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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

# FROM 3R TO 3S: AN APPROPRIATE STRATEGY FOR DEVELOPING COUNTRIES

It is an acknowledged fact that the quality and generation rate of municipal solid waste (MSW) is largely linked to the lifestyle, welfare and cultural level of a society, with a production per capita ranging indicatively from 0.1 kg MSW/d in low income rural areas to 4.5 kg MSW/d in urbanized industrialised areas of the world (The World Bank, 2018). Social and economic development are even more crucial with regards to waste management strategies and related technologies, although a series of other factors may play an important role (availability of land and energy, climate conditions, education, public opinion attitude, etc.).

On an international level, the classification of countries with regard to their economic level of development remains an open issue, largely due to the difficulties in defining concepts such as poverty, financial constraints, and conditions of development. Not wishing to enter into a discussion on these aspects of classification, in this note the Authors focus on areas presenting jointly critical economic constraints and poor waste management systems. These areas are generally characterized by a fast-growing population, high level of urbanization, lack of modern infrastructures, highly inhomogeneous level of education, inadequate public administration, and frequent political instability.  Areas featuring these characteristics can be identified with the so-called "Low Income Countries" but also with areas potentially present in countries with a more favourable classification.

In these areas waste management is generally characterized by the following features:

- Disposal facilities represented substantially by open dumps or poorly engineered and managed landfills;
- Uncontrolled waste burning;
- Widespread littering, very low waste collection coverage and precarious waste transport vehicles;
- Recovery of valuable waste resources by the informal sector (informal recycling and scavenging).

Under these conditions, environmental and health issues are of high concern (quality of drinking water, air quality, degradation of the urban environment, surface and ground water pollution, GHG (greenhouse gas) emissions, spread of infectious diseases, hazards for the scavengers, etc.).

Similar problems were also encountered in the past in wealthy, industrialized countries, although the situation has changed dramatically in recent decades due to the progressive increase of public awareness and perception of environmental issues, and scientific developments. These developments have focused prevalently on addressing a series of fundamental ecological issues (limited resources, climate change, widespread diffuse contamination, demographic growth, depletion of non-renewable energy sources, availability of land, etc.).

Nowadays, an environmentally-sound waste management system should satisfy the following requirements (Cossu, 2009a):

- Decrease in waste production;
- Efficient service of collection and disposal;
- Optimisation of material resource recovery;
- Minimisation of GHG emissions;
- Reduction of landfilled waste volumes;
- Optimisation of energy balance (reduction of energy consumption/waste to energy options);
- Reduction of emissions;
- Monitoring of toxicological effects and minimization of health risks, environmental sustainability.

These requirements should represent the conceptual guide for waste management in any corner of the world, irrespective of the level of economic development. Naturally, these requirements will need to be integrated into and evaluated in the various geographic contexts, taking into account economic, social and geomorphologic situations which may exert a strong influence on any choice.

The industrialized countries have attempted to meet the above-mentioned requirements by establishing a wide variety of approaches and technologies. Hierarchical Waste Management, zero-waste, Circular Economy, 3R (Reduce, Reuse, Recycle) are among the most popular concepts which currently contribute towards shaping national regulations. However, the practical application of these approaches has frequently been characterized by demagogueries, contradictory aspects, waste of economic resources, complicated and costly technologies, political speculation, misinformation of the public opinion, etc. (Cossu, 2009b, 2014, 2016, 2018).

Accordingly, the transfer of strategies and technologies from industrialized to developing countries should be carefully managed to avoid failures and mistakes and prevent export of outdated models or inappropriate or obsolete technologies.

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Transfer of proper management and technologies are generally hindered by several reasons:

- low education at different levels, resulting in unskilled technicians and widespread lack of environmental awareness;
- political instability with failure of long-term MSW management actions;
- MSW management is not always a high priority for local and national policy makers and planners;
- a scarce awareness of administrations with regard to the basic needs of the population and a lack of willingness to promote appropriate actions;
- ineffective institutional structures and pervasive corruption;
- inappropriate international funding and loans which support projects in the short-term, thus preventing the successful transfer of the project to the local authorities in the long-term;
- implementation of technologies of the highest standards, the operations of which are subsequently prevented due to lack of spare parts and/or well-trained personnel.

In line with the above considerations, when the circumstances are premature for the application of the 3R concept as part of a Circular Economy strategy, a 3S (Sanitisation, Subsistence economy and Sustainable landfilling) strategy should be implemented. The 3S approach, at variance with the 3R concept, is not perceived as a hierarchical structure, but rather is based equally on all three pillars (Figure 1).

*Sanitisation* aims to improve the standards of living in the country, achieving basic rules of hygiene in waste management.

In those countries in which people can count on a limited economical availability to support MSW tariffs, health and environmental protection constitutes a priority objective to be pursued beyond material and energy recovery. An inadequate waste disposal on the city streets entails a direct contact between wastes and the population. The population is therefore exposed to health issues including injury, diarrhoea, respiratory disorders and viral conditions, which are exacerbated by surface and groundwater contamination, air pollution from uncontrolled waste incineration, and soil contamination from leaching. The establishing of a stable waste collection system removes the waste from the residential areas, thus avoiding health issues. "Nothing is cheaper than not collecting solid waste" (Hoornweg et al., 1999).

*Subsistence Economy* is aimed at returning waste to the economy as a resource through the use of appropriate technologies, providing economic profits and new business opportunities and involving the informal sector activity in a remunerated and formalized way.

A robust and sustainable MSW management system should be designed and sized to meet local needs, at least over the medium-term. It should be resilient to political interferences and be flexible to further developments (e.g. market, technology, social). Custom-made technologies in line with social, cultural, economic and local requirements should be identified, being robust and well-proven, suited for management by local people.

Spontaneous recycling practices only occur when economically viable. Waste pickers worldwide are largely informal individual workers who are not supported by the government or included in insurance schemes or social welfare; they create an opportunity for self-employment in very difficult working conditions, strongly dependent on their capacity to sell collected material on a highly precarious market. In the presence of an informal sector, it is fundamental to involve these individuals in the operation of an MSW management system. The role of local authorities is critical in this context as solutions should be discussed and planned with the active involvement of the different stakeholders. Successful initiatives are represented by the organisation of informal recycler cooperatives (Gutberlet, 2015).

*Sustainable Landfilling* is needed to safely dispose of residues devoid of any economical or technical value.

Open dumps still constitute the most prevalent type of disposal facilities in developing countries, entailing a low level of technology and operational cost requirements. Open dumps are characterised by a lack of barriers for leachate containment and biogas control, uncontrolled waste discharge, presence of scavengers and uncontrolled waste burning to reduce the waste volume. This type of disposal results in environmental and health risks. Although awareness is increasing amongst both the public and politicians with regard to this dangerous situation, it is still insufficient and the achievement of sustainability remains a crucial challenge. Sustainable landfilling should be designed to reduce the emission potential in the long-term and to achieve an acceptable equilibrium with the environment within the span of one generation (30-40 years). In the presence of limited technical and economic situations, the following aspects should be integrated: low cost solutions in terms of development, operation and maintenance; simple, easily-implemented technologies, and maximum utilisation of natural resources and in situ materials (Lavagnolo M.C., 2018).

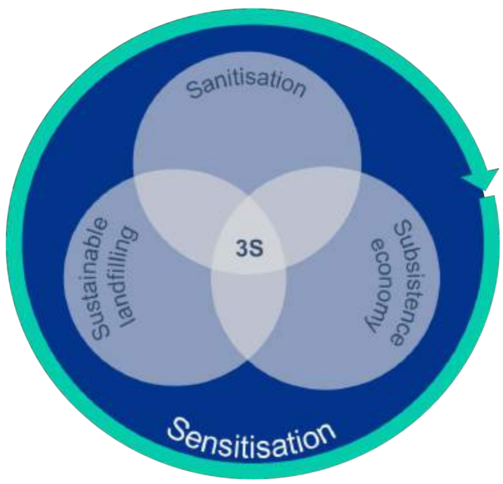


**FIGURE 1:** Graphical scheme of the 3S model proposed as a strategic tool to address the actual requirements of waste management in areas with economic constraints.

*Sanitisation, Subsistence economy and Sustainable landfilling* should be considered as complementary principles, the integration of which is strongly advocated. Sanitisation cannot be achieved in the absence of safe allocation of the collected waste. The recovery of valuable resources, which are removed from the main waste stream, reduces the volume and improves the quality of the disposed waste (e.g. treatment of food waste by means of composting or anaerobic digestion), thus promoting the landfill sustainability concept. Simultaneously, the safe disposal of worthless materials is ensured by Sustainable landfilling. Waste collection and organisation of the informal sector must be designed so as to achieve both sanitisation and recovery of valuable materials, thus supporting the local trade sector.

An essential tool for ensuring the successfullness of the whole 3S strategies is represented by the *Sensitisation* process of the local human resources.The lack of awareness of the stakeholders, mainly population and administrators, may lead to the absence of an active participation and to the inevitable failure of any attempt at implementing a sustainable SWM system.  An educational program should be carried out throughout the entire process, at different levels (schools, public administration, workers, citizens, etc.) using all media supports in order to reach the highest number of people (educational activities with children, local radio, social media by electronic devices, social events involving the community, seminars, etc.) An example of a successful initiative is represented by the establishment of a literary cafè in Youndé (Cameroun) as a meeting point for the sharing of knowledge and points of view on sustainable waste management (Lavagnolo and Failli, 2018).

Low income countries are in an ideal position to advance the most modern ideas in waste management, particularly by learning from the mistakes of the "developed" world. Indeed, in the near future we might reach the paradoxical realisation that a rich country is in many ways poor and, vice versa, a poor country is in many ways rich.

Maria Cristina Lavagnolo, Valentina Grossule
*University of Padova, Italy*
mariacristina.lavagnolo@unipd.it

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# TESTING OF 24 POTENTIALLY HAZARDOUS WASTES USING 6 ECOTOXICOLOGICAL TESTS

Jörg Römbke *

*ECT Oekotoxikologie GmbH, Böttgerstr. 2-14, D-65439 Flörsheim, Germany*

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**ABSTRACT**

The ecotoxicological characterization of wastes according to the European Waste List (EWL) is part of their assessment as hazardous or non-hazardous. Despite inclusion in national laws no methodological details have been fixed concerning the hazard property HP 14 ("ecotoxic").This paper intends to discuss the classification of wastes by ecotoxicological testing, using 24 representative samples of solid wastes (identified by their EWL number) with different properties. They were sampled according to standard methods, and, with one exception (galvanic sludge), ecotoxicologically tested. No chemical investigation of the samples was performed but they were characterized according to their properties in the ABANDA data base. Nearly all of these wastes were "mirror entries" in the EWL (i.e., they can be hazardous or not depending on the concentration of hazardous substances). For the ecotoxicological characterization three aquatic tests with eluates (genotoxicity, Algae, Daphnia) as well as three terrestrial tests with solid wastes (bacteria, plants, earthworms) were conducted. All investigations were performed as limit tests with three dilution steps. Algae, plants and terrestrial bacteria were the most sensitive organisms. Since no waste eluate showed any indication of genotoxicity, the genotoxicity test should be replaced by the luminescent bacteria test (ISO 11348-3). Proposals for toxicity criteria as well as hazard classifications were taken from the literature but they were modified according to own experiences. Using these concentration limits for the classification whether these wastes are ecotoxic or not, and using different versions of the hazard classification approach, 15-19 waste samples out of 23 waste samples were classified as ecotoxic (64-83%). It is proposed to perform a plausibility check of the respective HP 14-classification. The procedure used in this contribution (i.e. sampling of the wastes, their ecotoxicological testing as well as their hazard classification) could form the basis of a standardized hazard classification approach as proposed recently in the literature. In summary, this works confirms that ecotoxicological tests are practical and sensitive in order to be used for the ecotoxicological hazard classification of very different wastes.

## 1. INTRODUCTION

In the European Union, the hazard properties of wastes have to be determined following Commission Regulation (EU) N° 1357 (EU 2014). However, until quite recently it was not specified how the HP 14 property ("ecotoxic": waste which presents or may present immediate or delayed risks for one or more sectors of the environment) has to be assessed, but this situation has changed (see Council Regulation (EU) 2017/997). In fact, two approaches are possible:

1. Evaluation according to the classification of chemical mixtures (Classification, Labelling and Packaging CLP approach - EC 2008).
   The hazard of a waste is calculated based on its composition, i.e. adding-up the concentrations of all chemicals with chronic aquatic toxicity (Council Reg. 2017/997 (EU 2017)). Details on the pros and cons of this approach are, for example, given by Wahlström et al. (2016). However, wastes contain many, often unknown chemicals, and even in case they are known, it is not certain that ecotoxicological data are available for them (Eurelectric 2016). In addition, no interactions between the individual waste components are considered. However, according to the recent Council Regulation in case ecotoxicological tests were performed with a respective waste their results will prevail (EU 2017).
2. Ecotoxicological testing of wastes, using standard ISO (International Organization for Standardization) methods.

* Corresponding author:
Jörg Römbke
email: j-roembke@ect.de


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Originally, this approach was developed for the assessment of contaminated soils (see e.g. ISO 15799 (2002f) and ISO 17616 (2008b). In short, it consists of a battery of aquatic and terrestrial tests, whose results are assessed together. Recent discussions focus on the selection of the appropriate leaching method as well as on questions regarding the interpretation of the results (Wahlström et al. 2016). Recently, this approach has been (slightly modified) taken over in the Technical Report "Guidance on the use of ecotoxicity tests applied to construction products" products (CEN/TR 17105 (2017)).

Such an ecotoxicological test battery has been successfully used for different waste materials in France (Pandard et al. 2006), Germany (Römbke et al. 2009; Moser et al. 2011) and in particular in an international ring-test (Moser and Römbke 2009). One outcome of this ring-test was to use three test methods for waste eluates and three tests for solid wastes, but there was still some doubt which test methods exactly should be included in the test battery. For reasons of acceptance and data quality standardized test methods should be used (preferably, either ISO, CEN (Comité Européen de Normalisation) or DIN (Deutsches Institut für Normung)). It is also recommendable for any test battery to cover species from the three main trophic organism groups (microbes, plants and animals) in order to cover a wide range of physiological and ecological properties. Finally, the number of three test methods per compartment (aquatic/eluates and terrestrial/solid wastes) seems to be a good compromise between general coverage and practicability (i.e. the efforts in terms of time and costs are manageable).

