Methods | Description |
|---|---|
| A | Metals such as Au (III), Cu (II) and Fe (III) which have a redox potential greater than the anode potential (-300mV), are reduced on abiotic cathode. Process allows the metals to be directly used by the electro accepter with no additional power supply needed. |
| B | Cd (II) Ni (II), Pb (II) and Zn (II), have lower redox potentials than the anode potentials. For them to be reduced, external power is needed drive the electrons from, the anode to the abiotic cathode. |
| C | Microbial reduction of metal oxides such as Cr (VI) on a bio-cathode. The metal recovery process involves dissimilatory metal reduction through using the metal as an external electro acceptor. Dissimilatory metal reducing bacteria include Trichococus pasteurii and Pseudomonas aeruginisa. |
| D | This stage is a combination of both stages B and C. Metal conversion using a bio-cathode which requires external power. Metal ions can be extracted from solutions and adsorbed onto biofilms on electrodes. Microorganisms that are present on the electrode reduce the metals during microbial respiration. |

**TABLE 4:** Greenhouse Gas savings from use of biomethane (adapted from Bates, 2015).

| | Biomethane (kg $CO_2$ eq/GJ) | Conventional gaseous fuels (kg $CO_2$ eq/GJ) | % saving in GHG emissions |
|---|---|---|---|
| CBM from AD | 18.5 | 68 | 74 |
| LBM from AD | 19 | 75 | 74 |

**Excluding emissions occurring during use of Bio methane in vehicles as these are vehicle dependent. They will however be identical from use of bio methane or conventional fossil fuel.*

**TABLE 5:** Financial considerations of utilising biogas as a transport fuel (adapted from Kollamthodi et al., 2016).

| Financial Biogas upgraded fuel | Production costs (Euros/GJ) | Total cost for delivery and dispensing (Euro/GJ) | Price (Euros/ GJ) |
|---|---|---|---|
| Compressed Biomethane | 7.28-10.20 | 2.77-5.72 | 5.7 |
| Liquefied Biomethane | 12.86-15.76 | 2.73 | 6.4 |

**note euro: European biogas market is the most developed.*

into electricity, this electricity can then be fed back into the system, or, sold to the national grid (Capua et al., 2014). The tanks and channels within a WWTW generally allow the implementation of hydropower technology (Berger et al., 2014); however, the main challenge associated with such installations is that the new utilities may interfere with the flow rates and effectiveness of the facility.

Hydropower systems operate with the use of a turbine, which is selected based on the flow rate or head (water pressure) of the system (Capua et al., 2014). Kinetic energy, in the form of falling water flows through the turbine, whereby it is converted into mechanical energy as the turbine spins. (Sektorov and Savvin, 1967). The most likely location for a hydropower turbine within the WWTW would be the outlet after the tertiary stage of treatment, this way, no processes would be affected and the flow rate would be adequately high. It would also have the compound advantage of not requiring screens and rubbish racks to protect the turbines, since the water would have already been treated further back in the process (Berger et al., 2014).

UK legislation currently indicates that planning permission must be sought from the local planning authority for the introduction of hydropower infrastructure to a business. Environmental permits will also need to be obtained from the Environment Agency to ensure that the utilities do not have a negative effect on the quality of the water effluent (Environment Agency, 2013).

The framework for this power estimation is based off of a 2013 scoping study into the feasibility of hydroelectric power in the Upper Blackstone Water Pollution Abatement District (UBWPAD) WWTW in New England, USA. This site was selected to substitute missing data from Millbrook because of the similarity of infrastructure, PE, catchment size and flow rates (Capua et al., 2014).

The equation used to determine power output from a hydropower turbine is:

$$P = \eta \rho w g Q H$$

Where:
$P$ = power (kW)
$\eta$ = efficiency of the turbine (unitless)
$\rho w$ = density of the water (kg/m$^3$)
$g$ = acceleration due to gravity (m/s$^2$)
$Q$ = flow of water through the turbine (m$^3$/s)
$H$ = head (m).

- For efficiency, it is assumed that with modern hydropower turbine 90% efficiency levels can be reached in accordance with estimations (Environment Agency, 2009)
- The density of water is 1000 kg/m$^3$
- Gravitational acceleration is a constant of 9.81 m/s$^2$
- The maximum flow of water through the turbine at Millbrook is 850 l/s which is equal to 8.5 m$^3$/s
- The head (difference in height between effluent output and the height of the River Test is unknown and therefore will be assumed to be the same as in UBWPAD at 1.7 metres.

Therefore, the potential power output per hour is:

$$P = \eta \rho w g Q H = (0.9)(1000\ kg/m^3)(9.81\ m/s^2)(0.85\ m^3/s)(3.6\ m) = 27.02\ kW$$

The potential variations in power output depending on flow are outlined in Table 6.

The most tried type of hydropower turbine in large WWTWs are micro turbines. Micro turbines only require a medium-sized flow rate and a small head size (2-12m) (Capua et al., 2014) and the model suggested is manufactured by Toshiba and installation costs range from £5,700 - £25,000 depending on the precise size of the blades and mechanisms required. The Toshiba model is appropriate for Millbrook because the flow rates required for power output (150l/s - 900l/s) fall within the ranges of the WWTW (150 l/s - 900 l/s).

In all cases, the most conservative figures are used to estimate costs and potential savings, unless otherwise specified.

Table 7 outlines the cost-benefit analysis of hydropower implementation at Millbrook".

**TABLE 6:** Variation in power output.

| | Low Flow | Peak Flow | Median Flow |
|---|---|---|---|
| Flow rate (m$^3$/s) | 0.13 m$^3$/s | 0.85 m$^3$/s | 0.49 m$^3$/s |
| Power produced by turbine (kWh) | 4.13 | 27.02 | 15.57 |

**TABLE 7:** Cost-benefit analysis of hydropower implementation at Millbrook.

| | Low Flow | Peak Flow | Median Flow |
|---|---|---|---|
| WWTW Energy Use/year (kWh) | 5,194,680 | 22,408,080 | 13,801,380 |
| Hydropower energy production per year (kWh) | 36,178 | 236,695 | 136,393 |
| % of total energy made by hydropower | 0.7% | 1.1% | 1.0% |
| Current buying price of national grid for electricity (£/kWh) | | £0.09648 | |
| Money made per year if sold to national grid | £3,490 | £22,836 | £13,159 |
| Installation costs | | £30,000 | |
| Buy-back time (median flow) | | 2.3 years | |

## 4. DISCUSSION

### 4.1 Student engagement

Students initially were unfamiliar with the open ended, broad ranging concept of this research and report writing style. Typically within an education setting assessed summative work follows an almost standard protocol, whereby the students are directly led to a fixed set of answers. As the staff setting the work were undertaking the research in parallel with the students, some students initially felt overwhelmed. In contrast, the majority of students began to excel and enjoy the freedom that this style of open research allowed, enabling them to utilize all the skills they have developed throughout their degree simultaneously. All students benefited from the scaffolding provided to support their research and analysis with >1,000 site visits by the students to their online teaching resources. Students that struggled with the concept benefited from direct mentoring and intervention to guide them through the process.

### 4.2 Improvements to SW's systems

The students offered a range of suggestions that could assist SW to improve its waste management practices. These varied from the adoption of currently available recovery technologies, to incorporating novel local relationships with the solid waste management sector and other industries.

The majority of ideas involved altering the current waste treatment processes used at the Millbrook site, with other suggestions of upstream energy recovery and FOG recovery/ prevention. The combined use of student and staff research enabled a wide spectrum of ideas to be explored, with benefits and limitations fully explored. Several of these ideas are being taken forward by SW for further investigation.

### 4.3 Benefits of collaboration

The benefits of this collaborative approach to SW include:

- Access to internationally recognized academics with specialist knowledge and skills.
- Access to resources such as professional/expert/research journals and specialist software that would not normally be available to them.
- Access to new ideas, concepts, fresh approaches and modern thinking, as well as a more international outlook.
- Access to an independent and highly skilled workforce that provides broad and deep expertise and skills that are not available within the organization and who are not influenced by commercial pressures or constrained thinking.

The outputs from the students' work fed into a professional consultancy report commissioned by SW from the University of Southampton. Upon delivering the report to SW's operations, innovation and management leaders they took a highly positive view of the work, its message and recommendations made. SW recognized the value of continuing work with the University of Southampton via their subsequent desire to participate in and fund follow-up activities, including academic consultancy, MSc and PhD projects.

From the university's perspective, there are also numerous benefits from collaborative working with a commercial organisation:

- Access to practicing professionals with a deep understanding of the practical, logistical, financial and political implications of project/policy implementation.
- Improved employer engagement and student employability profiles and access to high quality work placements.
- Contemporary views of workplace timescales and the financial and other constraints faced by large commercial organisations.

However, organizing and managing this type of collaborative activity through to a successful conclusion is not straightforward, as outlined in Williams and Shaw (2017).

### 4.4 Student performance and feedback on the module and assessment task

Overall, the consultancy-style reports submitted by the 42 students ranged from weak to excellent (mark range 12-93%; mean 61±20%) with 18 students achieving distinction (mark >70%) grades and 7 students failing (mark <40%).

We have found that this type of bipolar mark range is common when we set real-world assignments for students. We discussed the reasons for this in detail in Williams and Shaw (2017) and the same points apply in this case study.

A selection of typical (unedited examples) of verbatim feedback from the students to the staff are provided below. In addition, a number of students provided direct verbal feedback which indicated that the overall experience was very positive.

- "I enjoyed the general quality of the lectures and the challenging assignments."
- "I thought the coursework was a great way of improving my understanding of the circular economy through realistic application in the wastewater sector."
- "Challenging coursework – inspirational."
- "Coursework excellent for mirroring industry – good to talk about in interviews."
- "Coursework made us think more deeply about environmental problems."

A small number of students were unhappy with various aspects of the assignment, as illustrated by the unedited quotes below; note there are some contradictions:

- "The coursework was too large for its weighting which tended to increase stress on time management."
- "Coursework is too big to be 30% of one module."
- "Coursework felt irrelevant to the lecture content, would have preferred a visit and coursework based on household waste handling."
- "Coursework was, in my point of view, too challenging in the timeframe I had with other commitments but the coursework opportunity was very good but more time was needed."

- “We’re not too fussed about early deadline. Trust us to manage our time. Our fault if we get it wrong.”

The response of the module team to comments from the students secured halfway through the semester and well before the deadline for submission of the assignment is provided in Box 1. We have only provided our response to the “negative” comments from the students. We feel that our response was timely, professional and courteous. We accept that not all the students would like or have the maturity to accept or appreciate our firm but direct and evidence-based rebuttal of some of the comments provided.

## 5. SUMMARY AND CONCLUSIONS

This paper has outlined how academics can address the challenge of simultaneously needing to develop students’ employability skills whilst covering core topics required by professional bodies by deliberately incorporating open-ended, real-world industrial activities into teaching and learning activities within assessed modules. It describes and critically evaluates the approaches taken by students undertaking waste-focused activities that involve collaboration with an external organisation. It outlines the results of the scoping reports, by cost benefit analysis, as well as discussing the environmental benefits, utilisation of industrial relationships, potential improvements and limitations of the approaches/processes selected. The benefits to SW, the students and the academics are discussed, together with the outcomes from this activity.

The report arising from this task and associated research has shown clear areas where SW can improve on its wastewater management strategy to reduce their environmental impact, recover nutrients and materials as well as lowering costs and producing new revenue streams. This could be achieved primarily by implementing upstream recovery of nutrients and heat energy which would have the combined effect of producing a revenue stream from the sale of a novel fertiliser, reduce maintenance costs associated with struvite precipitation and help displace fossil fuel derived fertilisers and energy. If implemented, SW could see large economic savings in the operation of their sludge treatment facilities. For example, SW could liaise with Southampton City Council and their local waste management contractor to collect food waste separately and to investigate co-digesting food waste on sludge treatment sites, reducing costs for the council, recovering energy and biomass, and producing a revenue stream for SW.

Targets for increasing nutrient and material recovery, energy usage reduction and energy production could be set and audits carried out periodically to check progress against these targets. Landfill and incineration cannot be seen as zero waste concepts and SW can help other companies in order to comply with the Waste Framework Directive (2008/98/EC) by the co-utilisation of their sludge treatment facilities for organic waste treatment.

There is no doubt that this student-led learning activity stimulated interest, discussion and debate and generally raised both students’ and SW’s employees’ awareness of CE issues. In an educational context, there is considerable merit in prompting action learning of this ilk. Although students’ levels of achievement and performance were highly varied, there is little doubt that this task stimulated independent learning and development of professional and employability skills. From the perspective of SW, the students’ activities and reports generated suggestions and recommendations that may not have been obvious means to achieve steps towards CE thinking at SW, and may not have otherwise been forthcoming.

As well as describing and evaluating the activity, this paper has showcased some of the learning materials developed, reported on the practical and logistical issues encountered, summarized results from the different activities, evaluated feedback from the students and the commercial organisation, and highlighted potential future developments.

**BOX 1:** Verbatim response of the module team to students’ comments about the assignment (provided halfway through the Semester).

*“The comments generally suggest that the coursework is too hard, too complex, contains too much science and engineering and that the deadline is too tight. We are obliged to set challenging coursework at this advanced level of study. We think the coursework mimics exactly what a waste manager / consultant would be required to do by a client and hence we believe that the assignment prepares you well for the “world of work.” We think you will start to appreciate the skills and attributes you have developed by tackling this assignment when you’ve had your feedback, mid-semester pressure has waned and you are feeling more reflective.*

*We thought we had made it clear in the module’s paperwork and via lectures that students could contact us at any time to discuss any aspects of the module. It is pleasing that a large number of students have emailed us to ask a question or to make an appointment to discuss questions relating to the assignment. Students also often use the break between lecture slots to ask questions. We have responded to and met with everyone who has made a request and answered their questions or provided clarifications. We would encourage any students with further questions to contact us immediately – we will be happy to help.*

*There are a couple of comments about the coursework hand-in date. We gave your assignments out in Week 1, Lecture 1. We outlined the assignments in the lecture and we explained in detail why we set the coursework deadline towards the latter end of November. It is normal for students to be feeling work pressure in mid-semester. The truth of the matter is that no matter when we set the deadline, someone would complain! We set it early so you have the choice when to start your work, according to your own personal schedule.*

*The coursework weighting is “tried and tested” so we feel it is appropriate. We try and get the right balance between natural sciences, maths and aspects of engineering, social, economic, operational, practical and political issues – Environmental Science is a “broad church” and we need to maintain this tradition on ENVS modules. A second site visit is not available because the site operator cannot schedule it. We’d like to respectfully point out that the coursework is NOT about wastewater – it is about the circular economy and how a waste that has traditionally been normally disposed of by “dumping” – raw sewage – could be turned into resources (including energy, nutrients, aggregates, other raw materials, etc etc).*

*Thank you for taking the time to provide us with this very useful feedback. We will reflect on all your comments again at the end of the Semester and make changes as necessary.”*

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# RELEVANCE OF SELECTED MEASURES IN TRANSITION TO A CIRCULAR ECONOMY: THE CASE OF THE CZECH REPUBLIC

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**ABSTRACT**
To accelerate the transition to a circular economy European member states have applied a broad range of policy instruments. Based on recycling rates and public participation in recycling efforts the Czech Republic would also appear to be on the way to a circular economy. However, the environmental effect of policy instruments (e.g. promotion and education campaigns) will soon be exhausted, as recycling rates are no longer rising significantly. Convinced recyclers are already carrying out effective recycling; however, how to motivate chronic non-recyclers to increase their public participation in recycling efforts is questionable. This paper examines the impact of social norms and the social environment in an attempt to explain recycling behaviour. Furthermore, awareness of waste separation options and programs is viewed as an important variable that determines how non-recyclers perform with regard to recycling.

## 1. INTRODUCTION

Undoubtedly, a significant driver of the contemporary environmental policy is the circular economy paradigm. Circular economy, unlike an economy based on linear material flows, is aimed at reducing the need for raw materials and waste disposal (Bilitewski, 2012; Elia et al., 2017), or maintaining the added value in products for as long as possible, and thus minimising waste production (Di Maio et al., 2017). Therefore, the circular economy paradigm is a combination of ecological, economic, technological, and social issues. As such, circular economy has gained increasing attention amongst scholars, policymakers, and industry representatives in recent decades (Geissdoerfer et al., 2017).

The implementation of the circular economy paradigm raises the question as to which are the priority fields of intervention. Based on the current state of knowledge, a circular economy constitutes an innovative business model with a focus on new approaches to product design and production (Mathews and Tan, 2011; Bocken et al., 2016; Ramani, 2010; Tukker, 2015). Haas et al. (2015) stressed the necessity of innovative high-quality recycling technologies. Furthermore, effective cooperation between key actors on the supply chain is crucial in order to reap the benefits of a circular economy (Desrochers, 2004; Chertow, 2007; Lehtoranta et al., 2011; Martin et al., 2015).

The intention to close the loop of product lifecycles through increased recycling and re-use cannot be achieved in the absence of well-designed incentives aimed at both the production sector (Hagelüken et al., 2016) and consumers, with regard to increasing the rates of recycling of municipal waste or to achieve a zero-waste target (Ghisellini et al., 2016).

How to convince residents to participate in recycling programs represents one of the main tasks of the waste management policy. People's willingness to recycle different materials is influenced by perceived convenience of separation, such as a short distance to waste bins (Domina and Koch, 2002; Hage et al., 2009; Ando and Gosselin, 2005; Mueller, 2013; Struk, 2017), ease of access of drop-off centres (Derksen and Gatrell, 1993; Domina and Koch, 2002; González-Torre and Adenso-Díaz, 2005), or space needed for the storage of recyclable materials at home (Ando and Gosselin, 2000; Bernstad et al., 2013). Furthermore, the perceived convenience is also dependent on efforts needed to recycle (e.g. time) (Garces et al., 2002), or cost spent on recycling activities (Ewing, 2001). Miafodzyeva and Brandt (2013) also indicate other variables including frequency of collection, technical mismatches, cleanliness of recycling sites, handling problems, or design of collection points as those that determine the convenience of recycling efforts.

However, convenience that depends on technical or or-

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ganisational conditions for separation reflects only a limited part of the story of consumers' recycling behaviour. According to Miafodzyeva and Brandt (2013), strong predictors of recycling behaviour are typically moral norms as well as available information and environmental concern. Other non-pecuniary predictors include warm-glow effect as individual's satisfaction from a pro-environmental action (Halvorsen, 2008) and social norms as shared perceptions of appropriate behaviour in a society (Abbott et al., 2013). According to Farrow et al. (2017), social norms via social interactions affect the behaviour of an individual in many different areas of pro-environmental actions. Some scholars even refer to social norms as possible solutions for many environmental problems (Nyborg et al., 2016).

In our study, we focus on the role of social norms and their influence on the recycling behaviour of consumers in greater depth. The Czech Republic may be perceived as a country with a mature recycling system as the convenience of recycling (in terms of distance and availability) has already reached a high level. A certain part of society however is still not willing to take part in recycling efforts. We will refer to these people as "chronic non-recyclers" and will seek to uncover the reasons for their non-recycling behaviour with a focus on their social environment and information available. Our aim is to identify ways of motivating chronic non-recyclers to participate in waste separation and recycling.

We analysed a sample of 1611 cases representing the Czech adult population using relevant statistical methods such as frequency analysis, reliability checks, and ordered logit model. Data originated from extensive research studies conducted during the year 2017.