However, a broad comparison of the toxicity of many different waste types investigated with exactly the same methods had not been done so far. Therefore, a project was funded by the German Federal Environmental Agency (UBA), whose results are summarized in this contribution. The aims of this work can be summarized as follows:

- Evaluation of the suitability (i.e. practicability, sensitivity, reliability and robustness) of six standardized test methods when used for the hazard classification of 24 waste samples which differ strongly in their physico-chemical and toxicological properties;
- Comparison of different assessment options using the whole data set from this exercise (including recommendations for a specific option);
- Comparison of the results of these tests with those tests proposed by Pandard and Römbke (2013); note that they recommend the luminescent bacteria test (ISO 11348-3 (2007)) but in this contribution a genotoxicity test was used instead of the bacterial luminescent test (see also Figure 1 and Chapter 4.2). This proposal is based on discussions in Workgroup 7 of CEN/TC 292/WG 7 "Characterization of waste - Ecotoxicological properties". It consists of an attempt to combine the two approaches for assessing the ecotoxicological hazard of wastes.

So far it is not known how many wastes would be classified as hazardous when their classification is based on ecotoxicological tests. The work presented here is intended to provide a first answer to this question by testing a high number of very different wastes in parallel.

## 2. MATERIAL AND METHODS

### 2.1 Tested wastes and their properties

In close co-operation with the German Federal Environment Agency (UBA) the wastes to be studied were identified using the following criteria:

- Not classified as hazardous;
- Economically relevant, mainly in terms of their amount;
- Broad coverage of the List of Wastes (EC 2000);
- Problematic due to variable composition with potentially hazardous properties;
- Difficult to classify as ecotoxic regarding the HP 14 property.

When applying these criteria, it quickly became clear that – for different reasons – it was often difficult to get the wastes we had selected, partly because they were not available in Germany or the owners were reluctant in providing them. In Table 1, the outcome of the selection process is listed. Wastes already classified as hazardous according to one of the other 13 hazard criteria, were – with one exception – not considered, since in these cases the fulfilment of the property HP 14 would not change the already existing classification as being "hazardous". It was planned to test all 24 wastes listed here, but when getting the waste classified as Code No. 110110 it was realized that it was in fact a highly condensed but still fluid galvanic sludge. Since this sample could not be tested in ecotoxicological tests, the final number of tested waste samples was 23.

Ideally, waste samples are characterized chemically and physically. However, due to the heterogeneous composition of these materials this is a very exhaustive and expensive exercise. Therefore, the origin of these waste types was compiled from the waste owners, while information on major contaminants were taken from a database named ABANDA (organized by the German state of North-Rhine-Westphalia) (Table 1). However, this information is not specific for the individual sample tested.

### 2.2 Sampling of wastes

Collecting representative samples from heterogeneous composite wastes still poses several difficulties. Here the definitions provided by the German "Federal Working Group on Waste (LAGA)" were used, which focus on the material quality of a waste as part of its characterization (LAGA 2004). A compilation of the currently available documents and recommendations in the context of waste testing are combined in the method collection of the LAGA-Forum "waste testing" (LAGA 2012). Thus, the following procedure was used for the sampling for ecotoxicological testing of wastes (Römbke and Ketelhut 2014).

In order to minimize the risk that an analytical result is

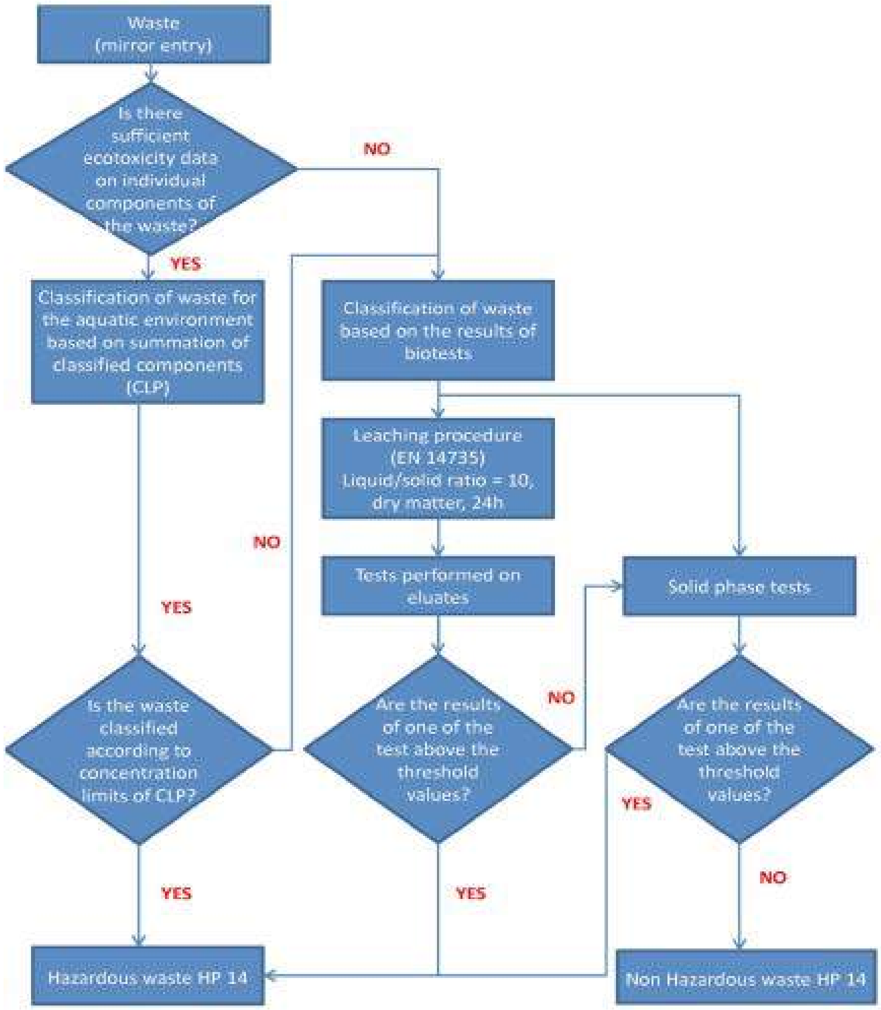


**FIGURE 1:** Flowchart for the assessment of the HP 14 property (Pandard and Römbke, 2013).

strongly influenced by singular particles samples should offer a tight spectrum of particle size. This could be proven by sieve analysis which is recommended to be done for any sample taken from heterogeneous waste materials:

- In a laboratory sample the mass above the 20th percentile of the sieve analysis should be represented by more than 20.000 particles.
- Any sampling should ideally be performed at random points in time across the whole transversal section of the particle mass flow falling from a conveyor belt. In case this is not possible the sampling should be performed from the heap of waste.
- When taking samples from a heap of waste, it should be secured that no phase separation did occur during the set-up of the heap.
- Independently from the size of the basic sample, at least 16 single samples have to be taken.
- The individual samples should be random samples. In other words: each particle of the basic population should have the same probability to be part of an individual sample.
- Sampling from a heap of waste could be performed using a wheel loader. The 16 samples taken should be combined to a two-dimensional flat layer providing a height of 1-1.5 dm. Samples could be randomly taken from random coordinates of this two-dimensional layer.
- All individual samples should be combined to one mixed laboratory sample.
- A sample size reduction without a previous reduction in particle size is not allowed.
- The addition of preservatives (e.g. acids) for the purpose of delaying chemical and biological processes does not conform to the standard CEN 14735 (2005).

The duration of the transport of waste samples was as short as possible (i.e. less than 48 h) and the samples were not stored for longer than two months (temperature: ≤ 8°C).

## 2.3 Test organisms and test performance

### *2.3.1 Aquatic tests*

The aquatic test methods used in this study are briefly described in Tables 2-4.

### *2.3.2 Terrestrial tests*

The terrestrial test methods used in this study are briefly described in Tables 5-7.

**TABLE 1:** Wastes tested, classified according to the Code of the European Waste List (EC 2000). * = waste classified as hazardous since it fulfills one of the 14 hazard criteria. One sample could not be tested (given in italics). Conc. = concentration. Note that most of these wastes could be classified as either hazardous or non-hazardous (i.e. "mirror entries"), with two absolute non-hazardous exceptions and one absolute hazardous exception. For "mirror entries", the most relevant code has been underlined according to their origin and contamination, when possible.

| Code | Waste type | Origin and Contamination |
|---|---|---|
| 01 05 05* | Oil containing drilling mud and wastes | Soil soaked with mineral drilling oil; probably high PAH conc. |
| 06 03 15*/ 06 03 16 | Metallic oxides | Waste from titan dioxide production; pH ca. 4; Cd, Cr, Cu, Ni, Zn content possible |
| 08 01 15*/ 08 01 16 | Aqueous sludges containing paint or varnish | Coating and point remnants from a car body shop, high Zn content, probably low biocide conc. |
| 10 01 16*/ 10 01 17 | Fly ash from co-incineration | Fine dust from electric filters in a coal plant. Very high Pb, Cu, Zn conc. |
| 11 01 09*/ 11 01 10 | Sludges and filter cakes | Waste water concentrate from a print shop (fluid galvanic sludge) |
| 12 01 16*/ 12 01 17 | Waste blasting material | Waste blast dust plus paint remnants from the body of a ship; Pb, Th, Zn conc. high |
| 17 01 06*/ 17 01 07 | Mixtures of concrete, bricks, tiles and ceramics | Mixed construction waste. Low PAH conc. |
| 17 02 01/ 17 02 04* | Wood | Old wood from construction sites; Pb + Zn contamination, paint remnants |
| 17 05 03*/ 17 05 04 | Soil and stones | Soil from construction sites |
| 17 05 05*/ 17 05 06 | Dredging spoil | Dredged material from Hamburg harbor. Probably contaminated by organo-tin-substances. |
| 17 05 07*/ 17 05 08 | Track ballast | Stoney material from rail tracks. High Cu, Pb, Zn + PAH conc. |
| 17 08 01*/ 17 08 02 | Gypsum-based construction materials | Fine gypsum material from construction waste sites, partly mixed with paper remnants |
| 17 09 03*/ 17 09 04-A | Mixed construction and demolition wastes (we got two of these samples; this was identified as A) | Mineral, metallic and woody mixture from waste containers, very heterogeneous, but no contaminants. |
| 17 09 03*/ 17 09 04-B | Mixed construction and demolition wastes (we got two of these samples; this was identified as B) | As sample 17 09 04-A, but higher plastic content and less insulation material |
| 19 08 02 | Waste from de-sanding | Material from waste water channels, mixed with polymer flocking agent. High PAH, Cu, Zn contamination |
| 19 08 13*/ 19 08 14 | Sludges from other treatment of industrial waste water | Filter press sludges from a painting plant; no specific contaminants |
| 19 10 03*/ 19 10 04 | Fluff-light fraction and dust | Selected light material from a reprocessing plant. High conc. of various heavy metals, incl. Hg |
| 19 12 05 | Glass | Origin: TV screens. High Pb conc. |
| 19 12 06*/ 19 12 07 | Wood (wastes from the mechanical treatment of waste) (we got three of these samples; this one did not get an additional identifier) | Wooden ULD pallets; maybe low Cr conc. |
| 19 12 06*/ 19 12 07 | Wood (wastes from the mechanical treatment of waste) (furniture) (we got three of these samples; this one was identified as A) | Construction and furniture material, 5 years stored; maybe low Cr conc. |
| 19 12 06*/ 19 12 07 | Wood (wastes from the mechanical treatment of waste) (mixed) (we got three of these samples; this one was identified as B) | Community storage pile, age unknown maybe low Cr conc. |
| 19 12 11*/ 19 12 12 | Other wastes (including mixtures of materials) from mechanical treatment of wastes | Mainly styrol-based plastic particles, few organic or metal parts; extremely heterogenous; high conc. of heavy metals possible, mainly Zn |
| 19 13 01*/ 19 13 02 | Solid wastes from soil remediation | Soil material strongly contaminated by PAH and mineral oil, but also Pb, Zn, Cu, Cr and PCB |
| 20 01 37*/ 20 01 38 | Wood | Wooden bulk trash, maybe low Cr conc. |

## 2.4 Control and mixture media in the tests

Ecotoxicity testing of waste requires the use of a dilution medium which does not affect the response of the test organisms and does as little as possible interact with the sample. The same medium must be used for both the control and the dilution series (see also CEN 14735 (2005)). Depending on the ecological requirements of the test species and the requirements listed in the ISO-standard different control and mixture media were used.

### *2.4.1 Aquatic tests*

The aquatic organisms were tested with eluates, which were prepared according to CEN 12457-2 (2003), i.e. with a

**TABLE 2:** Umu genotoxicity test (ISO standard 13829 (ISO 2000).

| | |
|---|---|
| **Test system:** | *Salmonella choleraesius subsp. choleraesius (formerly: Salmonella typhimurium)* TA 1535/pSK1002 |
| **Test duration:** | 4 h |
| **Test parameter:** | Comparison of the induction of the umuC-gene in comparison to spontaneous activations in the negative control |
| **Threshold value:** | Induction rate (IR) ≥ 1.5 |
| **Test medium:** | Tryptone, glucose, ampicillin (TGA) medium |
| **pH (control):** | 7.0±0.2 |
| **Temperature:** | 37 ± 1°C |
| **Light conditions:** | Darkness |
| **Test vessels:** | 96 well microtitration plates (optical clear) |
| **Volume / vessel:** | 380 µl |
| **Validity criteria:** | Minimum growth in the negative control = 140 FNU (formazine nephelometric units) |
| **Reference chemical / Positive control:** | 4-Nitro-quinolin-N-Oxid, 2-Aminoanthracen |

**TABLE 3:** Green algae growth test (ISO standard 8692 (ISO 2004a).

| | |
|---|---|
| **Test system:** | *Pseudokirchneriella subcapitata* |
| **Test duration:** | 72 h (permanently shaking) |
| **Test parameter:** | Growth in comparison to control |
| **Threshold value:** | 25% |
| **Test medium:** | Mixture of four nutrient stock solutions in water |
| **pH (control):** | 8.1 ± 0.2 |
| **Temperature:** | 21-24°C (fluctuations < 2°C) |
| **Light conditions:** | 60-120 $\mu E{*}m^{-2}s^{-1}$ permanent light |
| **Test vessels:** | 300 ml Erlenmeyer flasks (microtiter plates) |
| **Volume / vessel:** | 100 ml water |
| **Validity criteria:** | Increase of cell density in the controls by a factor of 67 after 72 h (i.e. growth rate ≥ 1.41); increase of pH ≤ 1.5 during the test; coefficient of variation in the control ≤ 5% |
| **Reference chemical / Positive control:** | Potassium dichromate or 3,5-Dichlorophenol? |

**TABLE 4:** *Daphnia magna* test (ISO standard 6341 (ISO 2012).

| | |
|---|---|
| **Test system:** | *5 juvenile Daphnia magna* (age 2-26 h) per replicate |
| **Test duration:** | 24 h |
| **Test parameter:** | Immobilization of the water flea |
| **Threshold value:** | 20% |
| **Test medium:** | Reconstituted water according to OECD 203 (1992) |
| **pH (control):** | 7.5-8.0 |
| **Temperature:** | 20 ± 2°C |
| **Light conditions:** | Permanently dark |
| **Test vessels:** | 50 mL vessels without lid |
| **Volume / vessel:** | 20 mL eluate/water mixture or water (controls) |
| **Validity criteria:** | Mortality in the control ≤ 10% |
| **Reference chemical / Positive control:** | Potassium dichromate |

solid/liquid dilution ratio of 1:10. The elution medium was distilled water and an end-over-end tumbler was used. After 24 h, the eluate was centrifuged for 20 min at 17000 x g and finally it was filtered (< 0.45 µm). In the Luminescent Bacteria test the control culture medium is TGA, consisting of tryptone, glucose and ampicillin. In the Algae test the control growth medium is a mixture of four nutrient stock solutions in water, which are defined as follows: No. 1: five macro-nutrients, No. 2: Fe-EDTA; No. 3: seven trace elements; No. 4: $NaHCO_3$; all of them in specific concentrations. In the *Daphnia* test the test medium is reconstituted water, which is a mixture of four nutrient salts in deionized water (Calcium chloride, magnesium sulfate, sodium bicarbonate, and potassium chloride) in specific ratios.