First, the current knowledge of social norms and social environment (Section 2), and of awareness of recycling (Section 3) is illustrated, continuing with a description of the Czech waste separation system (Section 4) and defining the methods used in our analysis (Section 5). In the last part, we present the results obtained and discuss them in detail (Section 6).

## 2. SOCIAL NORMS AND SOCIAL ENVIRONMENT

Recycling has been becoming a routine behaviour for individuals and a social norm following decades of promotion and education (P&E) campaigns aimed at increasing convenience and removing obstacles perceived by people when using separation systems. Nyborg et al. (2016) defined social norms as "a predominant behavioural pattern within a group, supported by a shared understanding of acceptable actions and sustained through social interactions within that group". However, Hage et al. (2009) argues that it is difficult to distinguish between social and moral norms as these social interactions activate moral norms. Defining social norms, the pressure of the community and potential sanctions are significant aspects in shaping behavioural patterns.

Hage et al. (2009) considers social norms as norms enforced by sanctions from others. According to Halvorsen (2010), when the social norm is strong, sanctions (and feelings of guilt) are significant predictors of pro-environmental behaviour. However, Abbott et al. (2013) stated that sanctions are not required, when: "social norms become internalised so that they do not require an external sanction mechanism or … the degree of conformity amongst the population and the level of expectation are sufficiently high for compliance without the need for the threat of external sanctions". Furthermore, Benabou and Tirole (2006) found that rewards and punishments aimed at supporting desirable behaviour produced perverse effects when intrinsic motivations were crowded out by extrinsic incentives. Thøgersen (2008) argued that when individuals identify the social norm as legitimate on their own, and not due to threats of sanctions, they will not attempt to evade.

Hage et al. (2009) mentioned social norms as a predictor of recycling behaviour particularly in situations in which recycling is a publicly visible activity and individuals face community pressure (e.g. neighbours, friends, colleagues). If recycling is not a visible activity (and no community expectations concerning the behaviour arise), moral norms have higher importance in predicting recycling behaviour (Hage et al., 2009). According to Abbott et al. (2013), kerbside collection of recyclables is a visible action apparent to peers that build a positive self-image.

Thanks to intrinsic motivation, i.e. moral norms or other internal variables (Barr, 2007; Saphores and Nixon, 2014) people recycle waste even in systems lacking the extrinsic motivation stimuli such as charges or fees. Nevertheless, part of the population maintains that it is normal not to recycle (so called 'chronic non-recyclers', or 'reluctant recyclers'). According to Thomas and Sharp (2013), some society fractions may hold even non-recycling norms. Based on the UK, reported recycling rates remain rather low, particularly among younger people aged between 18 and 24, in lower social classes, and among those living in flats or terraced housing. Their decision not to recycle is based on a relatively lower concern for waste separation in comparison with other social issues, inconvenience, or on lack of information relating to waste separation (Pocock et al., 2008). According to Yau (2010), the difficulties in a high-rise setting arise as a consequence of collective action problem typical for anonymous actors. However, when economic incentives (such as different types of rewards) are introduced, the motivation effect can be maintained.

Answering the question of how to engage the 'chronic non-recyclers' or 'reluctant recyclers' (label used by Brekke et al., 2007) in waste separation may be challenging. It is particularly difficult when the waste separation system already reflects the needs and expectations of households. These systems are highly mature and this attribute influences its performance (Miliute-Plepiene et al., 2016).

Above all, the social pressure on the individual's intention to recycle is based on a sense of community, or on socioeconomic status of the neighbourhoods (Kurz et al., 2007). Social approval is important for an individual. Blasch and Ohndorf (2015) describe social approval as a source of private utility – a kind of immaterial reward that individuals receive when they conform to social norms. These results correspond with the research of Vicente and Reis (2008), Halvorsen (2010), or White and Hyde (2012). According to these authors, the way in which individual behaviour is perceived by others, activates emotional reactions – e.g. bad reputation in the community increases a feeling of guilt and vice versa. If the individual expects recycling to help him gain social approval in the community, then the warm glow effect from this activity will increase.

The importance of social acceptance provided by family members, friends, or neighbours was confirmed by Vining and Ebreo (1990), Oskamp et al. (1991, Ewing (2001) and Bruvoll et al. (2002). Social pressure imposed by other family members (especially children) plays a special role in influencing attitudes towards recycling (Meneses and Palacio, 2005; Thomas and Sharp, 2013). As reported by Vicente and Reis (2008), children are not able merely to alert household members that recycling is worthwhile: their attitudes also represent an investment in the future. As adults, they will be responsible for implementing recycling patterns in their own households. Ewing (2008) added that approximately 50% of households reported how the opinion of other household members was highly appreciated and considered.

Aronson et al. (2005) described other important reasons producing a significant influence of social group on the behaviour of an individual - social norms are crucial in shaping the behaviour of people who are uncertain about what decision to take. Individuals believe that others are better informed and are therefore inclined to behave in the same way as their peers – these peers help individuals identify the solution to their uncertainty.

Abbott et al. (2013) reported that the advantage of social norms consists in the function of community as a monitoring and enforcement unit without governmental intervention if social norms are sufficiently activated. In this respect, public representatives might adapt measures aimed at activation of recycling efforts via activation of social norms together with implementation of social control opportunities, e.g. kerbside collection scheme.

## 3. WASTE SEPARATION AWARENESS

The willingness to separate waste is also influenced by knowledge of separate collection and recycling system. P&E campaigns may provide arguments supporting waste separation and initiate a two-step flow of communication aimed at encouraging people to discuss recycling amongst themselves. This is essential as current data show that opinion-leaders of chronic non-recyclers, people who are closest to the respondent, are usually opponents of waste separation, rather than supporters. P&E campaigns can play an important role in bringing the issue of recycling and its positive impact on the environment to the attention of opinion-leaders, who can consequently influence the attitudes and behaviour of other people in their social surroundings. However, Halvorsen (2010) concluded that the effectiveness of P&E campaigns has reached its limits as this measure has been used for a long time in the majority of countries and already reached a large part of society. To motivate those lacking motivation to recycle may prove difficult. On the other hand, to prevent recycling decay (decrease of public participation on recycling efforts) Woodard et al. (2005) recommended long-term P&E campaigns.

The more information people have regarding recycling issues (i.e. placement of collection points, information about materials that can be separated or collection times and frequency), the more likely they are to participate in waste separation (Hornik et al., 1995; Garces et al., 2002; McDonald and Oates, 2003; Barr, 2007). Furthermore, De Young (1989), McDonald and Oates (2003) and Ewing (2001) identified significant lack of knowledge as the main obstacle to public participation in recycling. The amount of information about recycling has been identified as a difference between recyclers and non-recyclers (MacDonald and Oates, 2003; Oskamp et al., 1991). However, Vining and Ebreo (1990) noted that the hypothesis of a correlation between knowledge and attitudes (or behaviour) was rejected by some authors.

The amount of available information affects not only the propensity to recycle, but also attitudes towards recycling. Without correct information about recycling, it becomes more difficult to participate in the recycling schemes (Alexander et al., 2009). According to Barr (2007), when speaking about knowledge, it is necessary to differentiate between abstract knowledge (i.e. general knowledge about the environment and overall awareness of environmental issues) and instrumental knowledge (especially the awareness about what, where and how to recycle). In this respect, instrumental knowledge was found to be a significant driver of behavioural change (Hornik et al., 1995; Shaw et al., 2006; Barr, 2007). To increase instrumental knowledge, education of what, where and how to recycle is desirable (Chen and Tung, 2010). P&E campaigns should focus on both instrumental and abstract knowledge (Lakhan, 2014).

To equip non-recyclers with information, either direct (leaflets, doorstep campaigns) or indirect (mass media) communication channels can be used. Some authors argue that direct communication is more effective (Vicente and Reis, 2008). Bernstad (2014) found written information to be ineffective in increasing the separation rate of food waste, as the knowledge of the receiver was overestimated; language difficulties have arisen, and timing of the P&E campaign and ambiguity of the message delivered have been indicated as potential issues. Concerning the effectiveness of oral communication – e.g. using doorstep campaigns, the results obtained are ambiguous. While Dai et al. (2015) or Bernstad et al. (2013) reported how doorstepping intervention led to a statistically significant increase of food waste diversion, Alexander et al. (2009) found doorstepping as ineffective in increasing the set-out rate. Alexander et al. (2009) considered doorstepping the best approach mainly in areas where the population was 'hard-to-reach' (e.g. block of flats).

Garces et al. (2002) saw P&E campaigns as key strategies of local representatives aimed at increasing public participation in recycling efforts. Meneses and Palacio (2005), or Timlett and Williams (2008) emphasised the educational system as a significant driving force in supporting recycling efforts. Woodard et al. (2005) or Shaw et al. (2006) found P&E campaigns important when separation systems alter and new schemes are established. Although education seems to be a significant determinant in shaping recycling behaviour, von Borgstede and Anderson (2010) stated that the lack of formal education is not a barrier to information attention.

P&E campaigns should be targeted at specific social groups. According to Chen and Tung (2010), education should be targeted towards children. Meneses and Palacio (2005) concluded that marketing activities should concentrate on the middle-aged (46-60) with primary education level. Miliute-Plepiene et al. (2016) stated that P&E campaigns should focus on promoting recycling efforts amongst those close to householders (e.g. neighbours, relatives, or friends), thus making recycling more visible in the social environment, particularly in early-stage recycling schemes.

## 4. SPECIFICS OF THE CZECH WAY OF SEPARATE COLLECTION

Pilot projects of the separate collection of recyclables in the Czech Republic were introduced in 1997 as the packaging recovery organisation EKO-KOM was established. To raise recycling and participation rates, policy-makers at national (Ministry of the Environment) and local (municipal representatives) levels used primarily non-monetary incentives such as P&E campaigns. As a legal obligation, municipal waste management ordinances mandated households to separate their waste, although this obligation was not regularly controlled and enforced.

P&E campaigns in the Czech Republic cover the entire range of communication channels, including mass media (TV, radio) and direct media (billboards, websites, etc.). On a local level, educational and informational activities for children and adults have been implemented. In 2003, the nationwide P&E campaign began using the slogan “Don’t be lazy: separate waste”, to gently skewer all sorts of excuses for why people do not recycle. In 2009, a new slogan was introduced, “It’s meaningful: separate waste”, focused specifically on non-recyclers featuring a series of reasons for avoiding recycling.

Not only non-monetary incentives such as P&E campaigns, but also monetary incentives are applied to increase public participation in recycling schemes. Monetary incentives (e.g. unit-based fees) have been introduced in almost 20% municipalities, which are not only effective (Sauer et al., 2008; Slavik and Pavel, 2013), but also positively influence administrative costs and total costs of the municipal waste management system (Slavik and Pavel, 2013). Unfortunately, unit-based fees are perceived as more exacting from an organizational point of view, and therefore, municipalities prefer fixed fees without any motivation (Slavik et al., 2009). Another monetary incentive - deposits - has only limited effectiveness as they are applied only on refillable glass bottles.

From an organisational point of view, the Czech municipal waste management system is based on kerbside collection of municipal solid waste and a relatively dense system of collection points (drop-off centres) for recyclables (paper and cardboard, glass, plastics, and beverage cartons). As the effectiveness of separate collection in the Czech Republic is highly influenced by the convenience of the system (expressed by the distance between households and collection points) (Struk, 2017), the increasing number of collection points (expressed also by decreasing proximity of containers for recyclables) is one of the key measures in increasing public participation in waste separation (see Table 1).

The effectiveness of separation (see Table 1) represents the amount of separately collected recyclables (paper and cardboard, glass, plastics, cartons) in terms of container-based collection (drop-off centres), and collection in bags. Municipal systems of separate collection are supplemented also by seasonal mobile collection for biowaste and bulky waste. Furthermore, some municipalities operate recycling centres where people can deposit hazardous waste or recyclables (usually without payment).

## 5. METHOD USED IN STATISTICAL ANALYSIS

To assess the statistical significance of our predictors, we used ordinal regression analysis – ordered logit model. We used this model for an ordinal character of predicted variables that provide information on ranking, but not the distance between the different categories. This approach of analysis resembles statistical methods used elsewhere, e.g. in Hage et al. (2009).

### 5.1 Variables in the model

In our research, we tested the impact of selected factors on waste separation declared by households. Self-reported recycling behaviour as the dependent variable is a common approach to study recycling behaviour; the same approach can be found in Derksen and Gartrell, 1993 or Miliute-Plepiene et al., 2016. However, we are aware of the limitations of this approach expressed by Bernstad et al. (2013), when not all households claiming to recycle actually do so regularly, or fail to recycle all materials. This overestimation of self-reported (declared) recycling is caused by a social desirability effect. Since data relating to actual recycling rates are not available, we focused on self-reported (declared) recycling.

In our study, all dependent constructs were operationalized using the answers to two statements for each construct. The respondent´s agreement with each statement was measured on a four or seven point Likert scale. The tested model was based on six predictors operationalized using responses to twelve statements (Figure 1). This figure also provides information about wording of the statements that appeared in the survey instrument (the questionnaire).

**TABLE 1:** Development of selected waste separation outcomes in the Czech Republic.

| | 2006 (*2008) | 2012 | 2016 |
|---|---|---|---|
| Effectiveness of separation (Kg/inhabitant * year) | 27.9 | 39.1 | 44.8 |
| Number of containers (Pcs) | 146131 | 229000 | 307000 |
| Proximity of containers (m) | 115 | 102 | 96 |

*Source: Grolmus (2009), Grolmus (2013), EKO-KOM (2017)*

The first hypothesis tested in our model is as follows:

H1: A higher level of waste separation awareness has a positive impact on declared waste separation.

In this respect, we distinguished two types of knowledge – general knowledge and instrumental knowledge (Barr, 2007). General knowledge comprises overall information about the waste management system and its usefulness, and about communication reminding individuals to separate their waste. This information may be an important waste separation driver as it enables comprehension of waste separation tasks and provides individuals with key information relating to the importance and impacts of their efforts. Instrumental knowledge covers practical information relating to performing waste separation under the given circumstances. Therefore, this information focuses on types of materials and packaging collected separately, on materials collected within the specific area, on location of the nearest containers etc.

Our second hypothesis reads as follows:

H2: There is a statistical significant relationship between social environment of an individual and his/her declared waste separation.

We entered four constructs into our model – attitudes of others, activity in discussions, behaviour of primary social group, and perceived social control. By variable "attitudes of others", we tested whether or not the opinion of family members, relatives, colleagues or friends (those we considered as a primary social group) influences respondents. In other words, we tested to which extent respondents display the same attitudes towards waste separation and recycling as those close to them. Moreover, we considered not only the opinion of others about recycling but also their behaviour, which respondents were able to recognise. Therefore, we included the reflection of relevant behaviour of primary social group into our model. We also hypothesised that discussions about waste separation might indicate the presence of the topic in everyday life of respondents, and therefore the taking place of similar discussions might increase waste separation efforts (this effect reflects the variable "activity in discussions"). Finally, our model involved perceived social control hypothesised as a predictor of waste separation behaviour, due to the fact that perceived social control imposes pressure on individuals to behave in a desired way.

As mentioned above, all original statements from the questionnaire were transformed into constructs suitable for regression analysis. Each construct (except for activity in discussions) was composed of three levels reflecting the intensity of the phenomenon. Knowledge (both general and instrumental) was then distinguished as high, medium or low; attitudes of others appeared as positive (i.e. in favour of recycling), neutral or negative; behaviour of others varied from being engaged in waste separation through being neutral, to rejecting waste separation, with perceived social control being high, limited or low. Activity in discussions was binary (high–low). Table 2 provides an overview of ordinal level of constructs.

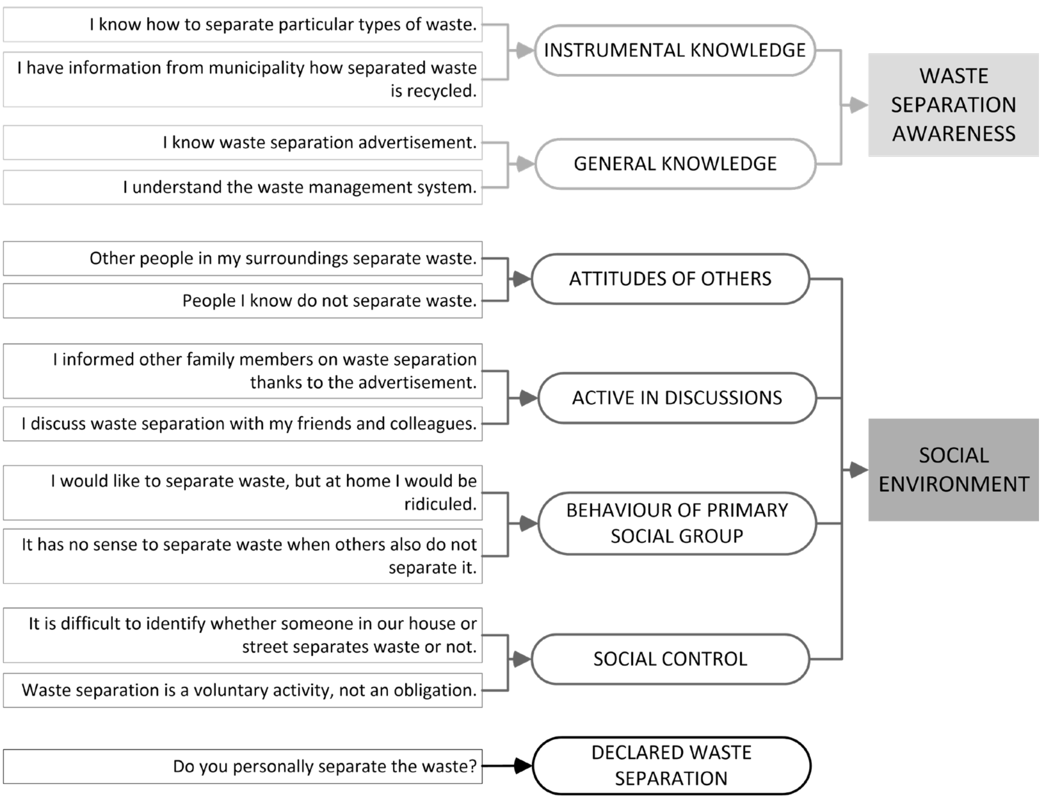


**FIGURE 1:** Wording of statements indicating key variables.

**TABLE 2:** Summary statistics of the key variables in the model.

| Variables | Coding | Mean | Std. Deviation |
|---|---|---|---|
| DEPENDENT VARIABLE | | | |
| Attitude towards waste separation | 1 = engaged | 1.53 | 0.700 |
| | 2 = neutral | | |
| | 3 = rejecting | | |
| DEPENDENT VARIABLES (PREDICTORS) | | | |
| Waste separation awareness | | | |
| Instrumental knowledge | 1 = high | 1.66 | 0.604 |
| | 2 = medium | | |
| | 3 = low | | |
| General knowledge | 1 = high | 2.27 | 0.625 |
| | 2 = medium | | |
| | 3 = low | | |
| Social environment | | | |
| Attitudes of others | 1 = positive | 1.98 | 0.598 |
| | 2 = neutral | | |
| | 3 = negative | | |
| Active in discussions | 1 = high | 1.86 | 0.348 |
| | 2 = low | | |
| Behaviour of primary social group | 1 = engaged | 1.19 | 0,457 |
| | 2 = neutral | | |
| | 3 = rejecting | | |
| Social control | 1 = high | 2.30 | 0.602 |
| | 2 = limited | | |
| | 3 = low | | |

### 5.2 Sample, sampling technique and data collection method

The target population was the general population of the Czech Republic aged 18 – 74 years comprising only Czech residents living permanently in the Czech Republic.

The sampling technique applied was the multistage random procedure using random route. Since no adequate sampling frame (register or list of inhabitants) was available, with respect to the method of data collection (face-to-face interviews) when random digit dialling could not be used, primary sampling units were selected. Subsequently, within primary sampling units, addresses were identified and households were selected. Finally, the interviewers visited the pre-selected addresses, attempted to contact the pre-selected households, identified the prospective respondent using the Kish-table (Kish, 1949) and invited the relevant individual to participate. Altogether 186 primary sampling units throughout the Czech Republic were selected; within each primary sampling unit a maximum of 20 addresses were identified. Interviewers contacted 3148 households and performed 1611 interviews (response-rate was 51.2%). However, due to incompleteness of some of these interviews, when respondents refused to provide key socio-demographic data, the database comprised a total of 1579 cases used for analysis.

Fieldwork took place during March 2017. Average duration of an interview was approx. 35 minutes; interviews were focused on waste separation solely. From a total of 1611 interviews, 20% were supervised (by check-backs) and verified in terms of compliance with ethical standards (e.g. confidentiality, informed consent).

## 6. RESULTS AND DISCUSSION

Table 3 provides the results of ordered logit model. Statistically significant variables comprise instrumental knowledge, general knowledge, attitudes of others, activity in discussions, and behaviour of social group. This supports both of our hypotheses that waste separation awareness, as well as social environment of an individual, are relevant predictors of waste separation behaviour. It is obvious that instrumental knowledge is a stronger predictor of declared waste separation than general knowledge. At the same time, the data indicate a strong effect of enabling environment (behaviour of primary group) and overall responsiveness (activity in discussions and attitudes of others). The only statistically insignificant variable in the model is perceived social control.

In the case of social groups, respondents share their attitudes toward waste separation with other members of their primary social group (particularly household or family). In accordance with Abbott et al. (2013) and Miliute-Plepiene et al. (2016), we also found that people

**TABLE 3:** Results of ordered logit model.

| Parameter estimates | Estimate | Wald |
|---|---|---|
| Threshold (constant) | | |
| Attitude towards waste separation | | |
| 1 = engaged | -3.407*** | 77.395 |
| 2 = neutral | -0.91** | 6.046 |
| 3 = rejecting | 0 | . |
| Location (predictors) | | |
| Instrumental knowledge | | |
| 1 = high | **-0.912*****| 16.104 |
| 2 = medium | -0.175 | 0.682 |
| 3 = low | 0 | . |
| General knowledge | | |
| 1 = high | **-0.576**** | 5.508 |
| 2 = medium | **-0.364***** | 9.019 |
| 3 = low | 0 | . |
| Attitudes of others | | |
| 1 = positive | **-1.029***** | 23.674 |
| 2 = neutral | **-0.422***** | 7.895 |
| 3 = negative | 0 | . |
| Active in discussions | | |
| 1 = high | **-0.941***** | 19.426 |
| 2 = low | 0 | . |
| Behaviour of primary social group | | |
| 1 = engaged | **-2.908***** | 77.77 |
| 2 = neutral | -0.271 | 0.646 |
| 3 = rejecting | 0 | . |
| Perceived social control | | |
| 1 = high | 0.159 | 0.506 |
| 2 = limited | 0.038 | 0.105 |
| 3 = low | 0 | . |

Note 1: Number of observations = 1573;
2 Log Likelihood = 592.414***
Chi-Square (Pearson) = 320.742
Nagelkerke = 0.402

Note 2: Threshold in the model corresponds to constant in linear regression models, location is the component that includes the predictors, estimates are the regression coefficients, and Wald is the parametric statistical test that measures the statistical significance of the coefficients.

Note 3: Parameters set to zero are redundant.
*** = significant at 1% level;
** = significant at 5% level;
* = significant at 10% level

who separate waste are surrounded by others who act in a similar way, whereas those who do not separate waste are surrounded by people who do likewise. This creates the basis for social determination of recycling.

In this respect, it should be mentioned that primary social groups (particularly families) are coherent in their approach to waste separation, as shown in Table 4. The majority of household-members either separate waste (see 38 percent who separate continuously and 31% who do so at least occasionally) or do not separate it at all, as is the case for 18% of respondents. Despite such strong coherence when 87% of households behave the same way, in a small number of cases the efforts of an individual may differ from the behaviour of other household members. In 4% of cases the respondent refers only occasional separation, although the household as a whole accounts for continual recyclers. Similarly, in 1% of households the respondent does not separate waste at all, in spite of the fact that other members separate at least occasionally. In 5% of households, the respondent separates continuously, whereas other members do so only occasionally, and in 2% the respondent refers an at least occasional separation, although the rest of the household does not separate at all.

Our results confirmed the importance of family members, and other close relatives in shaping attitudes towards recycling (Thomas and Sharp, 2013; Meneses and Palacio, 2005). Therefore, P&E campaigns should be targeted at those who live in close contact, i.e. relatives in the same household (compare with Miliute-Plepiene et al., 2016). However, these P&E initiatives are effective only when adequate recycling infrastructures (i.e. technical and organisational conditions) are available (Lakhan, 2014).

In accordance with Hage et al. (2009), perceived social control has a statistically insignificant impact on the decision of individuals to separate waste or not. This may be related to local institutional conditions. In our research, respondents found it difficult to identify those who separated or failed to separate waste in their house, block of flats or street (see Table 5).

Another explanation why social control only produces a low impact on waste separation is that waste separation is perceived as a voluntary activity, as a demonstration of willingness to do something for the environment or to satisfy an individual intrinsic motivation (see Table 6). The majority of people in the Czech Republic are not aware that waste separation is legally enforced by law (§ 17 of the Czech Waste Act No. 185/2001 Coll. states that individuals and other waste producers are obliged to separate waste). However, neither law nor social norms sanction non-recycling

**TABLE 4:** Relationship of waste separation of the respondent and the whole household.

| | | Waste separation by the respondents | | | |
|---|---|---|---|---|---|
| | | Separate continuously | Separate occasionally | Do not separate at all | Total |
| **Waste separation by anybody in the household** | Separate continuously | **38%** | 4% | 0% | 43% |
| | Separate occasionally | 5% | **31%** | 1% | 38% |
| | Do not separate at all | 0% | 2% | 18% | 20% |
| | Total | 44% | 37% | 19% | 100% |

**TABLE 5:** Results of questions focused on the opportunity for social control.

| It is difficult to identify whether or not someone separates waste in our house, block of flats or street. | |
|---|---|
| | Valid percentage |
| 7 = Definitely agree | 15% |
| 6 | 19% |
| 5 | 28% |
| 4 = Neither, nor | 22% |
| 3 | 9% |
| 2 | 3% |
| 1 = Definitely disagree | 5% |
| Total | **100%** |
| *Mean = 4.79; standard deviation = 1.540; missing cases = 90* | |

**TABLE 6:** Results of questions focused on the opportunity for social control.

| Waste separation is a voluntary activity, not an obligation. | |
|---|---|
| | Valid percentage |
| 1 = Definitely agree | 33% |
| 2 = Agree | 42% |
| 3 = Disagree | 17% |
| 4 = Definitely disagree | 8% |
| Total | 100% |
| *Mean = 3.00; standard deviation = 0.908; missing cases = 34* | |

behaviours (for further discussion, see Abbott et al., 2013).

The absence of sanctions, and consequently a lack of feelings of guilt (Halvorsen, 2010) might contribute towards decreasing the perceived importance of waste separation and recycling behaviour. However, sanctions based on social control are not necessarily a prerequisite of recycling behaviours. As Abbott et al. (2013) reported, when social norms are internalized by individuals, sanctions are not necessary. Furthermore, Benabou and Tirole (2006) warn against the perverse effects of sanctions when intrinsic motivations may be crowded-out by extrinsic ones.

Another explanation is related to the system of waste separation applied in the Czech Republic. Although container-based separate collection prevails, containers are not located primarily at public places where social control is possible. The location of containers does not reflect perceived social control argumentation, but rather other factors such as, for example, the proximity of containers from homes, containers must be located on publicly owned land and accessible to collection vehicles etc. Therefore, the strength of social control is limited by technical and organisational conditions and the role of moral norms increase (Hage et al., 2009). According to Abbott et al. (2013) who call for an increase in recycling efforts through the activation of social norms, together with the creation of opportunities for social control, we would also like to emphasise the suitable location of collection points. Kerbside collection does not seem to be an adequate solution, as this collection system is based on collection points.

As mentioned above, general and instrumental knowledge are statistically significant predictors of waste separation in line with the results summarized by Miafodzyeva and Brandt (2013). P&E campaigns seem therefore to be a good way of increasing waste separation efforts. However, as Miliute-Plepiene et al. (2016) reported, P&E campaigns are less effective in systems in which technical infrastructures enhance the carrying out of recycling without perceived barriers.

Targeting P&E campaigns is not an easy task, particularly when non-recyclers represent a heterogeneous segment of the population. Indeed, differences were observed between the two groups in terms of gender (when males dominate within non-recyclers), social status (higher share of lower social groups among non-recyclers) and housing type correlated with social group (Table 7). Another significant difference in composition of both groups is based on age; there is a concomitant higher share of younger respondents (particularly singles) and older respondents (living alone) amongst the non-recyclers. Therefore, it is difficult to deliver specific information directly to the relevant individuals through use of nationwide media or similar communication tools.

Another challenge encountered in attempting to further strengthen the waste separation effort is that the demand for information differs between recyclers and non-recyclers. Recyclers are interested in information such as how the waste is processed, what products could be produced from recycled material, where new containers will be placed, or what specific waste fractions can be separated. The wish-list of non-recyclers is however different; they prefer information on overall benefits gained by sorting waste, general information about the waste management systems as such, how much waste separation costs, who benefits from the system and what the perspective of the waste separation is (Barr, 2007). Starr and Nicolson (2015) reported how P&E campaigns are effective when the under-informed population is targeted. However, non-recyclers cannot be compared to an under-informed population, as their information demands are different. Thus, Lakhan (2014) pointed out that P&E campaigns should provide not only abstract, but also instrumental, knowledge. The importance of well-targeted P&E campaigns arises when the waste separation systems alter, or new systems are established (Shaw et al., 2006). P&E campaigns are an effective instrument when the aim is to increase public involvement in recycling (Read, 1999), but as Woodard et al. (2005) added, P&E campaigns need to run continuously to avoid drop-off of public participation.

Data relating to public participation in collection systems and the trend of recycling rates over the past few years in the Czech Republic (EKO-KOM, 2017) indicate that extensive growth of separated waste amounts resulting from an improved participation rate is reaching its limits. Recycling efforts are based on engagement of those who are willing to recycle. Further increase in effectiveness of recycling would therefore imply persuading the 'non-recyclers'. However, to achieve this by means of the traditional communication tools such as P&E campaigns, may prove expensive and inefficient. In this respect, new media and

**TABLE 7:** Socio-demographic variables and their relation to waste separation behaviour.

| | | Respondents' waste separation behaviour | |
|---|---|---|---|
| | | Recyclers | Non-recyclers |
| Gender | Male | 48% | 57% |
| | Female | 52% | 43% |
| | Total | 100% (N=1276) | 100% (N=303) |
| Age | Less than 20 years | 5% | 7% |
| | 20–29 years | 15% | 19% |
| | 30–39 years | 20% | 21% |
| | 40–49 years | 19% | 16% |
| | 50–59 years | 18% | 11% |
| | 60 years or older | 22% | 27% |
| | Total | 100% (N=1276) | 100% (N=303) |
| Social status | A (high class) | 6% | 2% |
| | B (higher middle class) | 21% | 10% |
| | C (lower middle class) | 41% | 39% |
| | D (lower class) | 19% | 30% |
| | E (underclass) | 13% | 20% |
| Housing type | Family house | 44% | 32% |
| | Block of flats | 56% | 68% |
| | Total | 100% (N=1271) | 100% (N=303) |

innovative below-the-line communication tools focusing on the social environment of individuals might play a crucial role in influencing behavioural patterns.

## 7. CONCLUSIONS

A whole range of policy measures (or instruments) is available with the aim of engaging people in systems of municipal waste separation and thus, achieving a circular economy. These measures may be both direct (affecting the decision-making of individuals, i.e. financial incentives) and indirect (e.g. P&E campaigns). In addition to these policy-driven (intentional) measures, spontaneous social processes that might affect the intention of individuals to behave in certain are also implemented. These spontaneous processes include the behaviour of other people reflected by an individual, declared behaviour of primary social group (family), participating in discussions about waste separation and social control.

The results obtained in our study confirmed that awareness of waste separation, together with the social environment of an individual are significant predictors of waste separation behaviour. On the other hand, perceived social control proved to be statistically insignificant in justifying recycling behaviour. These results dictate the need for adequate policy measures to increase public participation, particularly amongst those who are not willing to participate in waste separation. Of the currently available policy instruments, soft measures such as P&E campaigns seem to produce an adequate impact on recycling behaviour when the separation systems can count on adequate infrastructures.

However, targeting P&E campaigns is not an easy task when non-recyclers represent a heterogeneous segment of the population. Furthermore, the demand for information differs between recyclers and non-recyclers. Whereas recyclers are interested in information such as how the waste is processed, what products could be produced from the recycled material, where new containers will be placed, and what can be separated, the demands of non-recyclers is largely different. They prefer information relating to the overall benefits to be gained from waste separation, general and introductory information on waste management systems as such, information on the costs of waste separation, and the future of waste separation.

The influence of perceived social control on recycling efforts seems to be limited (at least under the examined Czech conditions). Individuals did not find it easy to identify those who separated or failed to separate waste in their house or street, with waste separation being perceived as a voluntary activity, demonstration of an effort to do something for the environment or to satisfy an individual intrinsic motivation, rather than an obligation. To increase social control, the organisation of separate collection would need to be rearranged.


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# END OF USE TEXTILES: GIFTING AND GIVING IN RELATION TO SOCIETAL AND SITUATIONAL FACTORS

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**ABSTRACT**
The clothing and fashion industry is associated with the seeking of new trends to meet and influence consumer demands. In consequence, the rates at which clothing and other textiles are purchased are high, as are the associated rates at which end-of-use items arise. Ensuring that methods and systems are in place to permit and encourage items deemed to be end-of-use by one person to be utilised to their full potential by other(s) is clearly desirable. This study aimed to elucidate how societal and situational factors influence the purchasing of clothing and other textiles, how decisions are made regarding end-of-use of these items, and the routes and means by which end-of-use textiles are subsequently passed on or disposed of. A comparison was therefore made of the public in Southampton (UK; relatively high income and established waste management systems) and Dakar (Senegal; relatively low income and with largely informal waste management systems) in which societal and situational factors contrast. Comparison of these two case studies was thus expected to provide insight as to the influence(s) of society and situation upon the generation and fate of end-of-use textiles. Through a questionnaire survey, the study found that factors leading to purchasing, decisions regarding end-of-use of items and post-use destinations differed markedly between these two contrasting cities. However, reuse of end-of-use clothing and textiles was common in both cities, which is desirable in reference to the aims and principles of the waste hierarchy. High levels of reuse occur despite the common belief that more developed and established systems provide better opportunities for effective waste and resource management.

## 1. INTRODUCTION

Whilst efforts to achieve more sustainable use of many materials through waste management have been made, the textiles sector has received relatively little attention. The global retail textile industry reached a value of over $1.2 trillion in 2014 (Resta and Dotti, 2015). As one of the world's largest industries, this sector impacts substantially upon the environment (Resta and Dotti, 2015) due to, for example, production processes that use a range of chemical and consume high levels of water and energy (Caniato et al., 2012).

Production, consumption and disposal rates of textiles are considerable. In the UK alone around 2.5 million tonnes of textile items are bought annually (Letcher and Vallero, 2011); over a million tonnes per year of textiles from the domestic waste stream may be destined for landfill (WRAP, 2013). Rapid rates of purchasing lead to high rates of disposal of unwanted fashion products (Letcher and Vallero, 2011). Further expansion of the fashion industry due to growing consumer demand (Pookulangara and Shepard, 2013) could lead to higher levels of consumption, with associated increases in the generation of textile waste (Birtwistle and Moore, 2007).

Textile waste generation is linked to economic growth and lifestyle aspirations. With growing population and rapid economic growth in developing economies, the middle income sector of the population has grown in number and is associated with a commensurate increase in product consumption (Pookulangara and Shepard, 2013). Historically, "make-do-and-mend" has been a way of life in less economically developed nations (Williams and Shaw, 2017). However, materialism and social comparison have gained high social importance in developing economies; individuals compare their possessions with those of others (Zhang and Kim, 2013). Buying luxury and new products appears to be an indicator of success. Individuals in emerging economies may seek to differentiate themselves from others and tend to "mimic" so-called Western attitudes (Markus and Kitayama, 1991). In contrast, as the circular economy gains

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traction in developed countries, reuse, refurbishment and repair of "pre-loved" or "pre-owned" items is increasingly becoming a lifestyle choice for many people (Williams and Shaw, 2017), especially a group of the so-called "Millennials" who are seeking a lifestyle that is sustainable, flexible, fair and tolerant (Students and Staff of the Centre for Environmental Science, 2017).

Rising levels of demand will likely lead to higher quantities of unwanted textiles. If unwanted items are destined for an unsustainable fate (e.g. landfill), the impacts will be (1) the resources consumed in producing textiles will be of limited value; (2) the utility of these items will be foreshortened; and (3) disposal, if inappropriate, will lead to impacts on the environment. The risks of these negative impacts occurring is higher if patterns of development exceed parallel development of waste management systems that reflect and respond to the local setting (Mukhtar et al., 2016). Whilst social, cultural, behavioural and situational factors are key drivers behind waste generation attitudes (Martin et al., 2006), such factors vary markedly across nations of contrasting economic status. It can be argued that published research has focused somewhat more upon developed nations than on emerging economies.

The waste management systems and infrastructure that have been established in developed economies do not currently exist in many emerging economies. The methods that deliver desired outcomes in such situations are by no means guaranteed to achieve their purpose in less developed economies without adequate recognition of and adaptation to the local situation, including social, cultural, and behavioural facets. By the same token, practices that have evolved in emerging economies in the absence of established infrastructure and services are likely more orientated around behaviour and thus may provide insight to waste management system design and delivery where infrastructure and services are fully established.

The aims of this study are therefore to:

- Elucidate factors that influence the generation and fate of end-of-use textiles in contrasting situations;
- Determine if and how improvements might be made in the management of textile waste adapting or adopting approaches from contrasting situations.

## 2. METHODS

A comparison was made of the public's self-reported actions in Southampton (UK) and Dakar (Senegal), cities in which societal and situational factors contrast markedly. Southampton has a population with relatively high income; the city has a long-established waste management infrastructure and services (since 1753), including systems provided by municipal authorities and third sector organisations (TSOs; charities and community groups). In contrast, Dakar's population has a relatively low income and its waste management infrastructure and services are largely informal and irregular. Informal networks provide opportunities for the giving of items to others, a practice that is aligned with religious and cultural norms. Formal charity schemes for clothing and textiles are present but at low scale (Ville de Dakar, 2015a). The differing cultural and situational contexts of these two cities permit insight to their influences upon clothing and textiles in terms of consumption, end-of-use decisions and the fate of unwanted items and, in so doing, identify opportunities for improving waste management for clothing and textiles.

A questionnaire survey (including quantitative and qualitative elements) was employed to elucidate societal and situational influences on buying habits, decisions regarding the end-of-use of items owned, and disposal/reuse options. Surveys of the public were conducted in situ in Southampton and Dakar.