**TABLE 5:** *Arthrobacter globiformis* test (ISO standard 18187 (ISO 2016)).

| | |
|---|---|
| **Test system:** | *Arthrobacter globiformis* (freeze-dried) |
| **Test duration:** | < 1 d (Incubation time 2 h) |
| **Test parameter:** | Dehydrogenase activity |
| **Threshold value:** | 30% |
| **Test medium:** | Mixtures of Quartz sand and waste material |
| **pH (control):** | 5.0-7.5 |
| **Moisture:** | 20% (up to 33% possible) |
| **Temperature:** | 30 ± 1°C |
| **Light conditions:** | Dark |
| **Test vessels:** | 24-well microplate |
| **Volume / vessel:** | 0.6 g weighed in a micro-well |
| **Validity criteria:** | Relative fluorescence increases by a factor > 5 during a measuring time of 0 to 60 min. Coefficient of variation for the average slope of relative fluorescence in the negative control replicates is less than 15% |
| **Reference chemical / Positive control:** | Benzalkonium chloride (BAC) (600 mg/kg) causes effects between 30% and 80% |

**TABLE 6:** Higher plant test (ISO standard 11269-2 (ISO 2004b)).

| | |
|---|---|
| **Test system:** | *Brassica napus* (turnip), 10 seeds per replicate (4 replicates per dilution step) |
| **Test duration:** | 14-21 d after 50% of seeds in the control emerged |
| **Test parameter:** | Determination of the emergence rate within the first week. At the end of the test determination of the fresh weight and visible damages |
| **Threshold value:** | 30% |
| **Test medium:** | Mixtures of LUFA 2.3 standard soil and waste material |
| **pH (control):** | Not specified |
| **Moisture:** | On demand |
| **Temperature:** | 25 ± 10°C |
| **Light conditions:** | Light/dark cycle: ca. 16/8 h; Light intensity: 13000 ± 5000 lx |
| **Test vessels:** | Plastic pots, diameter about 10 cm |
| **Volume / vessel:** | 900 g soil / soil-waste mixture (fresh weight) |
| **Validity criteria:** | Emergence rate in the control: > 70% |
| **Reference chemical / Positive control:** | EC50 (boric acid): 80 - 330 mg/kg soil (dry weight) for the endpoint shoot weight (see also Becker et al. 2011) |

**TABLE 7:** Earthworm avoidance test (ISO standard 17512-1 (ISO 2008a)).

| | |
|---|---|
| **Test system:** | 10 adult *Eisenia fetida* (biomass 250 – 600 mg/worm) per test vessel; 5 replicates per dilution step |
| **Test duration:** | 48 h |
| **Test parameter:** | Avoidance behavior determined at the end of the test |
| **Threshold value:** | 80% |
| **Test medium:** | Mixtures of OECD Artificial Soil and waste material |
| **pH (control):** | 5.5-6.5 |
| **Moisture:** | 40-60% of the WHCmax |
| **Temperature:** | 18-22°C |
| **Light conditions:** | 16 h light (400-800 Lux), 8 h dark |
| **Test vessels:** | Bellaplast vessels, 11 x 15.5 x 6 cm |
| **Volume / vessel:** | 500 g soil / soil-waste mixture |
| **Validity criteria:** | Mortality in the control ≤ 10% per vessel; distribution with same soil on both sides: 50±10% (see also Hund-Rinke and Wiechering 2001) |
| **Reference chemical / Positive control:** | Boric acid at 750 mg/kg soil (dry weight) should cause avoidance behavior |

*2.4.2 Terrestrial tests*

In the Bacteria-test the control was quartz sand (with 50% to 75% of sand with particle size between 0,063 mm and 2 mm). The natural standard soil LUFA Soil 2.3 was used in the Plant test, which fulfilled the following conditions: organic carbon content ≤ 1.5%, pH between 5.0 and 7.5, and the fine fraction should comprise less than 20% of the soil dry weight. In the Earthworm Test the control soil was OECD Artificial Soil, consisting of 10% dry mass Sphagnum peat finely ground and with no visible plant remains (particle size < 1 mm), 20% of Kaolinite clay containing not less than 30% kaolinite and 69% industrial quartz sand (dominant fine sand with more than 50% to 75% of particle size 0.0563 mm to 0.2 mm).

### 2.5 Test design

All tests were performed following an Extended Limit Test design, i.e. with three dilutions of the tested waste eluate or solid waste. These concentrations differed between the two compartments as follows:

- Aquatic tests: control (0%), D8 (= 12.5%), D4 (= 25.0%), D2 (= 50.0%).
- Terrestrial tests: control (0%), D16 (= 6.25%), D8 (= 12.5%), D4 (= 25.0%).

Note that due to technical reasons (number of wells on a micro-well plate) the dilution steps differed from the rest in the genotoxicity tests: Control (0%), D12 (= 8.3%), D6 (= 16.7%), D3 (= 33.3.0%), D 1,5 (66.6%).

These dilutions were chosen in order to include the general limit concentration LID (= Lowest Ineffective Dilution) of 4 and 8 for aquatic and terrestrial tests, respectively. This approach is widely used in Germany for the assessment of contaminated land but has rarely been used in other countries (e.g. ISO 17616 (2008). The reason for the different dilutions in the aquatic and terrestrial tests is caused by the lower availability of contaminants in the solid test media.

### 2.6 Threshold (reference) values as effect criteria for the individual tests

Depending on the biological variability of each test system the effect criterion (i.e. which difference between a tested mixture and the respective control is considered as an effect) differs too (Tables 2-8). Originally, these criteria have been proposed for the evaluation of contaminated soil, partly in the test standard itself (e.g. ISO 17512-1 (2008a), partly in regulatory documents (e.g. Moser 2008) or, just as an example, in other international standards (e.g. ISO 17616 (ISO 2008b)). These threshold (or reference) values are used in order to decide whether a waste sample tested at a specific dilution has ecotoxic effects or not.

## 3. RESULTS

### 3.1 Aquatic tests

*3.1.1 Umu genotoxicity test*

In Table 9, the results of these genotoxicity tests are summarized. All tests were valid according to the ISO standard. In one test (No. 08 01 16) the induction rate could not be determined due to cytotoxicity. All tests were performed with (+S9) and without (-S9) metabolic activation.

*3.1.2 Green algae growth test*

The results of the Algae tests are given in Table 10. All tests were valid according to the ISO standard. Effects of up to 100% were found in all dilution steps in four samples (Nos. 06 03 16; 08 01 16; 19 08 14; 19 12 12. In addition, complete inhibition was found in the samples 10 01 17 and 19 12 05 in the two higher solutions (D2 and D4). No effects on Algae did occur in the samples 01 05 05, 17 01 07, 17 05 04 and 17 05 08. Regularly, dose-response relationships were observed.

*3.1.3 Daphnia magna test*

An overview of the results of tests with water fleas is given in Table 11. All tests were valid according to the ISO standard. The daphnids reacted most strongly in four samples (Nos. 08 01 16; 10 01 17; 12 01 17; 19 12 05). Very rarely – actually just once (No. 06 03 16) - a dose-response relationship was visible. In all other samples, no – or almost none (No. 19 12 07-A) – effects on daphnids did occur. No other test showed such a strong dichotomy: either a waste did strongly affect the test organisms or no effect at all was observed.

*3.1.4 Summary of aquatic results*

In Table 12 all aquatic results are summarized. In the genotoxicity tests no effects at all were observed. In contrast, out of 23 waste samples 13 of them were identified as ecotoxic in the Algae tests. The results of the daphnid tests were different – only five samples have to be classified as ecotoxic when using this test system.

**TABLE 8:** Overview on the effect criteria for the individual tests as given in the literature, to be used as threshold (or reference) values for the ecotoxicological hazard assessment of wastes. These dilution rates of waste in the culture medium and these biological effects are taken as reference in this study to classify waste as ecotoxic.

| Test name and guideline | Ecotoxic if |
|---|---|
| Umu test (ISO 13829 (2000)) | IR > 1.5 at dilution 25% (LID 4) |
| Algae test:(ISO 6341 (1996)) | Effect > 25% at dilution 25% (LID 4) |
| Daphnia magna test (ISO 8692 (2004a)) | Effect > 20% at dilution 25% (LID 4) |
| Arthrobacter globiformis test (ISO 18187 (2016)) | Effect > 30% at dilution 12.5% (LID 8) |
| Plant growth test ISO 11269-2 (2008b)) | Effect > 30% at dilution 12.5% (LID 8) |
| Earthworm Avoidance Test (ISO 17512-1 (2008a)) | Effect > 80% at dilution 12.5% (LID 8) |

**TABLE 9:** Induction rates of the umuC-Gen (without pH adjustment) in the genotoxicity test with waste eluates of 23 different waste materials. Effect criterion: IR ≥ 1.5. * No IR determined because of cytotoxicity. N.d. Not determined. Tests showing effects at dilution step 6 or higher are indicated as dark-shaded. S9: rat liver extract; used for metabolic activation of the bacteria.

| | Waste code | | Dilution steps [waste eluates] | | | | $LID_U$-Value |
|---|---|---|---|---|---|---|---|
| | | | **D12** [8.3%] | **D6** [16.7%] | **D3** [33.3%] | **D1.5** [66.6%] | |
| Induction rate | **01 05 05** | -S9 | 0.51 | 0.49 | 0.54 | 0.48 | <1.5 |
| | | +S9 | 0.69 | 0.77 | 0.73 | 0.97 | |
| | **06 03 16** | -S9 | 1.25 | 1.37 | 1.40 | 1.42 | <1.5 |
| | | +S9 | 1.04 | 1.01 | 1.12 | 1.03 | |
| | **08 01 16** | -S9 | * | * | * | * | n.d. |
| | | +S9 | * | * | * | * | |
| | **10 01 17** | -S9 | 0.57 | 0.95 | 1.00 | 1.10 | <1.5 |
| | | +S9 | 1.02 | 1.05 | 0.99 | 1.06 | |
| | **12 01 17** | -S9 | 0.62 | 0.78 | 0.81 | 0.76 | <1.5 |
| | | +S9 | 0.84 | 0.93 | 1.03 | 1.00 | |
| | **17 01 07** | -S9 | 0.85 | 0.9 | 0.8 | 0.86 | <1.5 |
| | | +S9 | 1.01 | 1.04 | 0.91 | 1.03 | |
| | **17 02 01** | -S9 | 1.06 | 1.15 | 1.19 | 1.17 | <1.5 |
| | | +S9 | 1.09 | 1.14 | 1.19 | 1.03 | |
| | **17 05 06** | -S9 | 1.04 | 1.01 | 0.88 | 0.93 | <1.5 |
| | | +S9 | 1.00 | 1.12 | 0.95 | 0.91 | |
| | **17 05 04** | -S9 | 0.94 | 1.14 | 1.11 | 1.08 | <1.5 |
| | | +S9 | 1.14 | 1.10 | 1.00 | 1.08 | |
| | **17 05 08** | -S9 | 1.11 | 1.20 | 1.18 | 0.95 | <1.5 |
| | | +S9 | 0.94 | 0.79 | 0.67 | 0.60 | |
| | **17 08 02** | -S9 | 1.11 | 1.01 | 0.92 | 1.12 | <1.5 |
| | | +S9 | 1.13 | 1.04 | 0.92 | 0.74 | |
| | **17 09 04-A** | -S9 | 0.83 | 0.91 | 0.78 | 1.27 | <1.5 |
| | | +S9 | 1.00 | 1.05 | 1.06 | 1.02 | |
| | **17 09 04-B** | -S9 | 0.84 | 1.04 | 0.96 | 1.06 | <1.5 |
| | | +S9 | 0.85 | 1.06 | 1.34 | 1.27 | |
| | **19 08 02** | -S9 | 1.45 | 0.71 | 0.91 | 0.75 | <1.5 |
| | | +S9 | 1.29 | 1.21 | 1.01 | 0.74 | |
| | **19 08 14** | -S9 | 0.47 | 0.54 | 0.52 | 0.45 | <1.5 |
| | | +S9 | 0.63 | 0.72 | 0.74 | 0.79 | |
| | **19 10 04** | -S9 | 1.07 | 1.10 | 1.12 | 1.01 | <1.5 |
| | | +S9 | 1.12 | 0.99 | 0.88 | 1.03 | |
| | **19 12 05** | -S9 | 0.98 | 0.98 | 0.94 | 1.01 | <1.5 |
| | | +S9 | 1.05 | 0.94 | 0.85 | 0.97 | |
| | **19 12 07** | -S9 | 0.91 | 1.07 | 0.92 | 0.97 | <1.5 |
| | | +S9 | 0.90 | 0.99 | 0.84 | 0.92 | |
| | **19 12 07-A** | -S9 | 0.76 | 1.09 | 1.00 | 1.14 | <1.5 |
| | | +S9 | 0.98 | 0.80 | 0.98 | 1.06 | |
| | **19 12 07-B** | -S9 | 0.38 | 1.09 | 1.00 | 1.36 | <1.5 |
| | | +S9 | 1.02 | 1.13 | 1.12 | 1.08 | |
| | **19 12 12** | -S9 | 0.38 | 1.09 | 1.00 | 1.36 | <1.5 |
| | | +S9 | 1.02 | 1.13 | 1.12 | 1.00 | |
| | **19 13 02** | -S9 | 0.97 | 1.01 | 1.00 | 1.01 | <1.5 |
| | | +S9 | 1.04 | 0.96 | 0.90 | 0.88 | |
| | **20 01 38** | -S9 | 1.02 | 0.93 | 0.89 | 0.99 | <1.5 |
| | | +S9 | 1.09 | 0.80 | 0.78 | 0.63 | |

**TABLE 10:** Inhibition (in % of the control) of the growth of *Pseudokirchneriella subcapitata* in the Algae test with waste eluates of different waste materials. Effect criterion: 25%. Tests showing effects at dilution step 8 or higher are indicated as dark-shaded.