### 2.1 Case studies

#### *2.1.1 Southampton*

Southampton is located in Hampshire, in the south of the United Kingdom and has a population of ca. 250,000 (HCC, 2015). As a result of the UK's long history of development, infrastructure and services (Timlett and Williams, 2011), municipal waste management systems in Southampton are well-developed and well-established. Property-close collection systems, household waste recycling centres and recycling banks are established and the public are accustomed to using them. Other, complementary, systems also exist in the UK contribute to achievement of waste management outcomes that align with the principles of the waste hierarchy (Williams, 2015). Around 25% of textile items arising from the municipal solid waste stream is recycled by charity-related companies such as the Salvation Army Trading Company Limited who provide collection and distribution facilities for donated shoes and clothes (Woolridge et al., 2006). TSOs are important in textile waste management strategies in the UK (Brooks, 2013). Often working with local authorities – as in Southampton – TSOs provide re-use and donation pathways to encourage and facilitiate local communities to participate (Woolridge et al., 2006). Textile waste is mainly given to charities or destined for donation pathways in the UK and it then follows varied routes (DEFRA, 2016).

#### *2.1.2 Dakar*

Dakar is Senegal's capital and one of the main seaports on the western African coast. It is located in the Cape Verde Peninsula and is Africa's most westerly point. The Dakar region is the most populated in Senegal with ca. 3,140,000 inhabitants (Ville de Dakar, 2015a; ANSD, 2015). Few formal charity schemes such as 'Le Relais' or 'Frip Ethique' (organized by Oxfam) exist in Dakar (Ville de Dakar, 2015b). Infrastructure and services for waste management are mostly irregular and informal. Despite a substantial lack of environmental consciousness and resources, informal collection of end-of-use items takes place due to religious and cultural norms. Strong implementation of Islamic belief and societal values encourage the use of re-use and donation schemes according to the principle that 'the more you give, the more you will get rewards from God'. Textile waste is therefore commonly given by households to family members or local mosques to re-distribute to lower-incomes or 'talibés' (child beggars).

## 2.2 Questionnaire surveys and data analysis

After initial piloting and revision, surveys were conducted using a self-completed questionnaire comprising closed (e.g. Likert format) and open questions. Questions addressed attitudes towards textile purchasing habits, textile waste generation, end-of-use destination for textiles, and impacts of textile waste on the environment. The questionnaire was translated into French and Wolof, the two main languages spoken in Dakar. Questionnaire surveys were carried out during May to July 2016 and were completed by respondents either in digital format or face-to-face interviews and paper-based recording methods in public spaces. The online questionnaire was distributed through social media (Facebook & Twitter) that were exclusive to either Southampton or Dakar. Conducting face-to-face questionnaire surveys in both cities avoided bias towards individuals with internet access. Opportunities were taken to conduct surveys of the public in busy city locations to maximise the number of responses, and gather responses from range of individuals that was diverse in terms of age and income level.

All questionnaire data were coded for storage and analysis in the IBM SPSS Statistics package Statistical Package for Social Sciences (SPSS) computer programme. Descriptive statistics and statistical tests such as regression were used. Qualitative information gathered in questionnaire surveys was grouped into umbrella responses (Fink, 2003; Pope et al., 2006).

# 3. RESULTS

## 3.1 Questionnaire respondent profiles

When compared with the population profiles of Southampton and Dakar (Table 1), the respondent groups were broadly similar. Some differences were noted between the respondent groups and the population profiles (Table 1); where pertinent, bias in the respondent group profile will be considered in evaluating the observed responses.

## 3.2 Clothing and textile purchasing

When asked how often they purchased new clothing and textile items, respondents in Southampton reported that purchasing occurred at higher frequency than respondents in Dakar (Table 2). Although the differences in purchasing frequency were statistically significant (Table 2), it was noted that the modal frequency for purchasing clothing and textiles were once in 1-2 months at both locations.

Some of the stated reasons for buying new clothing or textiles differed between the two study locations. Fashion trends and public social events were more common reasons for purchases in Dakar than in Southampton (Table 2).

A significantly higher proportion of respondents indicated that purchases were motivated by need in Southampton than in Dakar (Table 2). Likewise, a significantly higher proportion of respondents indicated that purchases were for gifts in Southampton than in Dakar (Table 2).

**TABLE 1:** Profiles of respondent groups compared with reference population statistics. Authoritative data specifically for Dakar were not available; reference statistics are derived from national data for Senegal.

| Location | Variable | Respondents | Population | Notes |
|---|---|---|---|---|
| Southampton | Population | 356 | 249,500[1] | |
| Dakar | | 414 | 3,137,196[2] | |
| Southampton | Female | 57% | 50% | Gender bias in favour of female questionnaire respondents compared with population statistics[1,3,4] |
| | Male | 43% | 50% | |
| Dakar | Female | 54% | 50% | |
| | Male | 46% | 50% | |
| Southampton | Age 18-25 | 18% | 13% | Bias towards the two lower age classes for both Southampton[4] and Dakar[3]. No respondents from the >66 group in Dakar. |
| | Age 26-45 | 47% | 30% | |
| | Age 46-65 | 29% | 21% | |
| | Age >66 | 5% | 13% | |
| Dakar | Age 18-25 | 36% | 9% | |
| | Age 26-45 | 57% | 24% | |
| | Age 46-65 | 7% | 13% | |
| | Age >66 | 0% | 3.5% | |
| Southampton[5] | Muslim | 2% | 4% | Closer match of respondent group and population for Dakar than for Southampton |
| | Christian | 22% | 52% | |
| | None | 57% | 34% | |
| Dakar[6] | Muslim | 92% | 94% | |
| | Christian | 4% | 5% | |
| | None | 4% | - | |

*[1] HCC (2015), [2] Ville de Dakar (2015a) and ANSD (2015), [3] ANSD (2013), [4] ONS (2015), [5] ONS (2011) [6] ANSD (2006)*

**TABLE 2:** Purchasing frequency and reasons for purchasing reported for buying clothes and textiles in Southampton and Dakar.

| Question | Responses | Southampton (%) | Dakar (%) |
|---|---|---|---|
| How often do you buy clothes and other textiles items for your own use? | Every week or two [1] | 19 | 14 |
| | Once in 1-2 months [1] | 53 | 41 |
| | Once in 3-6 months [1] | 17 | 9 |
| | Once or twice a year [1] | 7 | 3 |
| | Depends on finding or need [2] | 4 | 30 |
| What is the main reason you buy clothes and other textiles items? | Fashion trend [2] | 10 | 26 |
| | Change of size [3] | 9 | 7 |
| | Only buy what I need [1] | 58 | 44 |
| | Public social events [2] | 0.8 | 17 |
| | Private social events [3] | 5 | 4 |
| | Gifts [1] | 14 | 0.7 |

[1] proportion for Southampton is significantly higher than for Dakar (one-tailed z test); [2] proportion for Dakar is significantly higher than for Southampton (one-tailed z test); [3] no significant difference between Southampton and Dakar (one-tailed z test).

### 3.3 End-of-use decisions for clothing and textiles

When asked to state the main reason why clothing and textiles are thought to be no longer needed, respondents in Southampton and Dakar differed. In Southampton, the main factors were when clothing and textiles wore out or became out-of-date, and when the individual's clothing size changed (Table 3). In Dakar, respondents' end-of–use decisions were most frequently motivated by the needs of others (Table 3). Having too many items, having too little space, changing tastes, and a desire for new items were more common reasons for end-of-use decisions in Dakar than in Southampton (Table 3).

### 3.4 End-of-use destinations of clothing and textiles

When asked to state the main destinations of end-of-use clothing and textiles, responses were markedly different between the two locations. In Southampton, the vast majority of respondents (89%) indicated that, when no longer needed or wanted, clothing and textiles were donated to a charity (Table 4). Charity donations were also identified by 30% of the respondents in Dakar, but the main recipients of end-of-life clothing and textiles in Dakar were friends and family (39% of respondents) and homeless people (25% of respondents) (Table 4). The proportion of respondents disposing of end-of-use items in a general refuse bin was significantly higher (5%) in Southampton than in Dakar (1%) (Table 4).

## 4. DISCUSSION

With regard to the factors that influence the generation and fate of end-of-use textiles in Southampton and Dakar, this study has revealed differences and similarities between respondents in these contrasting locations and situations. In terms of purchasing habits, the tendency of respondents in Southampton to purchase clothing and textiles more frequently than in Dakar (Table 2) is, arguably, symptomatic of the high levels of consumer demand (Pookulangara and Shepard, 2013) and could lead to high quantities of unwanted items (Letcher and Vallero, 2011). It was noted, however, that there was a strong desire for individuals to purchase new clothes and textiles in both relatively wealthy and relatively low income cities (Table 2). The modal reason for buying new items was also the same in both locations: "only buying what is needed" featured

**TABLE 3:** Reasons reported for deciding that clothes and other textiles are no longer needed in Southampton and Dakar.

| Responses | Southampton (%) | Dakar (%) |
|---|---|---|
| Too many [2] | 1 | 6 |
| To make space [2] | 1 | 4 |
| Change of taste [2] | 4 | 10 |
| Worn out/spoiled [1] | 52 | 9 |
| Change of size [1] | 14 | 19 |
| Old, out of date [1] | 24 | 9 |
| Want new [2] | 0.6 | 12 |
| Change of lifestyle [3] | 0.3 | 0.7 |
| Others may need it [2] | 0 | 26 |
| My education [2] | 0 | 2 |

[1] proportion for Southampton is significantly higher than for Dakar (one-tailed z test); [2] proportion for Dakar is significantly higher than for Southampton (one-tailed z test); [3] no significant difference between proportions for Southampton and Dakar (one-tailed z test).

**TABLE 4:** Primary destinations of end-of-use clothes and textiles reported in Southampton and Dakar.

| Responses | Southampton (%) | Dakar (%) |
|---|---|---|
| Charity/donation [1] | 89 | 30 |
| Homeless [†2] | 0.3 | 25 |
| Mosques/Churches [2] | 0 | 2.9 |
| Family & friends [2] | 3.9 | 39 |
| General refuse bin [1] | 4.8 | 1.4 |
| Griots [††3] | N/A | 0.7 |

[†] Includes talibés (usually young abandoned or orphaned boys; Dakar only: [††] storytellers and praise singers who generate income from contributions to celebrations; Dakar only). [1] proportion for Southampton is significantly higher than for Dakar (one-tailed z test); [2] proportion for Dakar is significantly higher than for Southampton (one-tailed z test); [3] not tested statistically. N/A not applicable.

strongly in both Southampton and Dakar (Table 2). In relation to the waste hierarchy (Williams, 2015), "only buying what is needed" aligns, in principle, with the avoidance or prevention of waste that is of highest preference. At the same time, an individual's differentiation between what is "needed" and what is "wanted" is subjective, being by no means consistent with the perspectives of others. Of the respondents in Southampton, for example, 38% indicated that they only buy what they need and stated that they buy something new every 1-2 months. The inference here is that ca. 6 to 12 new items of clothing or textiles are "needed" per year by these individuals; we suggest that this rate is symptomatic of buying habits that are driven at least partially by "want" as well as "need".

It was also observed that higher proportions of respondents in Dakar than in Southampton indicated fashion trends and public social events are motivations for buying new items (Table 2). A view may be taken that dressing for fashion and public events has an orientation towards the appearance of individuals and their associated concerns: the focus on personal appearance in public fora highlights the importance of social comparison (Zhang and Kim, 2013) and the desire of individuals to i) differentiate themselves from others (Markus and Kitayama, 1991) and ii) seek a sustainable lifestyle (Students and Staff of the Centre for Environmental Science, 2017).

When respondents were asked to state reasons for deciding that clothing and textiles had reached their end-of-use, responses differed markedly between Dakar and Southampton. Although the proportions of respondents stating various reasons for this decision were statistically different in several instances (Table 3), two key diffrences were particularly notable. First, the predominant reason for end-of-use decisions amongst respondents in Southampton were motivated by utility; over half of the respondents identified that their clothing or textiles were no longer wanted or needed due to a perception that they were worn out or spoiled (Table 3). In terms of resource efficiency, there is a positive message in that this response infers that individuals are using clothing and textiles fully in regard to their potential utility, a feature that corresponds with the principles of sustainable waste and resource management. However, the judgement regarding when an item loses its utility is again subjective; what constitutes "worn out" or "spoiled" (Table 3) is questionable and likely inconsistent.

In contrast, a key feature of the surveys in Dakar was the relatively high proportion of respondents whose end-of-use decisions were influenced by the potential utility of items by others (26% of respondents). These responses could be interpreted as indicating that (1) many individuals in Southampton and Dakar orientate end-of-use decisions around the utility of clothing and textiles, but (2) around half of the respondents in Southampton consider the utility to themselves whilst around half of the respondents in Dakar consider the utility to others. This higher apparent level of altruism in Dakar may be associated with societal norms, aligned with cultural practices and expections, and faith.

In terms of the destinations of end-of-use clothing and textiles (Table 4), it was notable that very few respondents indicted that end-of-use items were disposed of (Table 4). Given the contrasts in the situations in which Southampton and Dakar residents exist (see §2.1), this observation is highly notable: 95% to 99% of the respondents indicated that their unwanted items would be reused in some shape or form (Table 4). In terms of the destinations of these items, when they are no longer wanted or needed they are more likely to be given to a charity in Southampton than in Dakar and more likely to be given directly to a person or community-level organisation in Dakar than in Southampton. This situation likely reflects both the availability of charity collections (i.e. a situational factor) and the difference in cultural and societal norms. The intention that end-of-use items are passed on so that they can be used by others infers that the (stated) practice in both locations aligns well with the principles of the waste hierarchy insofar that reuse is preferable to recycling, value recovery (EfW) or disposal. Achievement of waste hierarchy aims is, however, incomplete. The reasons stated why choices are made to buy new clothing and textiles (Table 2) are partly aligned with loss of functionaility and partly by other factors not strictly aligned with function or loss thereof. The implication is that avoidance or prevention of waste has been partially but not yet fully reached.

With regard to if and how improvements might be made in the management of textile waste, the outcomes of this study highlight the role of altruism; end-of-use decisions were made in both locations to the benefit of others. Respondents in Dakar appear to favour "gifting" by which a donation of end-of-use items is made to a recipient via relatively direct means (Burke et al., 1978); the majority (89%) respondents in Southampton chose primarily to give their end-of-use items to charities. The distinction in this regard is that "gifting" results in directly tangible and/or visible outcomes, whereas the act of "giving" involves indirect connections between the donor and the beneficiary, with an associated lack of direct visibility of outcomes. Gifting has been mostly associated with collectivist societies (Cruz-Cárdenas et al., 2016) and implies giving back to people who are close at hand, whilst gifting (i.e. donating) is mostly used in individualistic societies (Brooks, 2013) and implies giving items to an organization.

The dominant pathways of end-of-use clothing and textiles in Southampton and Dakar (Figure 1) share common outcomes but contrast in terms of the means by which the benefits arise. Clothing and textiles that are no longer needed or wanted by one person tend to be given away such that they can be used, i.e. their utility is extended. Environmental benefits are thus gained: the resources consumed in and impacts of production (Caniato et al., 2012) are used to fuller extent and there is no short-term impact through disposal. There is a societal benefit in that the clothing and textiles are made available to others with needs. In the Southampton case, relatively low-cost items may be purchased by those with limited income and/or those who make a lifestyle choice to purchase used items (Williams and Shaw 2017). In the Dakar case, items are made available to low income and/or homeless individuals at no cost (Figure 1). There is a contrast, however, in that selling donated (gifted) items through charity shops provides financial resources to support charitable work, the beneficiaries of which may be proximate or distant to the donor.

Cruz-Cárdenas (2013) and Cruz-Cárdenas et al. (2016) suggested that collectivist societies prefer giving to others rather than disposal, whereas individualistic societies prefer donation to charity schemes. The present study reaffirms this proposal and presents futher evidence that gifting is an important part of material culture in collectivist societies (Brooks, 2013) and infers that the Southampton and Dakar case studies represent, broadly, more individualistic and collectivist cultures. Moreover, this study demonstrates the routes by which the donation of clothing or textiles can positively affect the donor via the "warm fuzzy feeling" associated with acts of goodwill and with benefits for both the environment and the well-being of others (Figure 1).

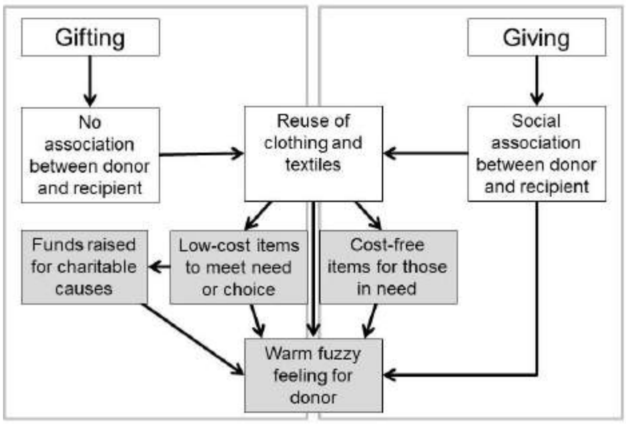


**FIGURE 1:** Schematic representation of the predominant pathways of end-of-use clothing and textiles and the benefits of their reuse. "Gifting" is more prevalent in Southampton; "giving" is more prevalent in Dakar (see Table 4).

Regarding if and how improvements might be made in the management of textile waste adapting or adopting approaches from contrasting situations, this study has highlighted the differences between broadly collective and individualistic societies (Figure 1). Although the reuse of end-of-use items (e.g. Table 4) is preferable in relation to the aims and principles of the waste hierarchy (Williams and Shaw, 2017), the opportunities to raise funds for charitable works appear less well represented in Dakar, whilst the association(s) between the donor and the beneficiary are weaker in Southampton. We propose that fund-raising in Dakar would likely be inappropriate given that "giving" occurs at already high rates (Table 4) notwithstanding the lack of infrastructure available in this regard.

Strengthening the relationship between the donor and the benefits of their donations may, however, present an opportunity to improve reuse rates in Southampton and perhaps reduce consumption rates of clothing and textiles. Clothing and textile donors in Southampton must understand the indirect means by which their donations make a difference to others, whilst the benefits of donations in Dakar are readily and directly perceived (Figure 1). Opportunities to achieve more direct and visible connection are perhaps more suitable for charities with UK if not local focus. Charities supporting the homeless, for example, have opportunities to sell donated items to raise funds and/or to provide clothing directly to those with need. Moreover, increased awareness of less fortunate members of local society may prompt those with higher incomes to reconsider their spending habits; perceived needs for new purchases may be re-appraised.

It should be noted that the respondent sample composition and size (Table 1) limit the outcomes of the study to providing an indicative but not absolute profile of population level responses. There is also a likelihood that responses are influenced by social desirability and may not truly represent the beliefs, attitudes or actions of respondents. We note that the data presented indicate the primary reasons, actions and behaviours of respondents; it is unlikely that all individuals are entirely consistent in these terms. The influences of demographic factors (gender, income, age and faith) will be further elucidated in future work.