| | Waste code | Dilution steps [waste eluates] | | | $LID_A$-value |
|---|---|---|---|---|---|
| | | **D8** [12.5%] | **D4** [25%] | **D2** [50%] | |
| Growth inhibition [%] | **01 05 05** | -16 | -15 | -5 | 2 |
| | **06 03 16** | 82 | >100 | >100 | > 8 |
| | **08 01 16** | >100 | >100 | >100 | > 8 |
| | **10 01 17** | 1 | >100 | ? | 8 |
| | **12 01 17** | 67 | 85 | >100 | > 8 |
| | **17 01 07** | -1.4 | 1.6 | 1.0 | 2 |
| | **17 02 01** | 13 | 41 | >100 | 8 |
| | **17 05 04** | -1 | -3 | 5 | 2 |
| | **17 05 06** | 0 | 9 | 53 | 4 |
| | **17 05 08** | -4 | -4 | -2 | 2 |
| | **17 08 02** | 7 | 3 | 21 | 4 |
| | **17 09 04-A** | 39 | 44 | 52 | > 8 |
| | **17 09 04-B** | 0 | 9 | 42 | 4 |
| | **19 08 02** | 5 | 18 | 71 | 4 |
| | **19 08 14** | >100 | >100 | >100 | > 8 |
| | **19 10 04** | 2 | 12 | >100 | 4 |
| | **19 12 05** | 34 | >100 | >100 | >8 |
| | **19 12 07** | 30 | 33 | 41 | >8 |
| | **19 12 07-A** | 13 | 18 | 44 | 4 |
| | **19 12 07-B** | 18 | 41 | 76 | 8 |
| | **19 12 12** | >100 | >100 | >100 | > 8 |
| | **19 13 02** | 13 | 28 | 50 | 8 |
| | **20 01 38** | 31 | 44 | 85 | > 8 |

## 3.2 Terrestrial tests

### *3.2.1 Arthrobacter globiformis test*

In Table 13, the results of the tests with this bacterial test are summarized. In one test (No. 01 05 05) an additional pasteurization was performed because of the high microbial activity of this sample. In one other test (No. 10 01 17) no dose-response relationship was observed. All tests were valid according to the ISO standard. No waste type caused a 100% effect on the dehydrogenase activity of *A. globiformis*. However, in 10 samples (Nos. 01 05 05; 08 01 16; 10 01 17; 12 01 17; 17 02 01; 19 12 07; 19 12 07-A; 19 12 07-B; 19 12 12; 20 01 38) the dehydrogenase activity was lower than the control by > 30%. In contrast, in seven samples no effect was found in all dilution steps. Dose-response relationships were almost never observed. A clear increase of the dehydrogenase active did not occur. Despite the small amount of waste in these tests (0.6 g) these results confirm the robustness of this test. However, the small sample size may have had an influence on the strong differentiation of the test results: only in two tests and LID-value of 8 was observed – all others were either very toxic or not toxic

### *3.2.2 Higher plant test (B. napus)*

In Table 14, the results of the tests with the plant *B. napus* test are summarized. All tests were valid according to the ISO standard. In three tests (Nos. 06 03 16; 08 01 16; 19 08 14) no seed germination (or at least no growth of the seedlings did occur) was observed. In addition, effects higher than 30% were observed in ten samples at all dilution steps (Nos. 01 05 05; 10 01 17; 17 02 01; 17 08 02; 17 09 04°; 17 09 04-B; 19 12 07; 19 12 07-A; 19 12 07-B; 20 01 38). In case effects did occur, they followed a dose-response-relationship. Only in eight tests no or low (i.e. <30%) effects were found. Very conspicuous is sample No. 17 05 06, (dredged material without contaminants) which caused a strong increase of growth; i.e. probably it contained nutrients.

### *3.2.3 Earthworm avoidance test*

In Table 15, the results of the tests with the earthworm *E. fetida* are summarized. All tests were valid according to the ISO standard. Only in one test an avoidance behavior of 100% in all dilutions was observed (No. 08 01 16). An avoidance behavior of more than 80% did occur in six samples (Nos. 01 05 05; 06 03 16; 10 01 17; 17 02 01; 17 09 04-A; 20 01 38). No avoidance effect was visible in six tests (Nos. 17 05 04; 17 05 06; 19 08 02; 19 13 02). No dose-response relationship was observed in four samples (Nos. 17 05 08; 19 12 07, 19 12 07-B; 20 01 38). In some tests, several samples seemed to be attractive the earthworms (No. 19 08 02 and 19 13 02) – especially at higher dilution steps (D8 and D16).

**TABLE 11:** Immobilization [%] of 20 juvenile water fleas (per test vessel) in the *Daphnia*-test with waste eluates of 23 different waste materials. Effect criterion: 20%. Tests showing effects at dilution step 8 or higher are indicated as dark-shaded.

| | Waste code | Dilution steps [waste eluates] | | | $LID_D$-value |
|---|---|---|---|---|---|
| | | **D8** [12.5%] | **D4** [25%] | **D2** [50%] | |
| **Inhibition of mobility [%]** | **01 05 05** | 5 | 0 | 0 | 2 |
| | **06 03 16** | 5 | 45 | 100 | 8 |
| | **08 01 16** | 100 | 100 | 100 | > 8 |
| | **10 01 17** | 100 | 100 | 100 | > 8 |
| | **12 01 17** | 90 | 100 | 100 | > 8 |
| | **17 01 07** | 0 | 0 | 5 | 2 |
| | **17 02 01** | 0 | 5 | 0 | 2 |
| | **17 05 04** | 0 | 0 | 0 | 2 |
| | **17 05 06** | 0 | 0 | 0 | 2 |
| | **17 05 08** | 0 | 20 | 0 | 2 |
| | **17 08 02** | 0 | 10 | 0 | 2 |
| | **17 09 04-A** | 0 | 0 | 5 | 2 |
| | **17 09 04-B** | 0 | 0 | 0 | 2 |
| | **19 08 02** | 0 | 5 | 0 | 2 |
| | **19 08 14** | 0 | 0 | 0 | 2 |
| | **19 10 04** | 0 | 0 | 0 | 2 |
| | **19 12 05** | 50 | 80 | 75 | > 8 |
| | **19 12 07** | 0 | 0 | 0 | 2 |
| | **19 12 07-A** | 0 | 0 | 75 | 4 |
| | **19 12 07-B** | 5 | 5 | 5 | 2 |
| | **19 12 12** | 0 | 0 | 0 | 2 |
| | **19 13 02** | 0 | 0 | 0 | 2 |
| | **20 01 38** | 5 | 0 | 0 | 2 |

**TABLE 12:** Results of the aquatic tests (LID-values) with waste eluates. * No IR determined because of cytotoxicity.

| Waste code | Umu-Test: $LID_U$ (*S. choleraesius*) | Algae: $LID_A$ (*P. subcapitata*) | Daphnia: $LID_D$ (*D. magna*) |
|---|---|---|---|
| **01 05 05** | <1.5 | 2 | 2 |
| **06 03 16** | <1.5 | > 8 | 8 |
| **08 01 16** | * | > 8 | > 8 |
| **10 01 17** | <1.5 | 8 | > 8 |
| **12 01 17** | <1.5 | > 8 | > 8 |
| **17 01 07** | <1.5 | 2 | 2 |
| **17 02 01** | <1.5 | 8 | 2 |
| **17 05 04** | <1.5 | 2 | 2 |
| **17 05 06** | <1.5 | 4 | 2 |
| **17 05 08** | <1.5 | 2 | 2 |
| **17 08 02** | <1.5 | 4 | 2 |
| **17 09 04-A** | <1.5 | > 8 | 2 |
| **17 09 04-B** | <1.5 | 4 | 2 |
| **19 08 02** | <1.5 | 4 | 2 |
| **19 08 14** | <15 | > 8 | 2 |
| **19 10 04** | <1.5 | 4 | 2 |
| **19 12 05** | <1.5 | > 8 | > 8 |
| **19 12 07** | <1.5 | > 8 | 2 |
| **19 12 07-A** | <1.5 | 4 | 4 |
| **19 12 07-B** | <1.5 | 8 | 2 |
| **19 12 12** | <1.5 | > 8 | 2 |
| **19 13 02** | <1.5 | 8 | 2 |
| **20 01 38** | <1.5 | > 8 | 2 |

**TABLE 13:** Inhibition of the dehydrogenase activity of *A. globiformis* in the Bacteria contact test with 23 different waste materials; effect criterion 30%. Lightly-shaded cells: no dose-response relationship. * Additional pasteurization performed. N.d.: Not determined (light-shaded cells). Tests showing effects at dilution step 16 or higher are indicated as dark-shaded.

| | Waste code | Dilution steps | | | $LID_B$-value |
|---|---|---|---|---|---|
| | | **D16** [6.25% Waste] | **D8** [12.5% Waste] | **D4** [25% Waste] | |
| Inhibition of the dehydrogenase activity [%] | **01 05 05 *** | 43.5 | 55.3 | 60.6 | > 16 |
| | **06 03 16** | 27.4 | 61.3 | 80.6 | 16 |
| | **08 01 16** | 83.5 | 91.6 | 96.1 | > 16 |
| | **10 01 17** | 98.1 | 92.7 | 81.9 | > 16 |
| | **12 01 17** | 30.7 | 43.6 | 79.6 | > 16 |
| | **17 01 07** | -1.9 | 16.6 | 34.5 | 8 |
| | **17 02 01** | 46.8 | 65.8 | 83.3 | > 16 |
| | **17 05 04** | 3.4 | -0.6 | -4.6 | 4 |
| | **17 05 06** | -3.5 | -0.9 | 5.9 | 4 |
| | **17 05 08** | -0.9 | 2.8 | 13.3 | 4 |
| | **17 08 02** | 1.3 | -5.2 | 5.3 | 4 |
| | **17 09 04-A** | 0.8 | 44.9 | n.d. | 16 |
| | **17 09 04-B** | 33.3 | 42.6 | n.d. | > 16 |
| | **19 08 02** | 10.2 | 8.4 | 37.2 | 8 |
| | **19 08 14** | 4.5 | 33.0 | 42.0 | 16 |
| | **19 10 04** | 25.3 | 33.2 | 75.1 | 16 |
| | **19 12 05** | 1.3 | 3.8 | 28.5 | 4 |
| | **19 12 07** | 58.4 | 82.1 | 85.8 | > 16 |
| | **19 12 07-A** | 60.9 | 84.9 | 94.8 | > 16 |
| | **19 12 07-B** | 75.8 | 93.9 | 96.1 | > 16 |
| | **19 12 12** | 33.7 | 52.7 | 77.3 | > 16 |
| | **19 13 02** | 8.0 | 10.7 | 13.4 | 4 |
| | **20 01 38** | 53.2 | 83.9 | 88.0 | > 16 |

*3.2.4 Summary of terrestrial results*

In Table 16 all terrestrial results are summarized. No test failed. In the *Arthrobacter* test 15 wastes were identified as ecotoxic, in the plant tests 14 and in the earthworm tests just six.

### 3.3 Summary: classification of all test results (aquatic and terrestrial tests together)

In this chapter, the results of the aquatic and terrestrial tests are presented together (Table 17). Afterwards, the assessment principles proposed by Pandard and Römbke (2013) will be applied (e.g. using the same threshold values and limit concentrations) with one exception: instead of the luminescent bacteria test (ISO 11348-3 (2007a) the genotoxicity test (ISO 13829 (2000)) was used. However, the latter one did not show any effects. Therefore, in the following the classification will be performed without the bacterial tests in order to avoid a bias when comparing the aquatic and terrestrial effects. However, and referring to Table 13, it should be kept in mind that in the terrestrial bacterial tests 15 out of 23 wastes were classified as ecotoxic.

According to the tiered approach proposed by Pandard and Römbke (2013), the results of the aquatic tests are considered first. Out of the 23 wastes tested 13 breached the LID of 4. Algae reacted more sensitively since they were affected in all these 13 cases. Only five wastes were toxic for water flea. No waste did only affect the daphnids. The different sensitivity pattern of these two organisms is also shown by the fact that the Algae were not affected at all by five wastes (LID = 2) and showed small effects in tests with another five wastes (LID = 4). In contrast, the daphnids showed a strong yes/no pattern: in 17 tests, there was no effect (LID = 2), meaning that only one waste caused a small effect. In summary, in this sample of wastes (with one exception all of them had no mirror entries) 57% of them are ecotoxic.

In the terrestrial tests 14 out of 23 wastes are considered to be ecotoxic. Again, the sensitivity of the two species differs considerably: all of these 14 waste samples were toxic to plants, but only six of them affected the earthworms strongly. Eight wastes did not affect the plants and just one caused a small effect. The respective numbers for the earthworms are 13 and four. No waste was classified as ecotoxic only in the earthworm tests. Looking only at the outcome of the terrestrial tests 61% of all wastes did affect terrestrial organisms. According to the proposed scheme, only the ten wastes evaluated as non-ecotoxic in the aquatic tests are assessed in the second step, using the results of the terrestrial tests. Five

**TABLE 14:** Reduction of biomass (in comparison to the control in %) of *B. napus* (turnip) in the plant growth tests with 23 different waste materials; effect criterion 30%. No conspicuous observations were made. * No seeds did germinate at all. Tests showing effects at dilution step 16 or higher are indicated as dark-shaded.

| | Waste code | Dilution steps | | | $LID_P$-value |
|---|---|---|---|---|---|
| | | **D16** [6.25% Waste] | **D8** [12.5% Waste] | **D4** [25% Waste] | |
| Reduction of biomass compared to control in % | **01 05 05** | 85.5 | 84.4 | 82.4 | > 16 |
| | **06 03 16** | * | * | * | > 16 |
| | **08 01 16** | * | * | * | > 16 |
| | **10 01 17** | 53.4 | 79.2 | 93.3 | > 16 |
| | **12 01 17** | -4.4 | 7.4 | -3.2 | 4 |
| | **17 01 07** | -11.8 | -15.9 | 12.4 | 4 |
| | **17 02 01** | 54.0 | 69.8 | 89.8 | > 16 |
| | **17 05 04** | -22.8 | -21.3 | -18.9 | 4 |
| | **17 05 06** | -46.0 | -61.3 | -65.7 | 4 |
| | **17 05 08** | 9.8 | 13.9 | 11.5 | 4 |
| | **17 08 02** | 36.1 | 54.6 | 63.7 | > 16 |
| | **17 09 04A** | 94.9 | 97.0 | 98.0 | > 16 |
| | **17 09 04B** | 68.7 | 72.2 | 82.5 | > 16 |
| | **19 08 02** | -2.7 | -0.6 | -2.7 | 4 |
| | **19 08 14** | * | * | * | > 16 |
| | **19 10 04** | 22.6 | 49.8 | 68.2 | 16 |
| | **19 12 05** | -1.5 | 5.5 | 7.2 | 4 |
| | **19 12 07** | 76.5 | 82.1 | 87.9 | > 16 |
| | **19 12 07-A** | 72.5 | 78.6 | 81.0 | > 16 |
| | **19 12 07-B** | 63.4 | 81.2 | 80.5 | > 16 |
| | **19 12 12** | 21.7 | 21.6 | 45.8 | 8 |
| | **19 13 02** | -31.2 | -3.6 | 28.8 | 4 |
| | **20 01 38** | 63.0 | 70.6 | 87.9 | > 16 |

out of the 10 wastes caused effects in the tests with soil organisms. Therefore, in total 18 wastes out of 23 (= 78%) are classified as hazardous following the concept of Pandard and Römbke (2013).

However, when comparing those nine cases in which either aquatic or terrestrial organisms reacted more sensitively it seems that sensitivity does not differ between both organism groups, since in five tests aquatic organisms were reacting stronger than their terrestrial counterparts – and in four tests this situation was just the other way around. Finally, the inclusion of the bacterial results should be briefly discussed. As mentioned earlier, the inclusion of the genotoxicity data (no effects at all) would not change the number of wastes classified as being hazardous from aquatic testing. Assuming that instead the luminescent Bacteria test would have been used either no change or an increase in ecotoxic wastes would happen. However, in the case of the terrestrial bacteria test we do have data (see Table 14): in 15 out of 23 tests with *A. globiformis* strong effects were found, but with two exceptions (Nos. 12 01 17 and 19 12 12) those were samples which already were identified as toxic in the plant or earthworm tests. In addition, both exceptions were also toxic to aquatic organisms, meaning that the overall results would not change.

## 3.4 Discussion of the toxicity of the wastes used in this project

Despite the fact that the quite high number of wastes tested here more data are needed in order to cover the full range of different waste types. This aim could either be reached by closing gaps regarding waste types, or by testing samples the same waste types as done here but coming from other sites in order to get an overview how much this property differs within one waste type.

The ecotoxicological test methods used here are robust, practical and reliable. Only one out of 24 samples (No. 11 01 10: highly condensed but still fluid galvanic sludge) could not be tested, partly because of its physico-chemical properties, partly because of its unknown human toxicology. Therefore, from a technical point of view the use of ecotoxicological test methods is recommended.

The results of such tests are neither correlated between them (i.e. each test has its own "effect profile") nor is the information gained redundant to information from other sources (e.g. chemistry) regarding the question whether a specific waste is classified as hazardous or not. Twenty wastes are mirror entries and can be hazardous or not depending on the concentration of hazardous substances. In contrast, in ecotoxicological tests almost 75% all test-

**TABLE 15:** Avoidance behavior (in comparison to the control in %) of the earthworm E. fetida in the earthworm avoidance tests with 23 different waste materials; effect criterion 80%. All tests were valid according to the ISO standard. Lightly-shaded cells: no dose-response relationship. Tests showing effects at dilution step 8 or higher are indicated as dark-shaded.

| | Waste code | Dilution steps | | | $LID_R$-value |
|---|---|---|---|---|---|
| | | **D16** [6.25% Waste] | **D8** [12.5% Waste] | **D4** [25% Waste] | |
| Avoidance behavior [%] | **01 05 05** | 88 | 92 | 100 | > 16 |
| | **06 03 16** | 68 | 88 | 100 | 16 |
| | **08 01 16** | 100 | 100 | 100 | > 16 |
| | **10 01 17** | 72 | 92 | 100 | 16 |
| | **12 01 17** | 14 | 68 | 88 | 8 |
| | **17 01 07** | -2 | 40 | 88 | 8 |
| | **17 02 01** | 52 | 82 | 88 | 16 |
| | **17 05 04** | 0 | 4 | 32 | 4 |
| | **17 05 06** | -24 | 40 | 20 | 4 |
| | **17 05 08** | 48 | 40 | 36 | 4 |
| | **17 08 02** | 10 | 52 | 56 | 4 |
| | **17 09 04-A** | 60 | 64 | 94 | 8 |
| | **17 09 04-B** | -32 | 44 | 52 | 4 |
| | **19 08 02** | -68 | -68 | -40 | 4 |
| | **19 08 14** | 16 | 28 | 56 | 4 |
| | **19 10 04** | 28 | 76 | 90 | 8 |
| | **19 12 05** | -10 | 8 | 30 | 4 |
| | **19 12 07** | 20 | 66 | 52 | 4 |
| | **19 12 07-A** | -34 | -16 | 4 | 4 |
| | **19 12 07-B** | 52 | -28 | 24 | 4 |
| | **19 12 12** | 22 | 36 | 46 | 4 |
| | **19 13 02** | -60 | -22 | 21 | 4 |
| | **20 01 38** | 44 | 84 | 78 | 16 |

ed wastes are classified as hazardous. So, it seems that this kind of testing is more sensitive in identifying wastes which could be ecotoxicologically hazardous.

Right now, it is impossible to say whether the results of this study are representative for the relationship between ecotoxicological testing and the classification of wastes according to the List of Wastes in general. However, further ecotoxicological tests with a broader range of wastes, trying to cover the range of the waste types and subtypes, are recommended in order to get a better understanding of the hazard properties of wastes. In case such studies will be performed a chemical characterization of the test samples is highly recommended. Assuming that such a data set will be available, the suitability and robustness of the different classification approaches could be assessed and recommendations for legal handling could be formulated.

## 4. DISCUSSION

### 4.1 Test performance, species selection and species sensitivity

*4.1.1 Test substrate characterization*

There is a need for an improved description of the sampling, handling (especially the pre-treatment, e.g. the particle size) and storage of waste samples before they are used in ecotoxicological tests. Without that kind of information comparability of results is hampered. In addition, the properties of the individual waste samples should be characterized as good as possible, both in terms of their physical appearance as well as their chemical composition. Data from general data bases (as used here) are not sufficient. Despite the fact that each waste sample by definition differs from each other such information could be used to understand better the reasons for ecotoxicity of waste samples.

*4.1.2 Selection of the ecotoxicological test methods and species*

Regarding the composition of the test battery it is recommended to follow all recommendations of Pandard and Römbke (2013). This includes the testing of waste eluates with the bacterial luminescent test (ISO 11348-3 (2007a) instead of the umu genotoxicity test (ISO 13829 (2000), as originally proposed after the ring test (Moser & Römbke 2009), for the following reasons:

- Genotoxicity is a very specific endpoint which seems to be rarely relevant for wastes, at least to my experience no sample tested in our lab did show any signs of genotoxicity;

**TABLE 16:** Results of the terrestrial tests (LID-values) with solid wastes.

| Waste code | Arthrobacter Test: $LID_B$ (*A. globiformis*) | Higher Plant Test: $LID_P$ (*P. subcapitata*) | Earthworm Avoi-dance Test: $LID_R$ (*E. fetida*) |
|---|---|---|---|
| **01 05 05** | > 16 | > 16 | > 16 |
| **06 03 16** | 16 | > 16 | 16 |
| **08 01 16** | > 16 | > 16 | > 16 |
| **10 01 17** | > 16 | > 16 | 16 |
| **12 01 17** | > 16 | 4 | 8 |
| **17 01 07** | 8 | 4 | 8 |
| **17 02 01** | > 16 | > 16 | 16 |
| **17 05 04** | 4 | 4 | 4 |
| **17 05 06** | 4 | 4 | 4 |
| **17 05 08** | 4 | 4 | 4 |
| **17 08 02** | 4 | > 16 | 4 |
| **17 09 04-A** | 16 | > 16 | 8 |
| **17 09 04-B** | > 16 | > 16 | 4 |
| **19 08 02** | 8 | 4 | 4 |
| **19 08 14** | 16 | > 16 | 4 |
| **19 10 04** | 16 | 16 | 8 |
| **19 12 05** | 4 | 4 | 4 |
| **19 12 07** | > 16 | > 16 | 4 |
| **19 12 07-A** | > 16 | > 16 | 4 |
| **19 12 07-B** | > 16 | > 16 | 4 |
| **19 12 12** | > 16 | 8 | 4 |
| **19 13 02** | 4 | 4 | 4 |
| **20 01 38** | > 16 | > 16 | 16 |

- At the time of the European Waste Ringtest there was no real alternative for a standardized microbial test in soil available.

This situation has changed considerably: the Arthrobacter Test, originally developed for sedimenst, was modified successfully in order to perform it both in soils and wastes (e.g. Marques et al. 2018). Still, the genotoxicity test is an option for those wastes where there are hints that the waste to be tested may contain genotoxic components. Focusing on the aquatic compartment Weltens et al. (2012) proposed it as screening tests for the hazard classification of wastes in a similar test battery, i.e. the Algae growth inhibition test, the Daphnia immobilization test and, in addition, the fish larval mortality test. These authors discuss the luminescent Bacteria test as a fast alternative.

*4.1.3 Sensitivity of the selected ecotoxicological tests*

Looking at the outcome of this project it could be argued that it would be sufficient to use in the future only those tests which have been most sensitive. In detail, the overall number of wastes affecting organisms would not change if only the Algae and the Arthrobacter tests would have been performed here. However, this conclusion is premature because it is based on just 23 waste samples. Experiences from other regulatory areas, in particular the testing of chemicals, has shown that it is not possible to identify "the most sensitive species" because it simply does not exist (Cairns 1986).

When studying different wastes in the international ring test (Moser and Römbke 2009) or fly ashes (Römbke and Moser 2007) with the umu test rarely genotoxicity was found. However, Brackemann et al. (2000) report genotoxic reactions in an acid eluate prepared from stoker-fired furnace ash as well as in three wastes from chemical industry. Since genotoxicity is a very important endpoint the low number of data should not be taken as an excuse to disregard this test.

The ecological relevance and sensitivity of the Algae test is often considered as high (Deventer et al. 2004). For example, they reacted often more sensitively in eluates from different wastes (mainly ashes) than daphnids or luminescent Bacteria (Kaneko 1996; Lapa, 2002a; Lapa 2002b). Therefore, they are regularly proposed as part of an ecotoxicological test battery for wastes (e.g. Pandard et al. 2006; Moser and Römbke 2009). However, it is known that the two species recommended in the Algae tests could react differently when exposed to the same waste (Moser and Römbke 2009).

Tests with the water-flea *Daphnia magna* have often been used for waste testing, in particular with waste incineration ashes (e.g. Kaneko 1996; Triffault-Bouchet et al. 2003; LFU 2004; Pandard et al. 2006; Römbke and Moser 2007). Results from the international ringtest confirm the

**TABLE 17:** Classification of the individual tests using the threshold values and limit concentrations described above. Grey cells: Ecotoxicological effects higher than the test-specific threshold values did occur at the respective limit concentrations (LID aquatics = >4; LID terrestrial > 8). Black cells: ecotoxicologically hazardous according to the HP 14 property. Note that the bacterial tests were not included since the umu-test did not show any effects.

| Waste code | Aquatic tests | | | Terrestrial tests | | | Overall Hazard evaluation |
|---|---|---|---|---|---|---|---|
| | $LID_A$ | $LID_D$ | Ecotox. LID > 4 | $LID_P$ | $LID_R$ | Ecotox. LID > 8 | |
| 01 05 05 | 2 | 2 | | > 16 | > 16 | | |
| 06 03 16 | > 8 | 8 | | > 16 | 16 | | |
| 08 01 16 | > 8 | > 8 | | > 16 | > 16 | | |
| 10 01 17 | 8 | > 8 | | > 16 | 16 | | |
| 12 01 17 | > 8 | > 8 | | 4 | 8 | | |
| 17 01 07 | 2 | 2 | | 4 | 8 | | |
| 17 02 01 | 8 | 2 | | > 16 | 16 | | |
| 17 05 04 | 2 | 2 | | 4 | 4 | | |
| 17 05 06 | 4 | 2 | | 4 | 4 | | |
| 17 05 08 | 2 | 2 | | 4 | 4 | | |
| 17 08 02 | 2 | 2 | | > 16 | 4 | | |
| 17 09 04-A | > 8 | 2 | | > 16 | 8 | | |
| 17 09 04-B | 4 | 2 | | > 16 | 4 | | |
| 19 08 02 | 4 | 2 | | 4 | 4 | | |
| 19 08 14 | > 8 | 2 | | > 16 | 4 | | |
| 19 10 04 | 4 | 2 | | 16 | 8 | | |
| 19 12 05 | >8 | > 8 | | 4 | 4 | | |
| 19 12 07 | >8 | 2 | | > 16 | 4 | | |
| 19 12 07-A | 4 | 4 | | > 16 | 4 | | |
| 19 12 07-B | 8 | 2 | | > 16 | 4 | | |
| 19 12 12 | > 8 | 2 | | 8 | 4 | | |
| 19 13 02 | 8 | 2 | | 4 | 4 | | |
| 20 01 38 | > 8 | 2 | | > 16 | 16 | | |

high practicability and sensitivity of this test system. It was by far the most often performed test but at the same time the one with the lowest number of invalid data sets (Moser and Römbke 2009). In the light of these experiences the low sensitivity in this study is difficult to explain. It might be that water flea react mainly to heavy metals (Seco et al. 2003), which – by chance – were not so often occurring in the 23 wastes tested.

Despite the fact that the Arthrobacter test has been used for quite some time (mainly in sediments) it has only been standardized for soils quite recently. Therefore, the number of experiences in waste testing are limited, mainly with incineration ashes (Deventer et al. 2004; Römbke et al. 2009). Positive experiences in the international ring test (Moser and Römbke 2009) and in a recent interlaboratory comparison test (Marques et al. 2018). In particular its high practicability (short duration, low costs, high sensitivity) has increased its usage.

When testing the effects of contaminant soils on plants often the dicotyledonous species *Brassica rapa* (or *B. napus*) is reacting most sensitively (Wilke et al. 1998; Kalsch et al. 2006b). In addition, the test is very robust, meaning that it is a regular part of terrestrial ecotoxicological test batteries. In fact, while the number of plant test with wastes is still limited it is the test most often performed with solid waste samples. Again, different ashes are the best studied samples (Wong and Wong 1989; Deventer al. 2004; Römbke and Moser 2007). It could also be shown that ashes with different physico-chemcial properties do cause different effects on plants (Quilici et al. 2004).

The earthworm avoidance test has been developed and standardized about ten years ago (ISO 17512-1 (2007)), i.e. the amount of data regarding its use for waste testing is limited. However, when used in the international Ringtest it became clear that it is much more sensitive than the earthworm acute test (ISO 11268-1 (1993)), which was proposed earlier for this task (Moser and Römbke 2009). However, the results were often variable – an observation which is known from tests with contaminated soils (Hund-Rinke et al. 2003). This might be caused by the fact that the worms do not only react to toxic contaminants but also to physico-chemical properties (Natal-da-Luz et al. 2008). Kobeticova et al. (2010) confirm the suitability of the earthworm avoidance test for waste evaluation, also pointing out that other oligochaete species such as enchytraeid seem to be less sensitive.