## 5. CONCLUSIONS

We conclude that, in Southampton and Dakar, it is important that citizens view end-of-use clothing and textiles in terms of their value as a resource. Purchases are partly made due to need and the decision that items are no longer wanted or needed is partly aligned with loss of functionality. In contrast, purchases and end-of-use decisions are also influenced by factors other than functionality. However, there is an evident appetite for reuse in both Dakar and Southampton; the vast majority of respondents' end-of-use textiles are destined for reuse in both locations. Given that end-of-use clothing and textiles are frequently destined for reuse, the systems in place in Dakar and Southampton appear to already meet – in part at least if not fully - the principles of the waste hierarchy. The value of translating methods and approaches to clothing and textile waste management between Southampton and Dakar appears to be limited in that reported rates of reuse are high. Outcomes of this study indicate that fitness-for-purpose exists in both settings; we suggest that this has arisen due to co-evolution of systems and practices in which situational and societal factors influence are recognised and influential. Promotion of reuse in a Southampton could, however, potentially be achieved by emphasising the benefits of donations to the less fortunate within the local society.

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# VISIBILITY OF FUNDAMENTAL SOLID WASTE MANAGEMENT FACTORS IN DEVELOPING COUNTRIES

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**ABSTRACT**
The development of solid waste management (SWM) has been closely related to factors that influenced waste practices in the past and have shaped contemporary waste management systems. Multiple influencing factors need to be considered if SWM is to be effective. We have identified non-measurable or "invisible" factors that are not easily quantifiable or routinely measured but may influence local waste management practices and behaviour. Although the degree of influence of invisible factors is varied and sometimes vague in terms of impact, they serve as a starting point to design more effective waste management strategies. The aim of this study was to identify factors in solid waste management and classify them into two broad categories: "visible" (usually measurable by specific indicators or scales, quantifiable, considered in decision-making and implementation processes, and publically accessible) and "invisible" (not usually measured or quantified but still likely to influence waste generation, behaviour and operational practices, and perceptions about waste). A PESTLE (Political, Environmental, Social, Technological, Legal and Economic) analysis was employed as the basis for categorization. We identified 43 fundamental factors that were divided into the six different PESTLE categories. Experts in waste management were consulted via a Delphi survey and were found to consider 34 (79%) of these 43 fundamental factors to be visible and 9 (21%) invisible. This study highlights the need to adopt new perspectives regarding the role of these fundamental factors in SWM and to understand better the nature and extent of their influence on progress towards cost-effective, efficient, locally-optimised and sustainable waste management systems.

## 1. INTRODUCTION

Waste generation is connected to the socio-economic status of nations. In developing countries, the management of waste is becoming more complex as a result of rapid urbanization and the increasingly heterogeneous nature of consumer products. Increasing population level, rapid urbanization, increasing economic activity and an increase in society's living standards in major cities in developing countries have led to substantial growth in waste generation. Large increases in global waste generation may be attributed to developing countries, driven by a combination of high urbanization rates and rapid economic development (Le Courtois, 2012). Specific socio-economic conditions prevail in developing countries, including rapid population growth, rural-urban migration, lack of funds and low-skilled cheap labour. SWM systems in emerging economies often lack facilities, and suffer insufficient service coverage, improper disposal and treatment methods that can lead to major environmental and social problems.

Mukhtar et al. (2016), in a review of the history of urban waste management, clearly showed that direct adoption of developed cities' approaches without proper consideration of the local circumstances may lead to unsustainable future waste management in developing cities. The significance of factors in SWM in developing countries has been assessed with regard to: socio-economic impacts on waste generation (Bandara et al., 2007), recycling (Johari et al., 2014), waste to resource initiatives (Storey et al., 2015), the collection of municipal waste (Coffey and Coad, 2010) and disposal of waste (Zurbrügg and Schertenleib, 1998). The roles of specific factors are not always well-defined nor their influence measured. The impact of these factors, when apparent, can be validated from historical evidence. Multiple factors affect the development of SWM, including legislative, environment, social, technical, health, market demand and economic aspects. These key factors need to be identified and their role(s) understood to ascertain

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whether proposed waste management plans are duly tailored to local requirements and are viable in environmental, social and economic terms (Mukhtar et al., 2015). The fundamental factors relevant to waste management systems can vary due to the differences between individual cities' characteristics (Contreras et al., 2010). Some factors are usually measurable by specific indicators or scales, quantifiable, considered in decision-making and implementation processes, and publically accessible (e.g. demographic indicators): these factors may be termed "visible". In contrast, there are "invisible" factors that are not usually measured or quantified but still likely to influence waste generation, behaviour and operational practices, and perceptions about waste. These factors potentially influence the need for development of a waste management system but are qualitative (e.g. behaviour, understanding and awareness) and may be important if local conditions are to be recognised and addressed in the design and implementation of waste management systems.

Various factors in SWM play different fundamental roles in waste management practices (Barr, 2007; Periathamby et al., 2009b; Wilson, 2007). Previous research studies have addressed the significance of factors in SWM, including: policy and strategy (Rudden, 2007; Taherzadeh and Rajendran, 2014; Wilson et al., 2011), age and aging communities (Pickerin and Shaw, 2015), community behaviour and interactions (Shaw, 2008), the socio-economic impacts on waste generation (Bandara et al., 2007), recycling (Johari et al., 2014), waste to resource initiatives (Storey et al., 2015), the collection of municipal waste (Coffey and Coad, 2010) and disposal of waste (Zurbrügg and Schertenleib, 1998). Although the degree of influence of invisible factors is perhaps varied and sometimes unclear in terms of impact, in principle they serve as a starting point to design more effective waste management strategies or policies based on tangible local trends or evidence, rather than adopting best practices from elsewhere which may not address local characteristics, customs, uniqueness or waste composition (Mukhtar et al., 2016).

In some cases, invisible factors' roles in shaping waste management in developing countries may be more important than in developed countries, depending on the combination and roles of other factors under the local circumstances. Due to the influence of these factors in specific situations and at local scale, invisible factors that worked well in one locality may appear to be not important at all in other areas and therefore direct adoption may not lead to similar outcomes. The complexity of a city/region's waste management system requirements need to be: (i) recognised, (ii) analysed and (iii) turned into infrastructure, service provision and information campaigns that lead to behaviour change. For example, cities with diverse ethnic groups in the community might consider the differences in culture and lifestyles of each ethnic group in terms of waste practices, resource consumption and awareness on proper waste management practices. In less diverse settings, any differences in waste-related behaviour among different ethnic groups may not appear to be important and may not need to be considered in waste management plans and systems. It is possible that approaches to setting up waste collection systems, selecting suitable treatment methods and public awareness-raising campaigns need to take visible and invisible factors into consideration in order to reach desirable results. In this paper, we first aim to identify fundamental factors in waste management through a review of multidisciplinary literature and classify these factors into PESTLE (Political, Environmental, Social, Technological, Legal and Economic) categories. The second aim of this study is to classify further these fundamental factors into two broad categories, "visible" and "invisible" by employing a Delphi study. Results will provide clear classification of visible and invisible factors in developing countries and how these factors can be connected within the local setting that can accelerate the development of SWM systems.

## 2. METHODS

This study comprised two phases. The first stage was a literature review to gather and collate a list of fundamental factors that are reported to be relevant to and important in SWM; factors were then classified according to the PESTLE system (section 2.1). To identify factors, the literature relating to SWM was characterized and critically evaluated (Pérez-Belis et al., 2015). Scholarly articles were searched and subsequently reviewed based on the title, abstract and keywords to evaluate the suitability of the factors highlighted in the documents. The factors as collated and classified were then presented to a consultative group, members of which were asked to specify whether they consider each factor to be "visible" or "invisible" in current waste management practices (section 2.2).

### 2.1 PESTLE classification of factors in solid waste management

Important factors in waste management development were first identified via a literature review. The literature review was intended to identify factors on a qualitative basis. The factors were grouped according to the PESTLE classification (e.g. Zhang et al., 2011; Kolios and Read, 2013; Srdjevic et al., 2012; Zalengera et al., 2014) to create an analytical framework. Those factors identified were not intended to represent an exhaustive list but to generate a set of factors for subsequent consideration by the consultative group in the Delphi study (section 2.2).

### 2.2 Delphi survey

The Delphi method was employed to establish views on fundamental factors in SWM from a group of identified international experts. The Delphi method is a systematic and interactive research technique to obtain the judgement of independent experts on a specific topic. Selection of appropriate experts for the Delphi panel is critical to the quality of the study (Hsu and Sandford, 2007). Candidate participants were selected using the authors' extensive knowledge of international waste management professionals supplemented by an online search to identify persons with expert knowledge, including members of editorial panels from waste management-related journals, academics in higher education and established professionals from selected waste management companies and municipal authorities. The structure of the Delphi questionnaires followed the key factors and PESTLE categorization (section 2.1).

Respondents were presented with the list of factors within PESTLE categories (Tables 1-6) and asked to classify each factor as either "visible" or "invisible". The questionnaire specified the meaning of each of these terms, vis-à-vis:

- Visible factors are usually measurable by specific indicators or scales, quantifiable, considered in decision-making and implementation processes, and publically accessible.
- Invisible factors are not usually measured or quantified but still likely to influence waste generation, behaviour and operational practices, and perceptions about waste.

Respondents were also asked to provide information regarding their own role, expertise and experience in SWM. The Delphi questionnaire was administered by iSurvey, a survey generation and research tool for distributing online questionnaires used by the University of Southampton (https://www.isurvey.soton.ac.uk/).

## 3. RESULTS AND DISCUSSION

### 3.1 PESTLE classification of factors in solid waste management

The 43 factors are identified and briefly described in Tables 1-6. Differences in the numbers of factors in each PESTLE class were noted. We note that the observations to hand (Tables 1-6) do not represent an exhaustive list of factors or a quantitative profile. The specific purpose (section 2.1) is to inform and guide the subsequent Delphi survey (section 2.2). In particular, this analysis provides the structural framework for the Delphi study and the definitions of each factor (Tables 1-6).

### 3.2 Delphi survey: respondent profile

The respondent group comprised professionals from academia, private SWM consultants and companies, regulatory, local authorities and national government, charity organizations, business and trade and politics. Participants were classified according to their current location and its associated economic status (Table 7). The classification of countries by gross net income (GNI) is considered appropriate, but does not necessarily reflect the status of development in countries within the same classification.

### 3.3 Delphi survey: classification of factors as visible and invisible

Experts were asked to classify 43 factors (Tables 1-6) as visible or invisible. For about 80% of these factors, more than 50% of the respondents judged them to be visible. Each group of factors is considered in relation to PESTLE categories (sections 3.3.1 to 3.3.6).

#### *3.3.1 Political factors*

Political factors, as classified by respondents, varied in terms of being considered visible or invisible (Figure 1). The local government plan was considered visible by

**TABLE 1:** Political factors in SWM: the ability and roles of government to affect management and regulation.

| Factor | Description | References |
|---|---|---|
| Government stability | Strong government can hold its power and control over the country with minimal external influence | Plata-Díaz et al. (2014); Wilson et al. (2001) |
| Corruption | Fraudulent conduct for personal benefits, typically related to bribery | Taherzadeh and Rajendran (2014); Jones et al. (2010) |
| Accountability of leaders | Responsible and trusted leaders | Jones et al. (2010); Rudden (2007) |
| Local government plan | The plan for future development of the local area | Rudden (2007); Wilson et al. (2001) |
| Government priorities | Focus and attention on specific issues by the government | Moh and Abd Manaf (2016) |
| Influence of politicians | Effect of politicians' behaviour and character on specific issues | Taherzadeh and Rajendran (2014) |
| Bureaucracy | Excessively complicated administrative procedure | Godfrey and Scott (2011) |

**TABLE 2:** Environmental factors in SWM: the ability of environmental elements and resources to influence waste management behaviour and directions.

| Factor | Description | References |
|---|---|---|
| Environmental guidelines | Local/national guidelines that set specific environmental standards | Li (2007) |
| Environmental targets | Specific goals on environmental standards to be achieved within certain period of time | Li (2007) |
| Climate change | Changes in global and regional climate patterns resulted from unsustainable human activities | Zaman (2013); Johnson et al. (2011) |
| Geographical landform | Different features of the part of the earth which makes the terrain | Li (2007) |
| Local weather | Specific weather conditions at a particular place and time | Emery et al. (2003) |
| Environmental awareness | Awareness on the adverse impacts onto the environment resulted from unsustainable human activities | Triguero et al. (2016); De Feo and De Gisi (2010) |

**TABLE 3:** Social factors in SWM: the functionality of humans and their responses towards changes in waste management.

| Factor | Description | References |
|---|---|---|
| Seasonal variations | Specific annual celebrations at particular times of the year to celebrate a change of weather, season, crop harvesting and also racial, religious or ethnic affiliation which may or may not be officially recognized by the government | Gómez et al. (2009); Emery et al. (2003) |
| Religion | System of faith and worship | Taherzadeh and Rajendran (2014); Mohamad et al. (2012); Mohamad et al. (2011) |
| Cultural | Social behaviour, belief, traditions of particular group of people | Thyberg and Tonjes (2015); Martin et al. (2006) |
| Ethnicity | A particular group of people with same races, religious and origin that may have different culture from other groups of people of a country | Perry and Williams (2007) |
| Local/national events | Special days of celebration include national holidays, commemoration and also racial or ethnic affiliation which are officially recognized by the government | Gibson and Wong (2011) |
| Discrimination | Unfair treatment of individuals or groups of people | Ma and Hipel (2016); Sembiring and Nitivattananon (2010) |
| Socio-economic indicators | Changes in particular demographic components which are measured periodically | Triguero et al. (2016); Pickerin and Shaw (2015); Contreras et al. (2010); Wilson et al. (2001) |
| Resource consumption patterns | Changes of natural resources use for human activities within particular period of time | Taherzadeh and Rajendran (2014) |
| Shared norms | Rules of behaviour that are considered acceptable in group of society | Binder and Mosler (2007) |
| Rural-urban daily migration | Movement of people from rural to urban areas on a daily basis, mainly due to the economic and tourism activities | Henry et al. (2006) |
| Philosophical change | The evolving thoughts and feelings on particular issues that reflected in the changing in behaviour | Wilson et al. (2001) |
| Attitude-behaviour relationship | The relationship between an individual's values or intentions and their actions | Triguero et al. (2016); Taherzadeh and Rajendran (2014) ; Barr (2007) |
| Resistance to change | Response(s) of individuals or groups of people when they perceive or interpret change as a threat to them | Taherzadeh and Rajendran (2014) |

**TABLE 4:** Technological factors in SWM: the ability to apply suitable technology towards the improvement of waste management.

| Factor | Description | References |
|---|---|---|
| Skilled workers; experts | Workers with specific knowledge, skills and ability to perform best in their work; those who are widely recognized as a reliable source of technique and skills | Periathamby et al. (2009a) |
| Application of suitable technology | Application of the appropriate technology that is best designed for efficient operation | Taherzadeh and Rajendran (2014) Contreras et al. (2010) Wilson et al. (2001) |
| Facilities availability | Adequate number of facilities are developed for specific deployment | Taherzadeh and Rajendran (2014) |
| Rate of technology change | Development of waste management-related technology over time | Zaman (2013) |
| R&D Activities | New or innovative research that changes facilities, management and practices | Periathamby et al. (2009a) |

**TABLE 5:** Legal factors in SWM: the attributes and obligations of local authority and as institutions responsible to comply with waste management guidelines..

| Factor | Description | References |
|---|---|---|
| International directives | Environmental guidelines and instructions drafted by international organizations to create uniformity and consistency | Contreras et al. (2010) Rudden (2007) |
| Local policy | Policy that sets guidelines that determine the decision and actions | Taherzadeh and Rajendran (2014) |
| Producer responsibility | Approach taken by the producers in managing waste | Triguero et al. (2016) |
| Consumer accountability | Responsibility of consumers in buying and consuming, and managing waste arisings | Triguero et al. (2016) |
| Relevant SWM law | Compliance and enforcement of the law | Contreras et al. (2010) Bai and Sutanto (2002) |

**TABLE 6:** Economic factors in SWM: the ability of economic status to determine the marketability of recovered materials and waste products.

| Factor | Description | References |
|---|---|---|
| Potential income from waste | Monetary benefits from waste | Taherzadeh and Rajendran (2014) |
| Trade restrictions on waste | Limitation on trade activities to selected waste | Ray (2008) |
| Third sector restrictions | Limitation on trade activities to informal business and/or charitable organizations | Williams et al. (2012) |
| Availability of funds | Financial assistance available for projects or initiatives | Taherzadeh and Rajendran (2014)<br>Wilson et al. (2001) |
| Interest and tax | Application of interest and tax on goods and services | Jones et al. (2010) |
| Economic growth patterns | Changes in the amount of goods and services produced per head of the population over a period of time | Johnson et al. (2011) |
| Incentives | Rewards offered for appropriate or desired actions | Jones et al. (2010) |

all respondents; the majority view was that government priorities, government stability and bureaucracy were visible. The accountability of leaders was viewed as visible and invisible by a similar proportion of respondents, whilst the majority of respondents considered corruption and the influence of politicians to be invisible factors (Figure 1). These observations (Figure 1) illustrate the importance of government in setting the focus and direction in future development of waste management. The experts consulted in the Delphi survey highlighted the visibility of both local government plans and government priorities in relation to waste management. It can be argued that in developing countries, there is a relatively high dependency on government to facilitate proper waste management services and facilities. Most respondents (70%) considered bureaucracy to be a visible factor in waste management (Figure 1). Bureaucracy, whilst often a visible factor, can exert negative impacts if, for example, administration procedures are excessively complicated; unnecessary procedures and approval processes can cause delays in decision-making and implementation.

Government stability was considered by most respondents (61%) to be visible (Figure 1). Changes of government can clearly influence plans and their implementation at local to national scale. Stable government and related institutions allow establishment and maintenance of good relationships between politicians and authorities, ensuring better co-ordination of efforts in planning and development of efficient waste management services. Less stable government can generate uncertainty within the governmental institutions and disrupt decision-making and executing of waste management plans.

**TABLE 7:** Classification of Delphi survey participants' current location and national economic status.[1]Economic status determined by the gross net income (GNI) per capita per year (World Bank, n.d.); [2]GNI per capita of <$1,025; [3]GNI per capita $1,026 to $4,035; [4]GNI per capita $4,036 to $12,475.

| Participants' location | Economic status[1] |
|---|---|
| Mozambique | Low income[2] |
| Tanzania | Low income[2] |
| Togo | Low income[2] |
| South Africa | Upper-middle income[4] |
| India | Lower-middle income[3] |
| Indonesia | Lower-middle income[3] |
| Malaysia | Upper-middle income[4] |
| Vietnam | Lower-middle income[3] |
| Argentina | Upper-middle income[4] |
| Brazil | Upper-middle income[4] |
| Peru | Upper-middle income[4] |

The accountability of the leaders was classified by 53% of respondents as invisible. Measuring the qualities of leaders is inherently subjective. Changes to waste management cannot be readily or reliably attributed to the contributions of individual leaders; efforts of government authorities in improving waste management are generally cumulative, arising from multiple contributions from many individuals. There may, however, be some attribution of broad-scale outcomes to leaders who have taken a key role in developing a waste management strategy, most likely at a local scale. Although fewer respondents considered corruption to be visible than did bureaucracy, these two factors may be interlinked; excessive bureaucracy can precipitate corruption. It is arguable that corruption, antagonistic politics and bureaucratic procrastination commonly exist in developing countries' government systems, which influence the decision-making and stakeholders' involvement in relation to SWM policy and practice. Corruption is more likely to occur when partnerships and relationships are poorly designed or defined; the efficiency of the networking then becomes inefficient (Taherzadeh and Rajendran, 2014).

With regard to politicians influencing decision-making and implementation of waste management systems, most respondents (69%) considered this to be an invisible factor (Figure 1). Influences of politicians can be notable, however. For example, following Malaysia's General Election in 2008, the change in political leadership led to the challenge for the federal government to implement finally the Solid Waste Management Act 2007. Following the result of the election, a contrast in political relations emerged between (i) states in the same political coalition as the federal government and (ii) states ruled by the opposition party and not aligned with the federal government. Changes of leadership in some of the states had caused the non-uniformity standards of waste services that led to problems in some areas.