Huguier et al. (2015) studied a wide range of (mainly) organic wastes in tests with various aquatic and terrestrial species (partly with more than one endpoint), trying to identify a suitable test battery. Plants (*Avena sativa, Brassia rapa*) and earthworms (*Eisenia fetida*), seem to be suitable for this specific group of wastes. In addition, the authors confirm the good comparability of results from avoidance and reproduction earthworm tests. This information supports the choice made here but cannot easily be transferred to wastes in general.

The information summarized in this chapter confirms that a battery of six test methods (plus, if needed, a genotoxicity test) selected by Pandard and Römbke (2013) is needed for the ecotoxicological characterization of wastes since they react sensitively to different stressors and their interactions.

## 4.2 Test design, threshold (reference) and limit values

### *4.2.1 Test design*

Both an "Extended Limit Design" (i.e. testing three (or more) dilution steps with fixed ratios) as well as the EC-approach do allow an ecotoxicological classification of wastes.

In Germany, the LID-approach (= Lowest Ineffective Dilution) is widely used for the evaluation of waste waters or contaminated soils, partly because the effort needed is relatively low (e.g. only three dilution steps are needed) (e.g. DiBt 2008). In order to assess the ecotoxicological hazard of a waste sample it is necessary to define a limit concentration, usually given in percent of the overall tested amount (e.g. 12.5% = LID 8). These limit concentrations cannot be defined based on test results but must be set-up before testing, using the following criteria:

- The limit concentration should be practical when classifying wastes;
- They must be protective, i.e. hazardous wastes must be clearly identifiable.

So far, legally no limit concentration has been fixed. Unfortunately, only few proposals have been found in the literature, mainly addressing aquatic tests (e.g. Kostka-Rick 2004b; DIBt 2008). However, based on these hints and the experiences made in the European Ringtest on Wastes, a LID = 4 for the aquatic tests and a LID = 8 for the terrestrial tests was proposed as being acceptable (Moser & Römbke 2009). The LID-approach did work well for this testing and assessment program, but it has its limitations (e.g. it highly depends on the concentrations tested). In any case it was possible to differentiate ecotoxic and non-ecotoxic wastes. Beyond this yes/no-decision it is also possible to assess, how toxic the respective waste or the specific test organism is, since three dilution steps were used. Only results from the umu- test are difficult to be assessed, since genotoxicity seems to be less dose-dependent than other endpoints. In addition, this test reacts only to specific contaminants, making it less sensitive in general.

In ecotoxicology in general the ECx approach is more common since it allows a more detailed but also robust assessment of ecotoxicological effects – as long as the whole response curve (ideally from 0 to 100%) is covered. In order to improve the robustness of this classification the use of an ECx design (aiming on the calculation of an EC50 value) would be better. In such a case, more concentrations than just three as in the "Extended Limit Approach" have to be tested (e.g. up to eight). However, when doing so the number of replicates per concentration is lowered, meaning that the overall testing effort would not increase very much. Pandard and Römbke (2013), recommending an ECx-design (i.e. an $EC_{50}$ as limit concentration), could show that such an approach is reliable and protective. First experiences show that the ecotoxicological characterization of wastes do not differ much between these two methods (Pandard and Römbke 2013). However, the number of such comparisons is still very low. Therefore, from a scientific point-of-view the determination of an EC50 is the better and more robust option, being in-line with other areas of ecotoxicology (e.g. the risk assessment of chemicals).

### *4.2.2 Threshold (reference) values*

The threshold values for the individual tests (e.g. 20% effect on the main test endpoint such as immobility in the Daphnia-test) used here are based on ideas firstly published by Moser (2008). Partly they are already mentioned in the respective test standards, partly they are specified in scientific publications focusing on waste classification. Ideally, such threshold values should be based on statistical considerations, but the respective comparable data sets are not (yet) available. Therefore, experiences with soil tests have been used too. In fact, there is a grey zone between contaminated soils and at least some wastes (e.g. in this project: No. 19 13 02: Solid wastes from soil remediation other than those listed in 19 13 01. Soil material strongly contaminated by PAH and mineral oil, but also Pb, Zn, Cu, Cr and PCB).

Independently whether the LID- or the ECx-approach is used there is always a third level of decision-making in case a test battery is used: in how many tests have the threshold values to be breached? For example, in order to characterize a waste as being hazardous at least one test has to show an effect on one aquatic or one terrestrial species. Another possibility would be to require effects on one species from each of the three taxonomic groups independently in which medium they had been tested. At this point, again the proposal of Pandard and Römbke (2013) is followed, meaning that a tiered approach is used. One aquatic or one terrestrial test has to show effects higher than the respective threshold value in order to classify the tested waste as hazardous according to the HP 14 property.

There has also been no criterion fixed legally so far regarding the outcome of the whole test battery. Previously – and following a recommendation given in ISO 17616 (ISO 2008b) for the assessment of contaminated soils – it was assumed that a waste is classified as "ecotoxic" in case the threshold values have been breached in one out of six tests. Later on, Pandard and Römbke (2013) modified this strategy (which works well as long as the number of aquatic and terrestrial tests is equal) in a way that the process is divided into two parts:

1. Assessment based on aquatic biotests: If one of the LID values in the eluate tests (the IR value of > 1.5 in the umu-test would be handled like a LID-value > 4) is above the proposed threshold values of 4 or 8, respectively, the waste is classified as hazardous and the overall procedure is stopped.
2. Otherwise, solid waste tests are carried out and the assessment procedure is repeated. The waste is considered as non-hazardous if the results of all biotests are below or equal to the threshold values.

### 4.3 Comparison of different approaches regarding the ecotoxicological characterization of wastes

The chemical composition of the tested wastes is not known. Therefore, it is impossible to calculate whether the respective waste sample has to be classified as "ecotoxic" or not. Alternatively, the List of Waste could be used as a reference, but in that list, only the "absolute" entries can be used to assess the proposed dilutions, meaning that a classification of the waste samples tested here is also not possible.

From the results of the tests (Table 17) and the proposition of dilution and effect rates for ecotoxicity classification (Table 8), 18 wastes are classified as ecotoxic (17 mirror entries and one hazardous 01 05 05*), and four wastes are classified as non-ecotoxic (three mirror entries and one non-hazardous 19 08 02). In other words, one hazardous waste is classified as ecotoxic, one non-hazardous waste is classified as non-ecotoxic and 20 "mirror entries" are classified 17 times as ecotoxic and three times as non-ecotoxic. Right now, it is impossible to say whether the results of this study are representative for the relationship between ecotoxicological testing and the classification of wastes according to the List of Wastes. However, further ecotoxicological tests with a broader range of wastes, trying to cover the range of the waste types and subtypes, are recommended in order to get a better understanding of the hazard properties of wastes. In case such studies will be performed a chemical characterization of the test samples is highly recommended. Assuming that such a data set will be available, the suitability and robustness of the different classification approaches could be assessed and recommendations for legal handling formulated.

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# ADVANCING THE CIRCULAR ECONOMY THROUGH GROUP DECISION-MAKING AND STAKEHOLDER INVOLVEMENT

John Bachér *, Hanna Pihkola, Lauri Kujanpää and Ulla-M. Mroueh

*VTT Technical Research Centre of Finland, Biologinkuja 7, 02150 Espoo, Finland*

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**ABSTRACT**

This paper discusses the potential of Multicriteria Decision-Making Methods (MCDM) to support co-operation between stakeholders and the ability to tackle bottlenecks in the recycling value chains. The empirical part of the paper is based on two MCDM exercises in which bottlenecks that hinder the efficient recycling of end-of-life vehicles and plastic packaging waste were evaluated and prioritized. The interlinked nature of the recycling chains poses challenges for the application of MCDM methods, and studies that apply group decision-making in the context of the circular economy are still rare. Despite these challenges, the findings of our case study indicate that group decision-making methods could be applied as participatory methods to enable the collection and integration of stakeholder views within circular economy policy development and implementation activities. A review of literature on existing bottlenecks for the circular economy indicates that there is an increasing need for co-operation and knowledge exchange between actors. It is proposed that in the future, MCDM methods could be used to create joint learning and idea exchange between value chain actors, as these are considered necessary for advancing the circular economy.

## 1. INTRODUCTION

A key priority of the EU Circular Economy Action Plan is improving resource efficiency in the EU. The move towards a circular economy (CE) requires integrating life cycle thinking in product design and all subsequent life cycle stages in order to enable efficient recycling, recovery, repair and re-use (European Commission, 2015). The basic principles of the circular economy include creating economic growth and increasing the well-being of people, while reducing resource use and greenhouse gas emissions, and operating within the planetary boundaries (European Commission, 2015).

However, several barriers hindering the transition towards a CE have been identified. These barriers are diverse and include technical, economic, institutional and social aspects (de Jesus and Mendonça 2018). According to Preston (2012), barriers to implementing CE include inter alia failures in company co-operation and limited dissemination of innovation. Other review studies mention information deficits (Rizos et al., 2015), minor consumer and business acceptance and a lack of awareness and information (Vanner et al., 2014) as the main barriers to CE.

Since the recycling chain consists of a series of processes handled by different actors, finding potential solutions to circular economy challenges requires co-operation along the value chain (Bacher et al., 2016). From a policy maker's perspective, transition towards a CE therefore may require balancing between trade-offs in economic, social and environmental sustainability related to material recycling and potential conflicts in stakeholder interests.

Methods for Multicriteria Decision-Making (MCDM) have been developed to help decision-makers to identify and select preferred alternatives when faced with a complex decision problem that is characterized by multiple objectives (Seppälä et al., 2002). In the context of the circular economy, MCDM methods could provide potential means for prioritising policy options and alternative implementation routes, which take into account the views of various stakeholders who are affected by the policies and who are in a key position to advance the circular economy. This paper discusses the potential of structured group decision-making methods to support cooperation between stakeholders and create learning that would be necessary in order to tackle the bottlenecks faced in the recycling value chains. The empirical part of the paper is based on two MCDM exercises in which bottlenecks that hinder efficient recycling of end-of-life vehicles and plastic packaging waste were evaluated and prioritized.

Previously, MCDM methods have been used in the context of waste management, especially for selecting preferable waste management strategies and locations (Goulart Coelho et al., 2017; Morrissey and Browne, 2004; Rousis et

* Corresponding author:
John Bacher
email: john.bacher@vtt.fi

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al., 2008; Soltani et al., 2015). According to a review of 68 studies by Soltani et al., (2015), the number of MCDM studies incorporating group decision-making has significantly increased during the past decade while less attention has been given to the interaction between the stakeholders involved. The importance of the latter arises from the diverse and conflicting interests of the decision-makers involved, which makes reaching an agreement challenging. Indeed, depending on the hierarchy and competitive positions of the stakeholders, each decision-maker considers the decisions of others on top of their own preferences.

Involving multiple stakeholders in shared decision-making can have a positive impact on the results. Van den Hove (2006) stated that participatory approaches in the current MCDM methods increase the transparency and fairness of the results. In addition to building consensus, MCDM methods can be used to highlight the diverse interests of the participating stakeholders (Kiker et al., 2005). In a participatory process, understanding the differences in opinions and preferences may be as important as reaching consensus.

The approach presented in this paper differs from earlier studies, since the aim of the MCDM exercises was to enable more efficient recycling and circular use of resources, by evaluating the bottlenecks identified within the value chains. By prioritising the bottlenecks, future policy and research actions could be targeted to activities which would lead to the most positive impacts in the whole value chain.Thus, the focus of the studies is not only on the end of the life cycle (for example in selecting a preferred waste management system or facility location), but also on the life cycle as a whole, since the bottlenecks were located in different parts of the life cycle.

The use of MCDM methods in the context of circular economy studies is still rare, even though the amount of literature discussing CE is growing rapidly. Thus, our case study brings an additional contribution to the expanding CE research, by applying MCMD methods within the context of two recycling value chains, and it focuses the assessment on the level of a supply chain. In their analysis of altogether 565 CE related journal articles, Merli et al., (2018) found that the majority of the papers the evaluated were modelling studies and case studies, and that the methods most commonly applied included Life Cycle Assessment (LCA) and Material Flow Analysis (MFA) types of methods. Almost half of the studies analyzed belonged to the category "Tools, models, framework and methods for decision-making, consisting of studies developing theoretical or empirical instruments and approaches to evaluate and develop CE (Merli et al., 2018). Interestingly, MCDM methods were not mentioned among the applied methods, although providing decision support for evaluating and developing the circular economy was a popular theme. Another finding was that most CE studies were focused on the macro-level (society, country) or on the micro-level (corporate and/or consumer level), and only a small number was focused on a supply chain in which more firms would be involved (Merli et al., 2018).

While potential barriers and success factors related to CE concepts have been studied on several occasions (for a review see, e.g. Winans et al., 2017), participatory, group decision-making methods have not been used in this context. Mahpour (2018) used a similar approach and applied a MCDM method (fuzzy topis) for prioritising barriers for adopting circular economy in construction and demolition waste management. However, in this exercise, expert opinions were collected using a survey, and thus the interaction between the experts was not included in the study. Zhao et al., (2017) used a hybrid MCDM approach to evaluate and rank potential environmental, economic and social benefits of eco-industrial parks in China. In their study, expert opinions were used to build an evaluation index system based on evaluations collected from experts representing several different fields. Also in this context, expert opinions were collected using questionnaires instead of group decision-making.

Thus, the main contribution and novelty value of this paper is to discuss potential benefits and challenges when applying group decision-making methods in the context of circular economy problem solving. The paper is structured as follows: Chapter 2 describes the case study and the evaluated value chains. Chapter 3 presents the applied MCDM approach and explains how the empirical case study was conducted in practice. Chapter 4 presents the case study results and related learnings, together with the considerations related to the limitations of the study. Conclusions are presented in Chapter 5 together with the potential policy implications and recommendations, which are considered as the main contribution of this study.

## 2. CASE DESCRIPTION

### 2.1 Introduction and context

The empirical part of the paper is based on a case study in which group decision-making methods were applied to evaluate and prioritize bottlenecks within two recycling value chains: End-of-Life Vehicles (ELV) and Plastics Packaging Waste (PPW). Identification of the bottlenecks was conducted in the context of a recent EU Horizon 2020 project NEW_InnoNet. In the project, two different value chains were selected in order to understand whether the bottlenecks were specific to a waste stream or whether there were commonalities between the value chains, and consequently, if some uniform solutions could be found which might help in solving the challenges faced in the recycling chains in general, despite the waste stream.

Bottlenecks were defined as factors that limit the performance and efficiency of the value chain and thus prevent or limit the move towards a near zero-waste value chain and circular economy. As part of the study, expert workshops were organized to discuss and prioritize the bottlenecks in the evaluated value chains. In the workshops, structured group decision-making methods were applied.