*3.3.2 Environmental factors*

Many of the environmental factors (Table 2) are considered to be visible by survey respondents. Environmental guidelines and environmental targets were highlighted as visible by the majority of respondents. Clear guidelines and targets on environmental aspects are vital for improving SWM: guidelines should provide procedures and methodologies for monitoring and enforcing the regulations; targets must be achievable and realistic to drive initiatives towards improvements. The importance of geographical landform on the development of SWM systems is also considered visible by most (70%) respondents. Vehicle-based collection in less accessible areas in developing countries may inhibit expansion of service areas in less reachable, mainly rural areas: some facilities, social and economic activities depend on the suitability of transport infrastructure. Spatial variation in this regard requires understanding of the local situation in order to plan for a workable and efficient waste management system. The quality and coverage areas of waste collection services in some of developing countries differ between urban and rural areas, which may explain the observed split between respondents considering geographical landform visible and invisible (Figure 2).

Environmental awareness was seen by most respondents as a visible factor (Figure 2): awareness underpins waste behaviour that can contribute to more sustainable SWM. The importance of having a population that is well-educated regarding environmental and waste management issues is thus highlighted and confirmed as commonly recognised and incorporated in SWM systems and approaches. This outcome is notable: enhancing awareness of good waste practices and sustainability has been stated as a key challenge in SWM in developing countries (Ferronato et al., 2017; McAllister, 2015; Storey et al., 2015). With environmental awareness commonly viewed as visible (Figure 2), there is potential to increase further awareness among public in developing countries to further progress initiatives towards sustainability in SWM.

Notably, Delphi respondents indicated that climate change is more commonly invisible than visible (Figure 2); less than one third of respondents regarded climate change as a visible factor in SWM. This observation is somewhat at odds with the general recognition of climate change as a major and global environmental problem for the waste sector (Turner, 2016). Omissions of climate change from visible factors in SWM policy and practice renders the impacts of SWM on climate invisible and can lead to decision-making that fails to reduce or even propagates waste-related climate change impacts.

*3.3.3 Social factors*

Respondents' views of social factors in SWM as visible or invisible markedly varied across the factors considered (Figure 3). Resource consumption patterns were regarded as visible by the majority of respondents; economic prosperity is commonly associated with demand for products and materials for consumption which in turn leads to higher demands on effective SWM systems. Experts mainly have considered that consumption patterns are already incorporated in SWM planning and system design. We note that preventing or inhibiting high rates of consumption and avoiding "throw-away" mentality could reduce waste generation by enhancing reuse (Williams and Shaw, 2017). Local/national events were considered to be visible in SWM by most respondents (Figure 3). Celebration of local and national events draws communities together, but can lead to notable quantities of waste that need to be dealt with, requiring additional resources. Seasonal variations were also considered to be visible by most respondents. Such celebrations are typically ethnic, cultural and religious events that occur within specific communities; the associated waste is often generated at a household level. For example, during the Ramadhan and Eid-ul-Fitr celebrations, food waste is generated in higher than usual quantities. Muslims tend to buy more food than their normal require-

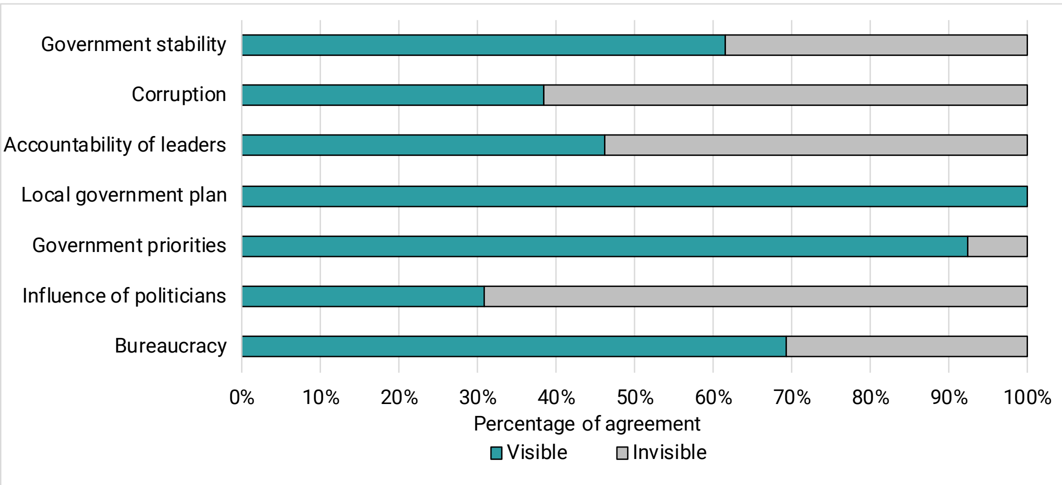


**FIGURE 1:** Proportions of Delphi survey participants classifying political factors (Table 1) as visible or invisible.

ment for self-consumption and guests; however, with limited time for consumption, food may not be efficiently consumed and ultimately ends up as waste. Changes in waste collection frequency, additional vehicles and workforce are needed in consequence. Likewise, Christmas and Easter celebrations can lead to increases in household waste comprising large proportions of recyclables, such as packaging, wrapping papers, greetings cards, glass bottles and food waste.

Most of the responses from the Delphi respondents identified rural-urban daily migration as a visible factor in SWM (Figure 3). Unequal economic growth distribution between rural and urban areas leads to daily commuting, mainly driven by the economic opportunities and access to education, health, commerce and trade, for example. Although daily migration is already taken into account in many respondents' views, these activities influence the quantities and locations of waste generated in a city and confound estimates of likely waste arisings on a per capita basis. Understanding the influence of daily migration on waste characteristics, generation trends and patterns is non-trivial but nonetheless informs and guides SWM policy and practice in most of the locations represented by the Delphi survey participants (Table 7).

There were five factors classified as equally visible and invisible: cultural, socio-demographic, attitude-behaviour gap, resistance to change, and shared norms. Religion and discrimination were considered to be invisible by most respondents (Figure 3). These factors thus appear to be incorporated in SWM policy and practice on an inconsistent basis. Religion is seen as influential force to transform public practices and behaviour in waste management (Mohamad et al., 2011; Taherzadeh and Rajendran, 2014). Discussions on the influence of religion in waste management, however, are limited and often included within broader sets of socio-demographic indicators (Mohamad et al., 2012). Religion is perhaps not widely considered in SWM systems despite the potential of religious organizations to assist in transforming the public's waste behaviour. Likewise, cultural factors and ethnicity may present opportunities to transform waste management behaviour through social groups and communities, although the visibility of these factors varies between settings according to the Delphi survey responses in this study (Figure 3). We note that factors are not necessarily mutually exclusive: ethnicity may, for example, be associated with cultural and religious factors and their related behaviour and values regarding resource consumption and waste management. We note that although culture, religion and ethnicity may well be closely associated, cultural factors are more commonly recognised and incorporated in SWM (Figure 3).

Discrimination was viewed as a visible factor in SWM by around 1 in 6 Delphi respondents. This is perhaps a weakness in many settings: urbanization and economic growth lead potentially to inequality, harassment and exclusion due to individuals' low social status. The few studies on this issue have highlighted, for example, discrimination of female SWM workers in developing countries (Ma and Hipel, 2016; Nunn, 2012), informal recyclers (Mull, 2005; Sembiring and Nitivattananon, 2010) and racism and social status of communities (Baabereyir, 2009). The intrusion of political agenda in solid waste management hindered the occurrence of social injustice which make discrimination factor is least considered.

### *3.3.4 Technological factors*

Most, but not all, Delphi respondents considered all technological factors to be visible (Figure 4). This outcome highlights the importance of available and suitable facilities for waste management activities that lead to positive waste management behaviour among the public and improve operational efficiency. A lack of suitable facilities can contribute to stagnation or decline of local SWM efficiency, whilst availability of appropriate facilities can motivate public participation. Suitable facilities for SWM also permit resource recovery from the waste stream and thus contribute to more sustainable resource use.

The needs for skilled workers and experts are commonly regarded as visible factors in SWM in developing countries; pertinent skills and expertise can enhance and improve initiatives for and operations of SWM. In contrast, an inadequate skills base can lead to inaccurate waste

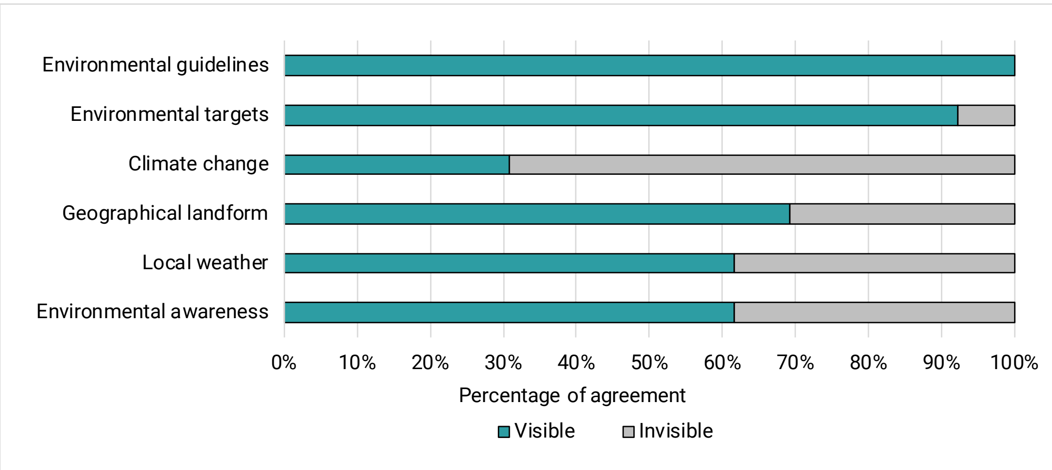


**FIGURE 2:** Proportions of Delphi survey participants classifying environmental factors (Table 2) as visible or invisible.

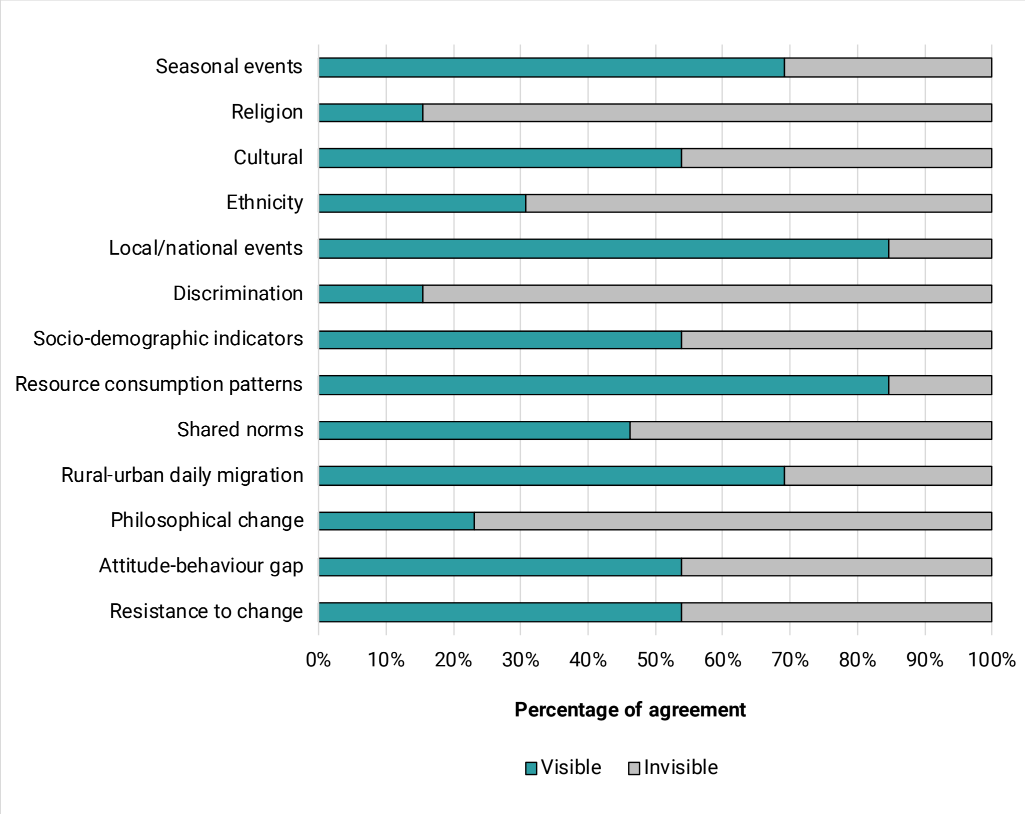


**FIGURE 3:** Proportions of Delphi survey participants classifying social factors (Table 3) as visible or invisible.

projections and ineffective planning. There are relatively few academic studies that have evaluated the importance of skilled workers and experts in developing sustainable waste management plans (Dinie et al., 2013; Hazra and Goel, 2009; Moh and Abd Manaf, 2014); there is thus a contrast in that knowledge appertaining to skills and expertise is somewhat poor, whilst skills and expertise are commonly incorporated in SWM. We note that waste management involves both technical and non-technical disciplines; therefore, skilled workers can contribute to the efficiency of operational issues. Optimizing the recovery of materials from the waste stream and reducing the maintenance cost of facility operation by proper handling of the waste treatment facilities, for example, relies on a suitably skilled workforce. Most Delphi survey respondents identified application of technology in waste management as a visible factor. Technology can conveniently and efficiently support the SWM systems, for example, when applied to waste treatment operations and recovery of resources from the waste stream. Developing countries, as indicated by the survey respondents, differ in terms of whether changes of technology and research and development are visible. There is a relative lack of financial assistance and allocation of funds for developing technology for SWM in developing countries. The lack of research and development activities in developing countries can lead to the selection of technology that is inappropriate in terms of local weather, waste characteristics, financial capabilities and availability of experts and skilled workers. Consequently, the selected technology may not operate effectively (or at all), thus wasting the resources allocated and causing social indignation.

*3.3.5 Legal factors*

Legal factors were all significantly classified as visible by the majority of respondents (Figure 5). Relevant SWM law and local policy are both considered visible factors by more than 95% of respondents.

Outcomes in this regard reflect the status of local government plans and government priorities as visible factors in SWM (Figure 1); laws derive in part from political ambitions and purpose. In developing countries existence of local government plans is clearly important and is already incorporated in SWM systems, as is relevant SWM law. Most respondents considered that accountability of consumers is a visible factor; management of post-consumer waste and producer responsibilities are key aspects. International directives were considered to be visible by most respondents; international directives on sustainability of waste management do not always apply and this situation is reflected in the responses received in this instance.

*3.3.6 Economic factors*

All of the economic factors considered were viewed by most respondents to be visible factors in SWM (Figure 6). Waste trading between developed and developing countries became an alternative solution to disposal for developed countries. This “symbiotic” relationship was appar-

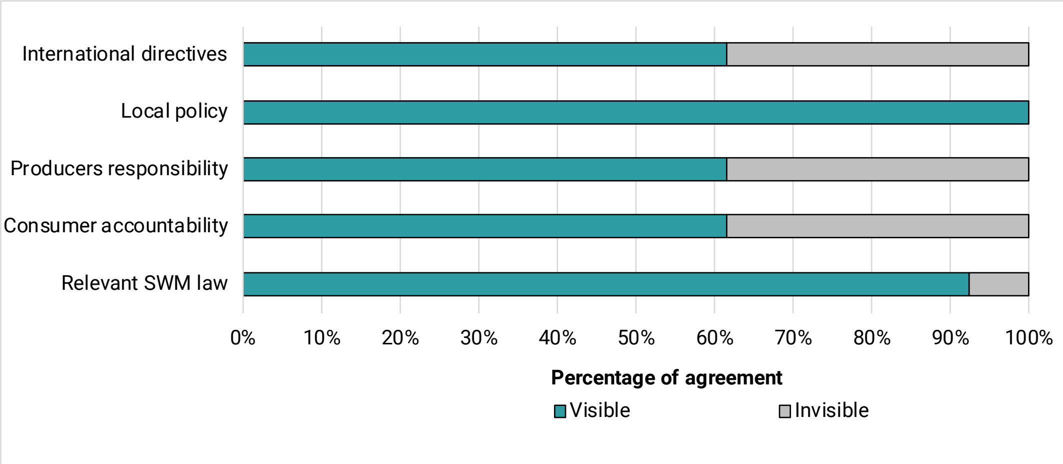


**FIGURE 4:** Proportions of Delphi survey participants classifying technological factors (Table 4) as visible or invisible.

ently beneficial to both partners; developing countries were generating income from recovery of resources from waste whilst developed countries benefits secured reduced disposal and treatment costs. However, this rapidly led to immoral and unethical practices that resulted in impacts to human health and the environment in developing countries. The strong agreement on the importance of available funds allocated for waste management projects was observed in developing countries with undivided agreement of 100%. Developed countries have more sources of financial support to develop their waste management systems when compared to developing countries (Periathamby et al., 2009b; Wilson, 2007; Wilson et al., 2001). Incentives for the use of selected waste management processes/ systems were viewed as motivational tools to reward good practice. The importance of offering incentives to improve further a waste management system was highly recognized and visible, which more than 85% of agreement from developing countries.

A similar pattern of agreement was observed regarding the implementation of tax and interest on waste trading and also the potential income generated from waste due to the public's waste practices. Respondents from developing countries had slightly less concern regarding the importance of these factors in their waste system, which was an unexpected finding given by the rapid growth of business activities relating to waste trade and resource recovery in developing countries like China and Indonesia (Damanhuri and Padmi, 2012; Hui et al., 2006). Emphasis on the investment in facilities and improvements in waste management services can be observed alongside rapid economic growth in developing countries. With a stronger economy, the consumption of the resources increases alongside waste generation and this obviously influences emerging waste management systems. The importance of economic growth is very significant in developing countries with 78% agreement from respondents. Overall, all economic factors were classified as visible, validating the importance of a strong economy to accelerate improvements to SWM systems.

## 4. CONCLUSIONS

This study highlights fundamental factors in SWM and classifies them into two broad categories; visible and invis-

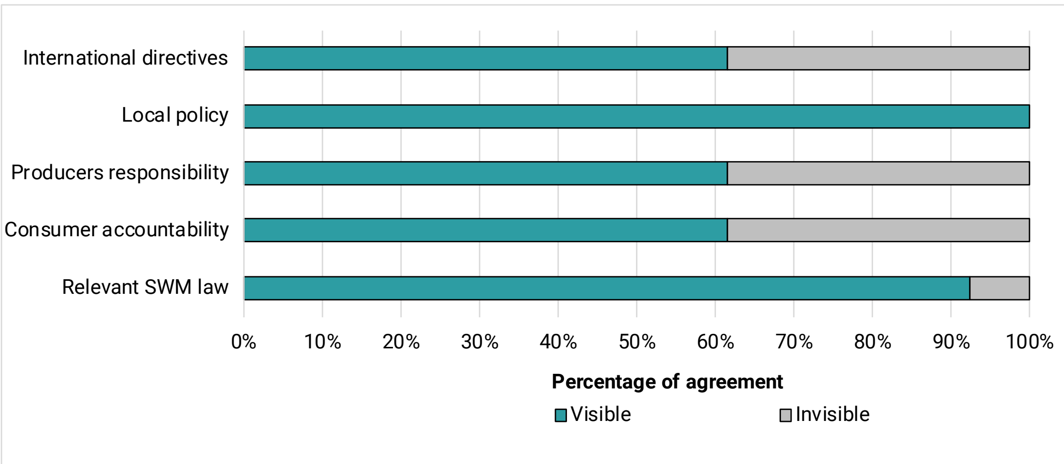


**FIGURE 5:** Proportions of Delphi survey participants classifying legal factors (Table 5) as visible or invisible.

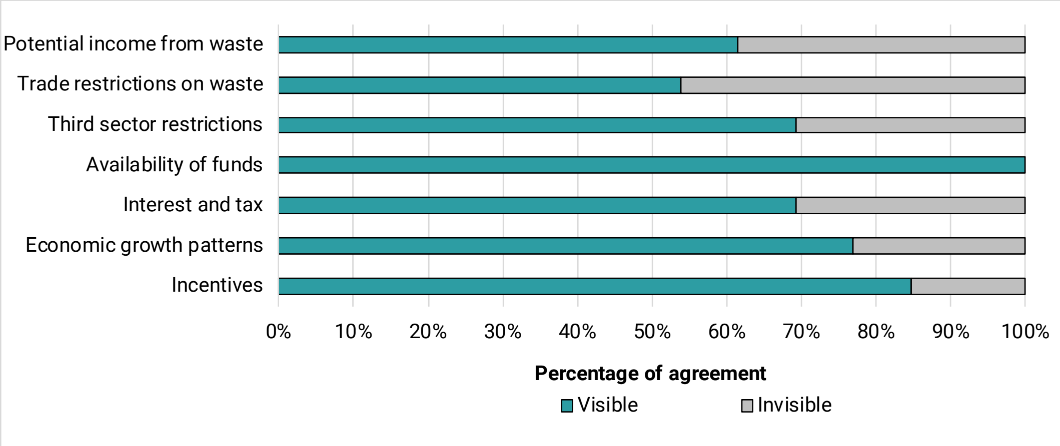


**FIGURE 6:** Proportions of Delphi survey participants classifying economic factors (Table 6) as visible or invisible.

ible, grouped using a PESTLE framework. From our Delphi survey, we concluded that environmental, technological, legal and economic factors tend to be classified as visible while social and political factors are generally regarded as invisible in developing countries. The recognition of, and emphasis on, invisible factors that are not routinely or commonly considered could potentially enhance the sustainability of a local waste management system. Sustainable waste management requires, for example, appropriate public waste management practices and participation: there may be a need to address social factors (e.g. Table 3; Figure 3) that are deemed invisible if public attitudes and behaviours are to lead to more and better participation in waste management activities and initiatives. Likewise, in terms of governance, there are often overlapping responsibilities and unclear assignment of responsibilities for tasks relating to solid waste management in developing countries. This situation can hinder the effective implementation of SWM improvement initiatives, thus political factors may be more fully considered in the waste management planning in order to accelerate improvements towards effectiveness, efficiency and (economic and environmental) sustainability.

On the basis of this study's outcomes, we propose that understanding the factors that drive the development in waste management systems in developing countries needs to be underpinned by evidence that is not only limited to waste management system, but also involves the characteristics of broader society, government administration and economic status of each country/city. Moreover, fundamental factors elucidated here may have to be considered in the local context to be effective; emphasis should be placed on those factors (if known) that most strongly influence the local conditions. For example, cities with diverse ethnic groups within their community might consider the differences in cultural and lifestyles of each ethnic group in terms of waste behaviour, resource consumption and awareness of waste management practices. In less diverse countries, any differences in waste-related behaviour among different ethnic groups may not appear to be important and may not be an important consideration in waste management plans and systems. It is possible that approaches to setting up waste collection systems, selecting suitable treatment methods and public awareness-raising campaigns need to take visible and invisible factors into consideration in order to reach optimum results.

The strength of influence of factors explored in this study – visible or invisible – remains to be elucidated. There is a prospect that the influence of invisible factors in particular is unique to specific setting of waste management systems at local scale, and that approaches to SWM that are workable in developed countries may not translate with guaranteed success to developing countries due to the differences in socio-cultural, economic and political structures. Even established technologies used in developing countries may not be suitable for other developing countries without modifications underpinned by detailed study and evaluation, and due recognition of both visible and invisible factors. Identification and emphasis of the role of invisible factors potentially helps to accelerate the improvement to success.

We contend that the visibility of factors needs to be evaluated to achieve a meaningful understanding of the factors underpinning the operation and enhancement of SWM. Moreover, there is a need to elucidate the strength of influence that these factors exert on the on a SWM system such that progress towards cost-effective, efficient, locally optimised sustainable waste management systems can be made.

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# FULL-SCALE PROJECT: FROM LANDFILL TO RECREATIONAL AREA

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**ABSTRACT**

Due to the harmful effect on the environment, landfill gas has to be collected and processed. One of the possible solutions would be to cover the landfill with a bioactive layer, where methane is oxidized. The main goals of the closure project at Kudjape landfill were to extract the fine < 40 mm fraction from the landfill in order to construct a methane degradation cover and to transform the closed landfill into a recreational area. Multi-purpose pathways were designed to cover the remediated landfill and were connected to the existing trails nearby. The trails can be used for jogging, hiking or biking and during winter time, for skiing and sledging. The surrounding area was cleaned from fly trash and new vegetation was planted into the cover layer. Car park and rest areas were created to attract people. The installation of the methane degradation layer along with the landscape restoration were finished in autumn 2013. Soon after the restoration it was noticed that some of the planted trees were not adapting to the environment and died off. The current study describes the extent of die-off over time, focuses on understanding the reasons for vegetation failure, and proposes measures to correct them. The research continues to monitor the processes within the methane degradation cover layer in order to provide recommendations for similar projects in the future.

## 1. INTRODUCTION

Many closed and reclaimed landfills are situated near to or within inhabited areas. This makes it attractive to use them for beneficial purposes. Concepts for remediation of large scale brownfields into multi-purpose recreational areas have been described by Gliniak and Sobczyk (2016), Nijkamp et. al (2002) and Limasset et. al (2018). Transforming previous landfills into green zones may be part of a general restoration program (Wong et al., 2013). Due to environmental and physical problems that impose restrictions, it is realistic to use them mostly as open spaces (parks, racing arenas, sporting grounds, etc.). For instance, municipal landfills are particularly difficult to reclaim to beneficial purpose because of landfill gas emission. Landfill gas is well known as greenhouse gas – a mixture of methane ($CH_4$), carbon dioxide ($CO_2$) and other trace gases, which is generated under anaerobic conditions. It can be collected and used for energy production or flared. However, the collection of gas takes decades, is not very efficient and is not feasible in low gas production phases, in low income countries where active gas collection system is beyond the budget, and in landfills where waste with low organic content is disposed of. One possible solution to avoid migration of gas would be covering the landfill with a bioactive cover layer, where methane is oxidized in-situ (Hilger and Humer, 2003; Barlaz et al., 2004; Huber-Humer, 2008; Scheutz et al., 2009). The methanotrophic bacteria uses methane as its carbon and energy source, and degrades it into $CO_2$ and $H_2O$ in exothermic processes (1):

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O + 882.6\ \text{kJ} \qquad (1)$$

In places with limited availability of mineral cover material, soil-like fine fraction could be extracted from the same landfill cell, thus offering an alternative technology for combating climate change: instead of installing an active gas collection system, methane is degraded in-situ. Combining landfill gas treatment in natural biocover processes with park-function is rare.

## 2. SITE DESCRIPTION

The landfill remediation activities took place in Estonia at Kudjape municipal landfill, located on the island of Saaremaa (N 58:16:06, E 22:32:23). The landfill consisted of an approximate volume of 200,000 m$^3$ of disposed waste. The waste was initially disposed of on a flat unlined area; it reached a final height of approximately 12 m by the landfill closure time in 2009. Average annual temperature at the site is 5.6 °C and annual precipitation is 594 mm (Ideon, 2011).

Knowing that clay or any other material with low per-

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meability was not available locally, and that the dumpsite was considered as a low-risk site, an alternative cover design was planned with methane degradation function. As prescribed by environmental authority, about 1.5 m thick cover layer had to be used to cap the landfill. Biocover was composed of 1.2 m porous organic-rich materials, and a 0.5 m mineral gas distribution layer with the main purpose of distributing gas evenly into the top-layer, where methane is degraded (Figure 1). The cover material was extracted from the same landfill using technology known as Landfill Mining (LFM).

The upper 6 to 8 m layer of the landfill was sieved by Doppstadt SM518 drum sieve for separation of waste into two fractions: the desired < 40 mm fine fraction (biocover), and > 40 mm coarse fraction (backfill). In total, 57,777 $m^3$ of disposed waste was sieved during the LFM project between August 2012 and September 2013. As much as 16,430 $m^3$ of fine fraction (FF) was produced, giving the ratio of fine/coarse as 30/70. The properties of waste fractions are described by Kaczala et al., 2017 and Bhatnagar et al., 2017 and the properties of the tailored cover layer by Pehme et al., 2014.

The concept of final use of the site was modified during the works. It was understood, that there were widely used Kudjape-Upa sporting trails nearby, which, unfortunately, were flat. The Kudjape landfill area, however, was going to be the highest peak in the surroundings, which seemed well suited for diversifying these trails. A landscape architect was contracted to draft an improvised project amendment with jogging and ski trails, a picnic corner and a sledging slope. The landfill site was designed to be as diverse as possible, taking into account both the technical construction features of the landfill closure design, the economic considerations, and sporting requirements. A total of 2,420 meters of trails were planned with different lengths and slopes: 1,360 meters are located on landfill body and 1,060 meters, around the adjacent recycling facility. The slopes are between 1:3 and 1:20. The difference in height is 12 m.

The four-meter wide trails were sited into a very soft biocover; therefore, geotextile fabric was rolled on the bottom of trails (Figure 2), which was composed of a 300-mm thick gravel layer and a 100-mm thick mixture of gravel and wood chips (Figure 3). A conflict between diminishing the prescribed thickness of the biocover (1.2 m) in view of the road surface height was minimized by lifting the trails slightly higher than the surface of the landfill.

Sufficient difference in height and a very long slope created perfect conditions for snowboarding and sledging for children. The kids' area was covered by pure soil for safety

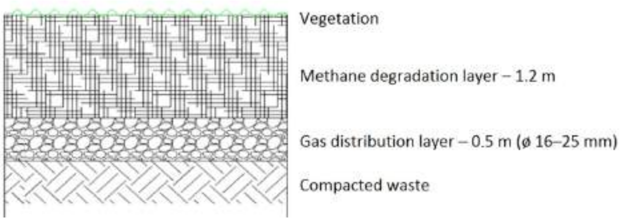


**FIGURE 1:** Design of the cover layer at Kudjape landfill.

**FIGURE 2:** Securing trails into soft biocover by supportive geotextile.

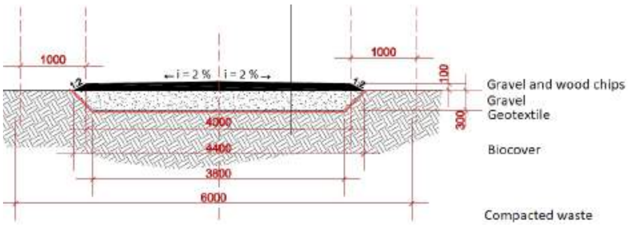


**FIGURE 3:** Design of the trails at Kudjape landfill.

purposes; the rest of the hill was covered by exposed fine fraction. Since winter sports trails, in particular, were not allowed to cross each other, a wooden bridge was built at the foot of the hill. Ditches two meters wide and a half-meter deep were excavated around the landfill, and a half-meter-deep pond in the northern part of the area was designed to collect stormwater during rainy periods. Rules of conduct were prominently displayed to keep the sporting crowd from falling into these ditches.

The whole area is covered by high and low vegetation, which highlights the relief along the slopes, diversifies the landscape, divides the area with different functions and stabilizes the biocover. To speed up the growth of the grass, hydroseeding was used throughout the landfill area. Lower shrubs (common honeysuckle, black elders, reeds), and trees Norway spruce (*Picea abies*), European larch (*Larix decidua*), Norway maple (*Acer platanoides*) and green ash (*Fraxinus pennsylvanica*) were planted into the cover layer. All trees were marked with numbered tags in order to create a database with tree species, initial height and GPS coordinates. In total, 338 trees and 959 shrubs were planted. All trees were at least 1 to 1.5 m in height; the diameter of root-balls was approximately 20 cm. Soon after planting it was noticed that some of the trees did not adapt to the environment and died off. Two growing seasons later, 50 species of Scots pine (*Pinus sylvestris*) and 50 species of Silver birch (*Betula pendula*) were added within the project warranty period to replace the lost species.

Elements of outdoor furniture and supporting infra-

**FIGURE 4:** View of remediated landfill in 2013 (Tammjärv et al, 2016).

structure, e.g. a staircase on top of the hill, an information board, picnic tables and litter bins were made by Elegrotech OÜ from recycled mixed plastics (Kriipsalu and Käsnar 2016), which actually contained plastic material, which had been excavated from the same landfill. The Kudjape Waste Recycling Facility is situated next to the landfill; therefore, the aim was to expose it to potential visitors, for educational purposes. Landscaping (Figure 4) was done within the existing closure permit, although the initially proposed (Figure 5) and final look (Figure 6) of the project were much different.

Landfill cover is monitored and maintained according to waste permit, which was issued by local environmental authority. Landfill owner has to monitor and report the following: groundwater level in four monitoring wells; quality of groundwater and surface water; landfill gas composition in one 10-m deep gas well; settling of waste masses, erosion of slopes. He has to maintain ditches, remove littering, and apply corrective measures against erosion. Grass is cut up to 15 times per season, which greatly exceeds the planned frequency (2 to 3 times). The reason for very good grass growth is nutrient-rich cover material and favorable moisture conditions in upper soil layer, as precipitation exceeds evapotranspiration by 200 mm in Estonia.

Objectives of the current full-scale project, where the landfill was remediated into recreational area, were: to monitor long-term functional performance of the cover layer (landfill gas safety); to monitor vegetation die-off; and to evaluate the social acceptance of the recreational area.

## 3. MATERIALS AND METHODS

Efficiency of the methane degradation layer was determined two years after the cover layer was installed. Methane and carbon dioxide emissions through the cover layer were monitored in twenty-nine measuring points which crossed the landfill (Figure 7). All measurements were taken by closed loop static chamber (40 L) method (Heinsoo et al., 2016), using portable GA2000 gas analyser (Geotechnical Instruments, UK) (Geotech, 2009).

The moisture content and temperature values were measured in situ with portable device Field Scout TDR 300 (Spectrum Technologies, USA). The soil samples taken from 29 measuring points were analysed in laboratory to measure pH and duplicate moisture content values. The pH was determined by the device pH/Cond340i (WTW, Germany) from the water extracts obtained from 5 g of the soil sample and 50 ml of distilled water. The water content of the soil samples was calculated from the dry matter content which was determined by gravimetric analysis.

ArcGIS for Desktop drawings were used to compose lo-


**FIGURE 5:** Layout of a remediated landfill as initially proposed (Kriipsalu et al., 2016).


**FIGURE 6:** Layout of a remediated landfill as actually constructed (Kriipsalu et al., 2016).

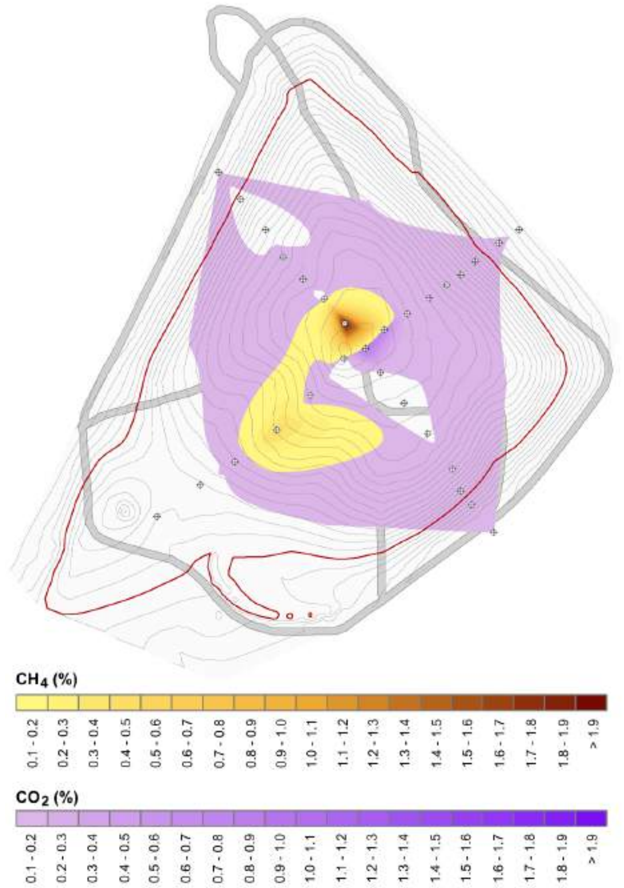


**FIGURE 7:** Distribution of landfill gas through the surface of the landfill in July 2015 (Heinsoo et al., 2016).

cation maps of measurement points. The Land Use Office orthophotograph was used as the base map. The interpolation of the measurement results was done using the natural neighbour method. All trees were inspected in 2014 and 2015 according to Tammjärv et al. (2016). Readings were inserted into the database and linked with numbered tags and GPS coordinates of trees.

Social acceptance was studied according to local media monitoring and assessment was made by interviewing personnel of the recycling center.

## 4. RESULTS AND DISCUSSION

### 4.1 Functional performance of the cover layer

It appeared that $CH_4$ and $CO_2$ emissions through the surface of the landfill were missing or very low (Figure 7). Methane was detected only from two measuring points out of twenty-nine and the values were low (maximum recorded value 2.1% vol) (Heinsoo et al., 2016). Spatially presented results demonstrate slight emissions of $CO_2$ (Figure 7), which is a result of degradation of methane (1.1). Emissions were missing during winter when the surface is frozen. From this perspective, the site is safe to visit. According to these results, the methane degradation layer which is made from excavated fine fraction serves as biocover very well.

### 4.2 Vegetation die-off

Two years after planting, 60% of the larches and 40% of the spruces had died off. The large number of dead trees suggests several possible reasons: a) wrong species; b) wrong planting time, planting methods or size of planted trees; c) unsuitable soil; d) unsuitable soil moisture, pH or gas content in soil; e) vandalism or other.

As seen from Figure 8, the pH was 7.5 to 8.0 in most places where trees were planted. It is higher than typical for Norway spruce (pH 5.2 to 5.5) or European larch (pH 6 to 6,5). The survived larches had a mean annual growth increment of 6 cm and spruces, 3.5 cm, which is at least 3 to 5 times less than expected. Scots pine and silver birch have adapted better in soils with higher pH and moisture fluctuations; this was confirmed visually.

Lack of moisture may also be a reason for die-off. Figure 9 shows lower moisture values than typical, but there was no obvious signs of withering or water stress yet.

Size of the planted trees and their root balls appears to be critically important. Root balls of pine and birch had the same size as spruce and larch, but their shoot-root ratio was less than 3:1, which ensures better adaptation to unusual growing media. The average height of pine and birch was 50 cm, compared to 1.0 to 1.5 m for spruce and larch. Spruce and larch had unfavourable shoot-root ratio (7:1), which might have promoted die-off.