### 2.2 Evaluated value chains: End-of-life vehicles and Plastic packaging waste

MCDM exercises were carried out for two recycling value chains: End-of-life vehicle (ELV) and plastic packaging waste (PPW). In Europe both of these waste streams are under the Extended Producer Responsibility (EPR) schemes which govern the end-of-life management through the ELV and PPW directives (Directive 94/62/EC; Directive 2000/53/EC). In general terms, value chains comprise actors that perform operations to deliver valuable concentrates/products or services to the market. Within the context of recycling and waste management, the value chain begins at the point where the consumer discards the product thus creating waste. Furthermore, if reuse is not possible, due to the loss of functionality, the theoretical value potential of waste equals the value of material. This drives the recycling industry together with increasing disposal costs, increased public concern about the health and environmental impacts of waste disposal as well as the fear of future scarcity of certain natural resources to separate valuable materials and concentrates for the refining industry for element recovery and refining (van Beukering et al., 2014). Commonly, the value chain ends at the point where material is semi-finished goods. A schematic description of a general recycling value chain is presented in Figure 1.

Typically, the first operation after discarding the product is collection, which aims at collecting maximal amount of waste to be treated properly. This step is realized by waste management companies and different collection points. In order to meet the requirements of EPR schemes, manufacturers commonly establish Producer Responsibility Organizations (PRO) which are responsible for taking back used goods and for sorting and treating for their eventual recycling (Monier et al., 2014). However, in practice PROs may purchase the collection from a third party such as a waste management company. Collection methods vary between the waste streams. For PPW it can be arranged by separate collection, deposit system, kerbside collection or at various collection points (da Cruz et al., 2014; Dahlbo et al., 2018; Groot et al., 2014; Hahladakis et al., 2018). For ELVs, the collection is often managed by specific collection points. During the ELV collection, a depollution step is usually carried out to remove fluids, tyres, batteries and other possible hazardous components which are required by the ELV directive (Directive 2000/53/EC).

After collection, sorting and mechanical treatment are carried out to separate possible components/parts for reuse and to separate valuable materials which can be different metals in the ELV value chain and different plastic types for the PPW chain for further refining as well as removing harmful components for proper disposal (Inghels et al., 2016; Shen and Worrell, 2014). Especially for the ELV value chain, parts or components may be removed for reuse or separate treatment prior to the shredding process (Andersson et al., 2017a; Inghels et al., 2016). Commonly, recycling companies are responsible for this stage. Finally, valuable concentrates from ELV recyclers are refined for intermediate products such as steel billets for the manufacturing industry. In metals production, different smelters and metallurgical companies carry out the refining (Andersson et al., 2017a). In plastics production refining includes melting or processing which is conducted by plastic converters (Shen and Worrell, 2014). In Table 1, a summary of typical characteristics of ELV and PPW streams is presented.

The nature of ELV and PPW differ significantly, especially considering the average product size and lifespan (Table 1). In addition, material composition of the waste stream is different. This together with the waste generation affects

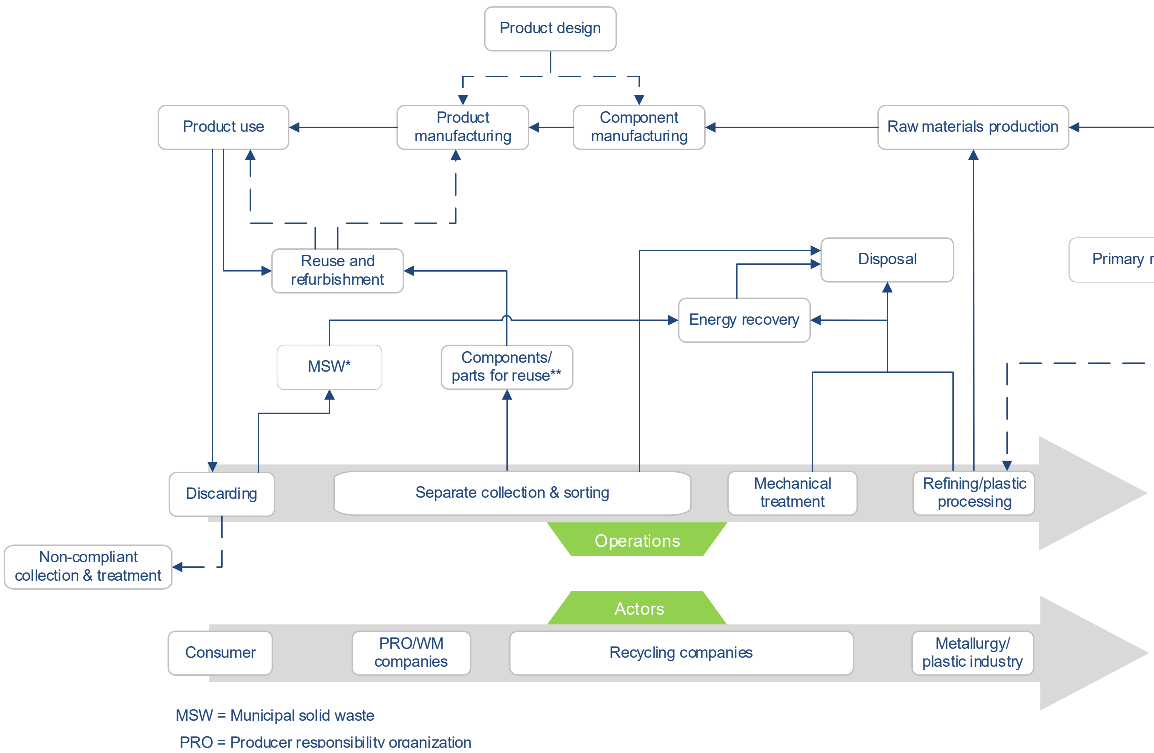


**FIGURE 1:** Overall description of a value chain within the recycling context.

**TABLE 1:** Typical characteristics of ELV and PPW value chains. (Directive 94/62/EC; Directive 2000/53/EC; Geyer et al. 2017; Eurostat, 2018a; Eurostat, 2018b; Inghels et al., 2016; Worrell, 2014).

| Feature | ELV | PPW |
|---|---|---|
| Quantity (Mt/year) 2009-2015 | 6-8 | 14.5-16 |
| Rough composition | Metals, plastics (different type), glass, wood, minerals, rubber, fluids, others | Mainly different type of plastics (PET, HDPE, PP, PVC), but may contain in small quantities other materials and residuals |
| Collection method | Specific collection points | Separate and mixed collection, refund, drop-off, curbside |
| Current recycling rate in 2015 (%) | 87.1 + | 40.2 + |
| Recycling rate target (%) | 85 * | 55 ** |
| Typical product size | Hundreds of kilograms | Below one kilogram |
| Average lifespan of the product | Over10 years | Below one year |

*+ Estimate by the Eurostat / * From year 2016 onwards / ** Proposed in the Directive 94/62/EC by the year 2025*

the value chain actors and their operations. It should be noted that the waste generation quantities in Table 1 are based on statistical data that does not take into account emerging quantities outside official monitoring. By looking at recycling rate targets, beside the high metal content of cars, the long history of ELV recycling (Andersson et al., 2017b) reflects in the high recycling rate targets compared with the PPW which has gained attention only during the last decade.

Despite the current relatively high recycling rate for ELV, complex vehicle design integrating materials together and the increasing share of light materials generate challenges for current and future recycling (Soo et al., 2017). On the other hand, increasing valuable and scarce metal concentrations due to electrification of vehicles generate opportunities for recycling scarce metals that are currently lost (Andersson et al., 2017a). To further increase the recycling rate in the PPW chain, key challenges include reaching better quality recycled materials despite the complexity of the input materials. This can be achieved through good integration of collection, recovery and separation technology (Shen and Worrell, 2014).

## 2.3 Bottlenecks identified in the value chains

Industrial experts and researchers identified the bottlenecks within the value chains prior to the prioritization exercises. The bottlenecks that hinder or limit the transition towards a zero waste society and circular economy were both technical and non-technical (financial, organizational, political, legal or societal) in nature. Further, they affected several points of the value chain. Altogether 10 bottlenecks were identified within the PPW chain, and 15 within the ELV chain. All the bottlenecks were presented and discussed during the expert panel workshops, and participating experts acknowledged the existence of these bottlenecks. However, due to the lack of background data and availability of measured information on the potential impact of bottleneck removal, only 5 out of 10 identified bottlenecks in the PPW chain and 5 out of 15 identified bottlenecks in the ELV chain could be evaluated within the MCDM exercise. In Table 2, bottlenecks that were included within the prioritization using MCDM are presented with a short description. Excluded bottlenecks are listed below the table.

Bottlenecks identified in the PPW value chain but excluded from the prioritization were:

- Lack of common calculation methodology to calculate EU recycling targets, including more measuring points (collection, sorting and recycling) to efficiently measure the material flow;
- Supply of highly heterogeneous and/or contaminated plastics from collection leading to downcycling and high rejection rates;
- Lack of market trust in products containing recycled plastics; absence of quality requirement (end of waste criteria) for recycled plastic waste, on both the supply and demand side;
- Product standards limiting the use of recycled material
- Uneven playing field for environmentally sound recycling plants because of non harmonized EU legislation in Member States and regions.

Bottlenecks identified in the ELV value chain but excluded from the prioritization were:

- Lack of accreditation and standardization for ELV recycling operators;
- Counterproductive regulation prohibiting improving recycling;
- Lack of Europe-wide harmonization in regulation, poor governance and ineffective enforcement of regulation;
- presence of unwanted substances prohibiting further qualitative application;
- Secondary material pricing is benchmarked against primary material pricing;
- Rapidly changing vehicle designs (model updates), technical compositions and higher contributions of consumers in vehicle design;
- Transportability of goods and materials - High share of small regionally operating companies in the vehicle end-of-life chain;
- Lack of stability (volume/economic) in secondary material supply chain is not motivational for material producers to integrate recycled streams;
- No incentive for manufacturers to develop recyclable products, as vehicle use emissions are dominant design choice in total environmental performance;

**TABLE 2:** Bottlenecks included in the priorization exercise. (Bacher et al., 2016).

| Waste stream | Bottleneck | Description |
|---|---|---|
| PPW | Limited source separation of plastic packaging waste | No or inappropriate systems for source separation of plastic packaging are in place. Recyclable plastics end up in residual waste stream and are diverted to the corresponding treatments (disposal/ incineration). |
| PPW | 'Bad' product design | Product design and use of composite materials are not adapted to current technologies for sorting/ separating materials. This results in plastics technically or economically not fit for recycling, leading to loss of material value and decreased yield in pre-treatment. |
| PPW | Export of plastic packaging waste for recycling outside EU | Lower costs for recycling (because of lower human and environmental standards) causes export of plastic waste. The potential to create benefits from recycling is consequently reduced in the EU. |
| PPW | Performance of separation / sorting technology | Part of the plastic packaging waste ends up in disposal or incineration due to technology not being able to sort new packaging product designs. The root causes are presence of unwanted substances in the plastic waste streams, high cost for removing them and finally product development going faster than the recycling technology development. |
| PPW | Performance of recycling technology | Part of the plastic packaging waste is ending up in disposal or incineration due to slow development in the recycling technology, reducing the yield of recycling. |
| ELV | Inadequate performance of the separation, sorting and refining technology | Vehicle (material) innovation in the construction phase leads to higher levels of intermingled, alloyed and glued material particles. New components are required to be lighter by weight, but with similar or better operational performance. In the current recycling system, this leads to a higher degree of materials with overlapping properties |
| ELV | Inadequate performance of vehicle dismantling and reuse application | Construction complexity and smart connected parts leads to higher effort required to dismantle components for a reuse application. High-voltage components require more safety measures by the collection and dismantling chain. The opportunity to dismantle parts for material recycling decreases as intrinsic material value is depleted |
| ELV | Limited and low quality application outlets of non-metallic ELV materials | Economic and technical feasibility to sort, separate and refine non-metallics is low due to the heterogeneous composition of 'shredder output'. Materials are sorted, and due to their low economic value, can only be recycled in low-grade applications |
| ELV | Inadequate performance of ELV collection and monitoring | Interpretation of what actually an ELV is, how it should be recycled and how the recycling sub quota should be monitored and judged depend on many factors. This creates unclarity for stakeholders and provides incentives for substandard treatment. It further results in a lack of reliable data availability on vehicle registration and composition, ELV arising and vehicle / ELV trade |
| ELV | Low-cost of energy recovery and landfilling alternatives compared to material recovery | In some EU Member States, overcapacities (and competition) in incineration facilities and landfill deposits, as well as low taxation rates, lead to low gate fees. This creates an uneven playing field compared to material recycling, of which the operational costs are usually higher than for incineration and disposal. |

- Total arising from ELVs is dropping and ageing, due to 'fewer accidents', higher safety measures and vehicle exports to foreign destinations (outside the EU).

The bottlenecks included in the MCDM were selected based on the ability to measure their potential impacts in quantitative terms. This was considered as a limitation, but the participants approved the decision. Practical and methodological reasons for excluding part of the bottlenecks and reducing the applied evaluation criteria are further discussed in Chapters 3 and 4.

## 3. MATERIALS AND METHODS

### 3.1 Applied MCDM methods

The ability of MCDM methods to produce viable results is based on breaking down complex problems into manageable components, which usually are, as reviewed by Goulart Coelho et al., (2017), (i) goal and scope definition; (ii) theoretical framework definition; (iii) criteria and indicators selection; (iv) data normalization; (v) weighing attribution, and (vi) sensitivity analysis. When applied for group decision-making, MCDM can be used to highlight the similarities and potential causes of conflicts between stakeholder views, and thus improve a shared understanding of the problem (Kiker et al., 2005).

There are many different systematic MCDM methods available for evaluating alternatives based on multiple criteria. However, they often share a similar approach to structuring the decision problem into a set of alternatives and a goal that can be divided into non-redundant lower level objectives and related criteria (Kiker et al., 2005). Consequently, a matrix of alternatives and their performances in each criterion can be created (Figure 2). MCDM methods (for problems where an alternative needs to be selected or ranked) can be divided into value-based, outranking-based or reference-based methods (Goulart Coelho et al., 2017). The value-based methods produce a single aggregated numerical score for each alternative on a cardinal scale, while outranking methods aim to compensate for less easily measured criteria performances by indicating the extent of how much an alternative dominates another (Kiker et al., 2005). Reference-based methods indicate the best or worst alternative by measuring their distance to an ideal or worst possible solution (Goulart Coelho et al., 2017).

Two value-based methods, the Analytic Hierarchy Process (AHP) (Saaty, 1980) and Multi-Attribute Utility and Value Theories (MAUT/MAVT) (Dyer and Sarin, 1979; Keeney and Raiffa, 1994) were used in this study. According to a recent review by Soltani et al., (2015), the Analytic Hierarchy Process (AHP) (Saaty, 1980) was by far the most commonly used MCDM method in waste management studies involving multiple stakeholders, followed the by outranking methods PROMETHEE and ELECTRE and the reference level model TOPSIS. Contrary to the preceding, however, Goulart Coelho et al., (2017) reported in their review study that the Multi-Attribute Utility and Value Theories (MAUT/ MAVT) (Dyer and Sarin, 1979; Keeney and Raiffa, 1994) were the second most common MCDM methods in waste management studies. The most common usage of AHP and MAUT/MAVT in the studies reviewed by Goulart Coelho et al., (2017) was the selection of a facility location. Both MAVT and AHP are optimization methods that produce an aggregated overall numerical score of each alternative in a single cardinal scale (Kiker et al., 2005).