The planting time of spruce and larch (end of August) was unusual due to the end dates of the project. Soil char-

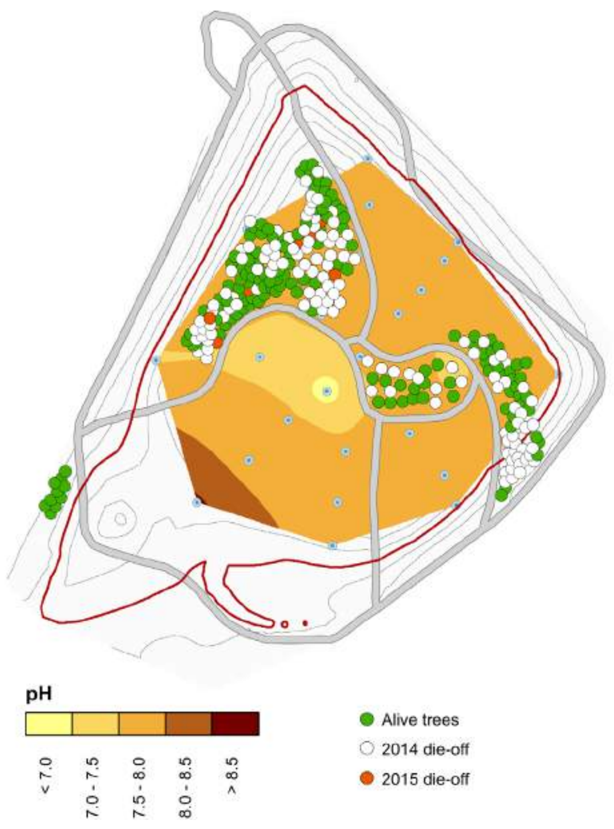


**FIGURE 8:** Tree mortality one and two years after planting, considering pH of cover layer (Tammjärv et al., 2016).


**FIGURE 9:** Tree mortality one and two years after planting, considering pH and moisture of cover layer (Tammjärv et al., 2016).

acteristics are expected to be highly dynamic in time, because of mineralization of organic material in biocover. Therefore, the growing conditions will continue to change in time. Tree species should be selected on different bases as compared to regular forestation or landscaping. It is useful to test various tree species in actual conditions before large-scale planting works.

### 4.3 Social acceptance

After the recreational area was opened it was immediately accepted among local people. Initially, people visited it out of curiosity, however more and more came for sport or just spending time actively. The number of visitors is increasing. The area is being used for walking, jogging, Nordic walking, Nordic skiing, biking, snowboarding, sledging and most frequently just for walking dogs. Recently, disc golf was installed, and is in active use. Visitors are coming from the nearby town Kuressaare. The travel distance is 4 km from the center, and 1.5 from town border. There is a public recycling center at the site, which provides a double reason to visit the park. The multi-purpose pathways existed near the landfill site long before the opening of the recreational area, therefore people were used to using the surrounding area for sports and recreation.

In early days, there was also some criticism among the visitors. They didn't like the little pieces of glass and plastic that could be noticed on the ground, even if the material for trails was free of any waste. These complaints stopped after the grass grew. The landfill area is continuously maintained and that guarantees its pleasant look. Some visitors were disturbed by the sight of dead trees. The response was to plant replacement trees.

Additionally, there were some complaints about the sporting trails. Some kids who rode bicycles in the opposite direction of the planned route got into a bicycle accident. After placing additional signs describing the proper use of sporting trails there were no more complaints.

## 5. CONCLUSIONS

The biocover concept may be considered as a useful and sustainable tool for remediation of small to medium landfills with low-level $CH_4$ emissions. It is possible to extract cover material from the existing landfill body. The results of the study show that the spatial emissions of $CH_4$ and $CO_2$ through the surface of the landfill were very low or missing. This demonstrates that the design of the methane oxidation layer was appropriate.

The planting of trees requires good planning and some trial and error. The die-off of larches and spruces was unacceptably high – 60 and 40% respectively. The die-off of pine and birch, however, was very small. It is recommended, that in addition to selecting species which tolerate soil gases, unusual pH and unfavourable moisture conditions, the plants should be rather small. Shoot-root ratio should not exceed 3:1, and planting time should be in early vegetation period.

It was possible to create a fully functioning recreational area from old landfill. It was accepted among local people from the first day. As it is still an anthropogenic landscape, the information about proper use of the area should be provided for visitors in order to make its use safe and enjoyable. It accomplished its goal of restoring the previous landfill area for the benefit of the community.

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CISA

# DEVELOPING COUNTRIES CORNER

## A LITERARY CAFE IN YAOUNDÉ, CAMEROUN

In 2015, the University of Padua, specifically the Environmental Engineering research group, developed a bottom up cultural project in Yaoundé, Cameroon, to raise awareness in environmental topics, particularly solid waste management, involving local people of different cultural levels.

### The background

Yaoundé counts more than 4.000.000 inhabitants with a rate of population growth in the range of 7-10%. Nearly 80% of the urban population lives in informal settlements occupying 72% of the urban space. 46% of the population has no access to improved water sources and uses latrines as wastewater devices, mainly located close to water wells (UN-Habitat, 2015). The weekly production of municipal waste in the town has been calculated in about 1200 t.

Less than 50% of waste produced is collected by the commissioned company HYSACAM, and only along the main roads, while in the poor neighborhood (shanty towns) the wastes are abandoned on the road or in water channels and often burned in the open air (Figure 1). No formal recycling is planned and only 5% of waste is informally recycled by private citizens. Awareness of the people with regard to environmental and health issues linked to wastewater and solid waste management should be urgently improved, although no educational material is available, particularly for the unschooled people.

Within this framework, a decision was reached to design a venue open to the public of all ages and levels of education, where they can find more or less detailed information (video, books, journals, etc.) relating to environmental issues, can connect to the internet to browse international related platforms, and in which seminars on specific topics can be organized to encourage debate and raise awareness.

### Specific aims

To meet the main objective, the project has been designed to include a series of specific aims, based on the model of the literary cafe, seeking to promote culture particularly on environmental topics. Unlike traditional literary cafés, the venue in Yaoundé likes to welcome common people rather than intellectuals, since the priority is to contribute towards solving the daily problems caused by poor sanitation in town. Local and visiting experts are invited to promote dialogue.

The physical space has been created to host creative interactions and conversations, outreach and training seminars: to organise meetings to facilitate the discussion between groups of people on specific topics; to promote the link between different networks; to support young people or students in improving their knowledge.

The literary café was initially financed by a series of different institutions, but in order to reach the economic sustainability, after a start up phase, the organizers set up a business activities.

### Designing the project

The project provides for the realization of an appropriate cultural space in which to collect and boost interest on environmental topics, promoting awareness and disseminating the best practice with regard to daily solid and liquid


**FIGURE 1:** Waste in Yaoundé: a) in a poor neighborhood; b) along a main road in town.

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waste management.

Basing on a preliminary survey, the space should be integrated into the local social context in order to involve the different sections of the population (housewives, schoolchildren, university students, educators, researchers) and organized in such a way as to implement the different levels of awareness.

To promote the success of this venture, a local no-profit organization, TAM TAM Mobile (Figure 2), an apolitical company founded in 1997 by the director Simon Pierre Etonga with the specific ethic aims of contributing to environmental protection, has been involved.

TAM TAM Mobile organizes informal door-to-door solid waste collection in the poorest neighborhoods where Hysacam is not present, employing approx. 20 young operators, using hand barrows when feasible, since many houses are not accessible.

The activity is economically supported by those residents who are willing to pay small amounts for the service. Waste is collected at home and then disposed in the nearest Hysacam container along the main road: 20000 people are served in 10 different slums and 960 t/y of waste is collected. This practice is combined with an awareness-raising activity addressed to the population and local authorities on the potential risks to man and the environment linked to poor waste management.

## The literary cafe

A cultural and convivial space has been designed as a pole of interest for the citizenship, a reference point for educational institutions and a meeting place and debate on environmental issues for the population. The space has been identified within a popular district through the collaboration of TAM TAM Mobile (Figure 3).

The project was then finalized, obtaining the necessary authorizations, assessing the restoration works and selecting the furniture in such a way that the space created can provide different types of services: bar, restaurant and catering, services that can attract and welcome diversified people and that, after an initial start-up phase, facilitate economic autonomy; consultation of brochures, books and magazines at different levels of in-depth analysis of envi-


**FIGURE 2:** The TAM Tam Mobile team, Yaoundé.


**FIGURE 3:** Different phases of the literary cafe: a) the project (courtesy of arch. Rosalba Giani), b) the restoration works, c) the realisation

ronmental issues; use of computers equipped with an internet connection in order to carry out research in the relevant sector, consult online journals, join and keep in touch with thematic networks; organization of small events like public meetings to inform, seminars and discussion groups, projections of documentaries and other material relating to environmental issues, laboratory activities of sensitization.

## In action

The literary cafe opened at the end of 2015 and is managed by TAM TAM Mobile. From that moment many activities have been carried out (Figure 4).

The activities were particulary based on:

- *Awareness and information campaigns on environmental issues for all*
  Seminars and training on specific topics (municipal garbage, WEEE recycling, computer laboratory); production and distribution of informative material explaining the actions needed to prevent health risks due to poor sanitary conditions due to the huge presence of abandoned waste.
- *Education, training for women and young people in situations of vulnerability*
  To meet the most basic needs, laboratories have been set up as awareness campaigns against gender-based violence and to promote women in the business world. At the end of each event, a technical report is produced and distributed ex post to the participants highlighting the principal results obtained during the discussion that follows.

## Partners

The project has been funded by the following organisations:

- Regione del Veneto
- University of Padova
- Rotary Club of Padova
- Faber Libertatis
- IWWG-International Waste Working Group

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(a) (b)

**FIGURE 4:** Event organised in the literary café, (a) and typical final report used as follow up document (b).

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CISA

# WASTE ARCHITECTURE

## FROM A BROWNFIELD TO A RECREATIONAL FACILITY: THE CASE OF THE OLD GABORONE LANDFILL, BOTSWANA

Although landfills are a cheap and environmentally acceptable repository of waste, there are some documented post closure concerns such as visual intrusion of the adjacent land uses, particularly where such sites are in the vicinity of built up areas. For example, Schwarz et al. (2012) found that property values decrease with brownfields such as landfills for up to a mile. Furthermore, concerns over urban sprawl have made it necessary to promote sustainable land use by many European governments through land regeneration incentives and taxation (Doick et al., 2009). There is however an increasing recognition that landfill regeneration is a key element of sustainable urban development. This has by and large driven the urban revival agenda through landfill redevelopments into greenspaces. Rehabilitation of landscapes through planning and design has been found to enhance both ecological attributes and visual preferences of users groups such as property owners in their vicinity.

The old Gaborone landfill in Botswana, is one of such brownfields (see Figure 1). It covers an area of approximately 20 hectares. The landfill was operated operated beyond its design capacity, hence reaching a maximum elevation of 1018m above sea level. It is sandwiched between an office park to the west – Fairgrounds Office Park, a commercial centre to north – Riverwalk Shopping Centre, Gaborone dam to the south and Notwane River to the east. The landfill is generally barely vegetated. It is visible from most sides of its surrounding developments hence visually intruding them. However, its visual impact from the neighbouring land uses has not been objectively investigated.

Binary visibility analysis and scenario mapping were used to determine the visual impact of the old Gaborone landfill from two observation points of Fairgrounds Office Park and a point on a road that passes adjacent to it. The analysis showed that the landform is generally of high scenic quality, while the vegetation is of low scenic quality (see Table 1). Visual impact assessment on the simulated landscape following showed that the landform is generally of high scenic quality, while the vegetation is of low scenic


**FIGURE 1:** Aerial View of Old Gaborone Landfill and Disposal Site.

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**TABLE 1:** Landscape attributes.

| | Scenic quality | | |
|---|---|---|---|
| | High | Medium | Low |
| Landform | High/steep | Rolling | Flattish |
| | Isolated | Rounded | No dissections |
| | Focal points | Broad valleys | No definition |
| | Distinctive/unusual | Shallow gorges | |
| | Complex | Small rock outcrops | |
| | Incised | Regular | |
| | Strong valley form (V or U) | | |
| | Cliffs | | |
| | Ridges | | |
| | Colour contrast | | |
| Vegetation | Strongly defined | Industrial patterns | Large areas of similar vegetation |
| | Natural edges | Large clearings | No discernible patterns |
| | Mix of vegetation within communities | Coarse texture | |
| | Combination of vegetation types | Slight variations | |
| | Dramatic seasonal colour | Medium | |
| | Different shapes and sizes, some tall | | |
| | Irregular | | |

*Adapted from WEDC (2011)*

quality. In that respect, it is recommended the City of Gaborone should have a deliberate policy to convert brownfields such as the old Gaborone landfill into green fields particularly where urban sprawl has consumed almost all green spaces.

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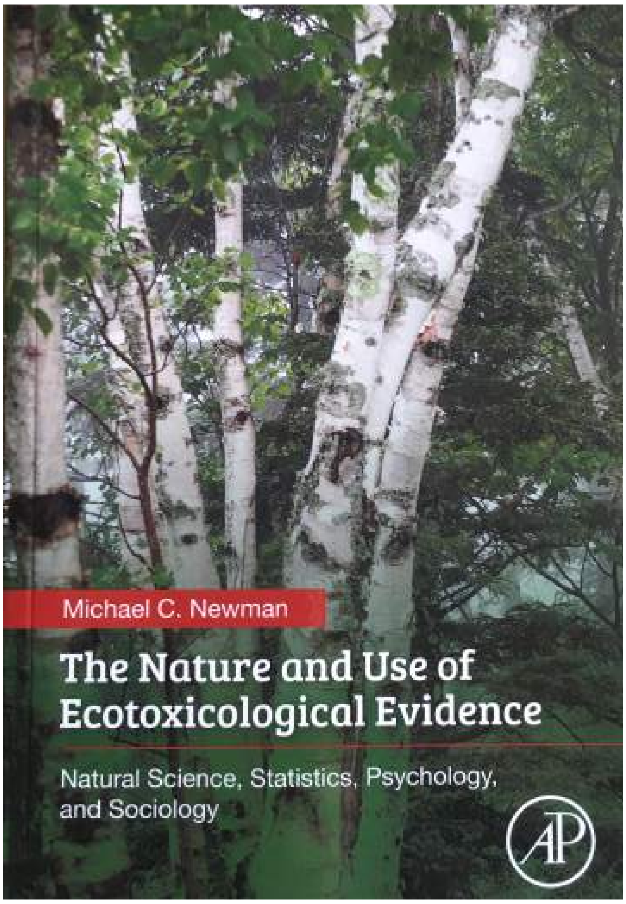
# BOOKS REVIEW




## THE NATURE AND USE OF ECOTOXICOLOGICAL EVIDENCE: NATURAL SCIENCE, STATISTICS, PSYCHOLOGY, AND SOCIOLOGY

**by Michael C. Newman**

"Weight of evidence" (WOE) is a commonly used term in scientific literature, particularly in forensic sciences, environmental risk assessment and in policy decision-making processes. Its concept, however, is often misunderstood. In fact, as Weed (2005) reported in his review on the use of WOE in literature, three main interpretations can be identified: (1) metaphorical, where it refers to a collection of studies or to an unspecified methodological approach; (2) methodological, where it points to established interpretative methodologies or implies that "all" rather than some subsets of the evidence is examined, or rarely, where it points to quantitative methods for evidence estimation; and (3) theoretical, where it serves as a label for a conceptual framework.

In the field of ecotoxicology the book of "The Nature and Use of Ecotoxicological Evidence: Natural Science, Statistics, Psychology, and Sociology" by Michael Newman clearly analyses this problem, introducing correlated topics and original considerations on the role of social dynamics. I agree with the author when he states: "the most serious impediments to wise action" such as actions to reduce chemical pollution "are the misconstruing of evidence by the scientific community and miscommunicating evidence to regulators and the public. …What evidence comes to dominate the exchange among scientists, regulators and decision makers depends on both scientific soundness and social circumstances".

The book contains 9 chapters grouped into four broad sections. To avoid conceptual dissonance, most chapters include brief overviews of the relevant concepts.

Section I is an "introduction" illustrating the history of pollution and the reasons why timely and sound evidence is now absolutely essential for human wellbeing.

Section II focuses on individuals and starts with a chapter commenting on a series of tendencies (twenty-seven!!) with the potential of compromising cognition by individuals, including scientists and risk assessors. This is followed by two shorter chapters that look into how individual scientists reason, and perhaps, make errors in the process. The fourth chapter highlights statistical methods as the gold standard of objective scientific inference; several quantitative methods are commented on, including Fisherian significant testing, Nyman-Pearson hypothesis testing, confidence intervals, information-theoretic methods and Bayesian inference.

Section III, "how groups weigh and apply evidence" broadens coverage from interactions on a microlevel to those at a macrolevel or group interactions, particularly as they influence evidence-based judgments. With regard to microlevel interactions the following topics are examined: naïve realism, groupthink, satisficing and polythink; when referring to macrolevel interactions, first a basic description of types of networks is provided, and qualities and related metrics are then explored.

In Section IV the "conclusion" consists of a single chapter that brings together the most relevant points and potential remedies for the issues discussed in the book.

Finally, two appendices are included. The first examines 18 ecotoxicological innovative survey methods (at least for non-experts in ecotoxicology) used as examples to analyse how innovations enter into and move within groups. The second focuses on a series of publication indexes (h-index, Research Gate Score, etc.) for 80 anonymous ecotoxicologists used in chapter 8 as examples of social network analysis.

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In summary, there is a wealth of information within the covers of this book on how pollutant-related evidence is gathered, assessed, communicated and applied in decision-making drawing on concepts and techniques from the natural, social and mathematical sciences. I am personally convinced that reading of the book will instil in the reader an increased awareness of the suggested means of reducing impediments to our "unbiased freewill and discriminating judgment".

The well-written and carefully structured chapters comprising this volume will be of value to environmental scientists involved in issues relating to chemical pollution, including the majority of readers of Detritus Journal.

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## REFERENCE

Weed, D.L., 2005. Weight of Evidence: A Review of Concept and Methods. Risk Anal. 25, 1545–1557. https://doi.org/10.1111/j.1539-6924.2005.00699.x.

## ABOUT THE AUTHOR
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# A PHOTO, A FACT, AN EMOTION

## RECYCLING IN LEON, NICARAGUA

**Timothy Bouldry, USA**

*"In Nicaragua, most dumpsites are unregulated and open for entry. They usually have a community attached to the dump where parents and children wait for the trucks so that they can have first pick of the recyclables. This photograph was taken in Leon, Nicaragua where there was a recent fire that set all the trash ablaze including the little sorting plant they attempted to create in an effort to formalize jobs for the "waste pickers" or informal recyclers. Sites like these occur from lack of funding, incorrect budgeting, corruption and/or neglect.*

*This dumpsite is called El Relleno or "The Fill" and it one of hundreds of unregulated sites in Latin America. 2015".*

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This photo won the first prize of Waste to Photo 2017, the photo contest connected to the Sardinia Symposium, International Waste Management and Landfill Symposium organised by IWWG.

Photography is a powerful means of communication, of visual expression, as well as an extraordinary form of art available to all and capable of evoking emotions, portraying social transformations, denouncing serious situations or acting as a testimony to important changes.

Waste to Photo, born in 2015, aimed at recreating a scenario representing the global situation with regard to waste and landfill.

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## About the Author

**Timothy Bouldry**

Timothy Bouldry photographs, explores and educates people about open dumpsite activity and the communities living from them. He works with activists, scientists, environmentalists and humanitarians to help create cases for governmental powers to understand the changes these places need. He currently resides in Nicaragua where he is photographing and running scholarship programs for kids living at these dumpsites.

*(Contents continued from outside back cover)*

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