In the case study, both MCDM exercises were carried out as group exercises during a two-day expert panel workshop. The time available for group discussions and reaching consensus, together with the availability of background data affected the selection of the MCDM methods.

For the plastic packaging value chain, MAVT was chosen due to availability of detailed numerical data on the bottleneck performances in a defined set of criteria. In MAVT, the Decision Maker's (DM's) preferences are modelled as value functions and their weights (Equation 1). A value function $v_i^N(x_i)$ transforms any measured variable $x_i$ to a number representing its subjective value for the DM. The value functions $v_i^N(x_i)$ were assumed linear in the study, as their detailed elicitation was judged too time-consuming for the time and resources available.

According to MAVT, the overall value of an alternative is calculated using the additive value function:

$$V(x) = \sum_{i=1}^{n} w_i v_i^N(x_i) \tag{1}$$

Where:
$V(x)$ is the overall value of an alternative,
$v_i^N(x_i)$ is the normalized value of a criterion performance of an alternative and
$w_i$ is a weight given for a criterion.

$$v_i^N(x_i^0) = 0 \tag{2}$$

and

$$v_i^N(x_i^*) = 1 \tag{3}$$

apply for the normalized value functions. $x_i^0$ equals the worst and $x_i^*$ the best performance level for each criterion (in Equations 2 and 3).

Criterion weight $w_i$ reflects the increase in overall value when the criterion performance is changed from the worst level $x_i^0$ to the best $x_i^*$. The following equation applies for the criteria weights (Equation 4):

$$\sum_{i=1}^{n} w_i = 1 \tag{4}$$

The weights $w_i$ were taken as averages of the individual DM's answers during the Expert Panel Workshops (EPWs). The DMs' preferences were elicited using trade-off weighing. The DMs were asked to compare the best possible performance in each criterion against an equally valued hypothetical performance in the most important criterion. Before presenting the trade-off questions, the most important criterion was selected as a shared decision by the expert panel.

Less data was available in the ELV value chain on the criteria performances of the bottlenecks, thus demanding expert judgement during the EPWs. Consequently, the AHP method was applied to ELV value chain in order to prioritize the identified bottlenecks. In AHP, a DM forms local priorities by comparing the importance of each same level elements (alternatives or criteria) against each other regarding each element on the next level upwards (lower levels objectives or the main objective). In the EPW on the ELV value chain, the DMs were first asked to compare the importance of each criterion in achieving the goal. Once this was done, all the alternatives (bottlenecks) were compared against each other regarding their performances in each criterion. The fundamental scale from 1 (equally important) to 9 (extremely more important) presented by Saaty (1980) was used in the comparison of same level elements.

## 3.2 Selection of the expert panel and organization of the decision-making exercises

Invitations to the workshop were sent to selected experts and organizations active within the case value chains and acknowledged by the project partners. Therefore, the input to the case study was obtained by non-ran-

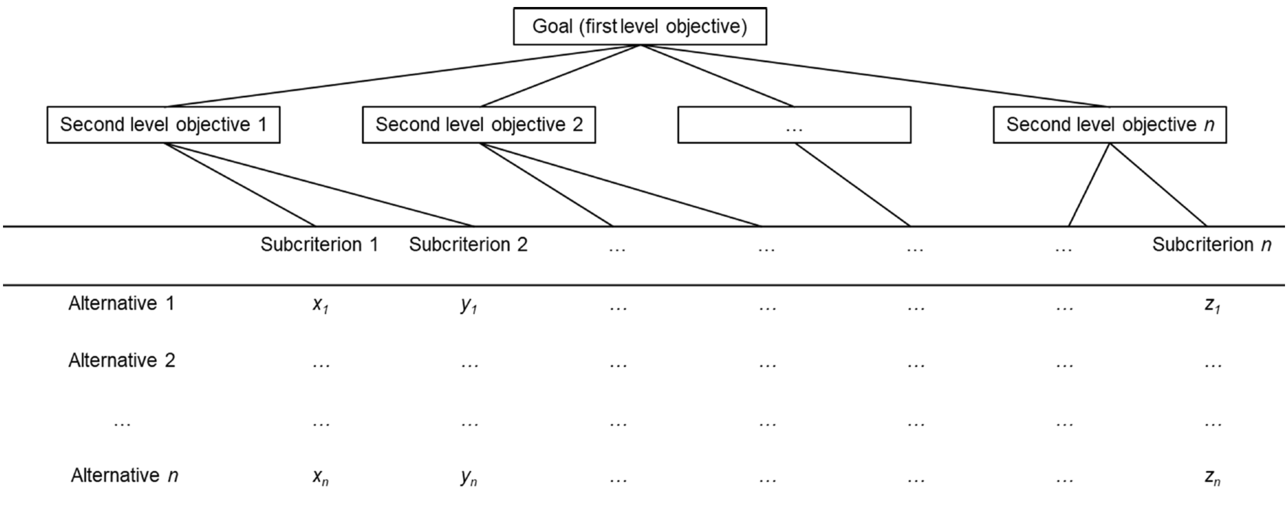


**FIGURE 2:** A generic example of a structured decision problem with alternatives evaluated by performances in multiple criteria.

dom expert sampling. Experts from universities, recyclers, car manufacturers, environmental protection agencies and both automobile producer and importer agencies were contacted for participation in the ELV workshop. Experts for the PPW workshop were invited from public waste agencies, plastic recyclers and polymer producers, ministries and research institutes. Project partners, including experts who were involved in the bottleneck analysis, definition of the theoretical framework, criteria and indicators or data collection were not included in the expert panels. The workflow of the EPWs is presented in Figure 3.

Firstly, the value chain and the objective of the MCDM were presented to the expert panel members. Once a necessary understanding of the decision problem and the system boundaries was reached, the decision alternatives (bottlenecks) were presented to the panel members. This was followed by an interpretation of the decision criteria, where the criteria measures' definitions were revisited in order to direct the group of experts towards making comparable decisions. The material on the value chain, MCDM goal and criteria and the bottlenecks was also given to the invited expert panel members before the workshop. Making sure that all participants understood the applied criteria and parameters was necessary so that the model would really be in line with preferences of the decision-makers and that the calculations produced priorities that really represent their preferences (Kangas et al., 2001).

After ensuring all the expert panel members were sufficiently informed and sharing the same ground about the decision-making problem, the actual elicitation of preferences was begun on the second day of the workshops using the appropriate MCDM method, AHP (ELV value chain) or MAVT (plastic packaging value chain). The resulting criteria weights and performances of the alternatives were subsequently combined to determine the preference order of alternatives. The results were presented to the expert panel to consider whether their preferences indeed were aligned with the final criteria weights and the priority order of the alternatives. An option to make correcting value statements was given to the individual panel members to ensure an agreement on the final priority order of the alternatives.

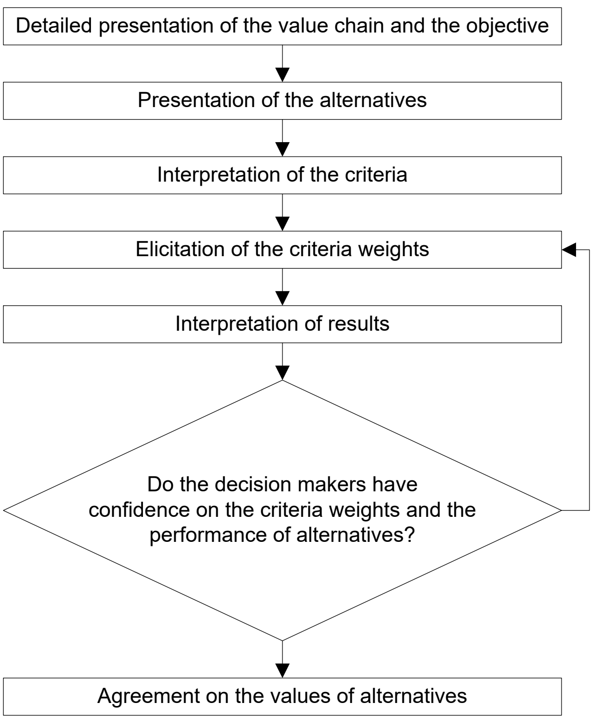


**FIGURE 3:** Workflow of bottleneck prioritization in expert panel workshops using Multiple Criteria Decision Making (MCDM).

The expert panel in the plastics value chain consisted of three DMs during the elicitation of the criteria weights. The affiliations of the experts were to a waste recycling company, a national environment institute and an EU-level trade association.

Eight experts participated to the elicitation of weights for the selected evaluation criteria in the ELV value chain. The participants included representatives of companies working in the ELV value chain, related interest groups and associations and researchers working within the field of ELV. Due to time limitations, the elicitation of the bottleneck performances in each criteria of the ELV value chain was conducted after the workshop via an electronic questionnaire sent to all workshop participants. Six of the experts completed the survey and thus participated in prioritizing the bottlenecks. The joint findings from the workshop and the questionnaire were sent to all respondents for information and comments.

### 3.3 The evaluation criteria for the bottlenecks

The goal of the MCDM was to establish the priority order of the bottlenecks to be removed based on their impact on the transition toward a circular economy. The interpretation of the goal and its division into suitable non-redundant criteria was done by the project group. The criteria defined during the planning stage before the value chain analysis covered the economic, environmental and material efficiency effects of the bottlenecks, as these were considered important lower level objectives.

During the identification of bottlenecks and later on in the discussions during the decision making exercise, vague definitions of circular economy and its relation to the zero waste concept were highlighted. Both concepts were discussed and considered to represent similar objectives. At the time of the workshop, the final version of the EU Action Plan for the Circular Economy was not yet available, and thus no official European definition was available. During the discussion with the experts, the move towards a zero waste society was seen as a more concrete definition and goal, and practical for the purposes of the MCDM. Participating experts agreed that a zero waste society aims at promoting the realization of the waste hierarchy (prevention, reuse, recycling, recovery and disposal) to minimize and reduce waste throughout the life cycle of an application. Overall, removing bottlenecks for recycling was considered important in any case, whether the overall goal would refer to a zero waste society or to a circular economy.

In order to carry out the MCDM, the effect of bottleneck removal on the entire value chain had to be assessed. This was found challenging, as information of the material flows within the recycling chains on the European level is inadequate, and total amounts of materials that end up outside the recycling chains are not known. Consequently, many assumptions had to be made and a lot of uncertainty was related to the data and the applied parameter values. However, coping with uncertainty is typical for the MCDM situations and has to be taken into account when conducting a sensitivity analysis and analyzing the results (Kangas et al., 2001).

Within the PPW chain, the effect of bottleneck removal was evaluated based on data available from the literature, and the information was shared in advance with experts who participated in the group decision-making.

The criteria included within the assessment of the PPW chain were:

- Losses of plastics in the recycling system;
- Total EU cost of the recycling scenario per ton recycled plastic packaging waste;
- Total EU revenues of the recycling scenario per ton recycled plastic packaging waste;
- Total of GHG emissions inside the EU due to the recycling scenario per ton waste generated;
- Employment (Jobs/ton recycled);
- Employment (Total jobs);
- Feasibility.

In the ELV chain, the evaluation of importance of the bottleneck removal was carried out by expert panel members based on their own expertise due to the lack of background data. Due to lack of data, also the number of criteria that could be included within the assessment of the ELV chain was reduced and only criteria referring to material savings potential and cost of recycling could be included. The expert panel members formulated the final definition of the criteria for ELV as part of the workshop.

The criteria included within the assessment of the ELV chain were:

- Reducing losses of plastics;
- Reducing losses of ferrous metals;
- Reducing losses of non-ferrous metals;
- Reducing cost of recycling.

## 4. RESULTS AND DISCUSSION

The results from the MCDM consist of criteria weightings and the final prioritization of the bottlenecks in the evaluated value chains. Chapter 4.1 presents the results for the plastics packaging chain and Chapter 4.2 the end-of-life vehicles chain. Experiences and learnings from the two cases are discussed in Chapter 4.3, as these are considered important for evaluating the applicability of the MCDM methods in this kind of problem solving. The final part of the chapter (Chapter 4.4) considers the limitations of the study and their impact on the results and their usability.

### 4.1 Priorization of bottlenecks within the plastic packaging value chain

The expert panel selected "Losses of plastics" in the PPW recycling system as the most important criterion (Figure 4). Regardless of the critique expressed on the spatial scope of the criteria, "GHG gas emissions within EU" was judged as the second most important criterion. The weights of these two criteria were clearly distinguished from the rest of the criteria. Even more clearly distinguished was the "Employment per recycled amount of plastic" (jobs/ton recycled) which was by far the least valued criterion.

The results show (see Figure 5) that solving the "Limited source separation" of plastic packaging waste would provide most value from among the bottlenecks. This is due to good performance in creating jobs, reducing GHG emissions within the EU, reducing costs per recycled plastic and most of all, reducing the losses of plastics in the recycling system. The value of prioritizing the solving of limited source separation was roughly three times the value of the next most valued alternative, "Improving the performance of separation/sorting technology". "Improving the performance of recycling technology" came third, followed by limiting the "Export of plastic packaging waste" and improving recyclability in the "Product design phase". Of all the alternatives, "Improving the performance of separation, sorting and recycling technologies" had the best feasibility values among the alternatives. In other words, they were judged as the alternatives being the most ready for implementation.

The expert panel, when presented with the criteria weights and total values of the alternatives, were quite content with how their views were translated to the results. The panel members asked to make only minor adjustments to their elicitation answers, and especially the effect of giving more or less value to the feasibility was tested. However, the priority order of the bottleneck remained.

### 4.2 Priorization of bottlenecks within the ELV value chain

In the ELV workshop, "Cost of recycling" gained highest weight whereas "Losses of Fe-metals" was ranked as the least important criterion (Figure 6). "Losses of non-Fe metals" was seen as the second most important criterion with a rather significant weight. The weight of "Losses of plastics" criterion was notably lower than for the first two, but was still considered more important than "Losses of Fe metals". In the discussion of the results between experts, the results did not come as a surprise and actually represented rather well the overall impression of the situation.

The outcome of the weighting was a result of the joint discussions during the workshop, and based on the interpretations on the available information. Thus, in practice, the answers given by the experts and consequently the results are dependent on the situation, and would vary in time.

The AHP highlighted (Figure 7) "Inadequate performance of vehicle dismantling and reuse application" as the most valued decision alternative of the five bottlenecks analyzed. However, "Inadequate performance of the separation, sorting and refining technology" as well as "Inadequate performance of ELV collection and monitoring" were valued nearly as strongly as the first one. The remaining two bottlenecks of qualitative nature gained clearly lower importance than the three technical ones. A possible reason for this is that the defining and quantifying of these two

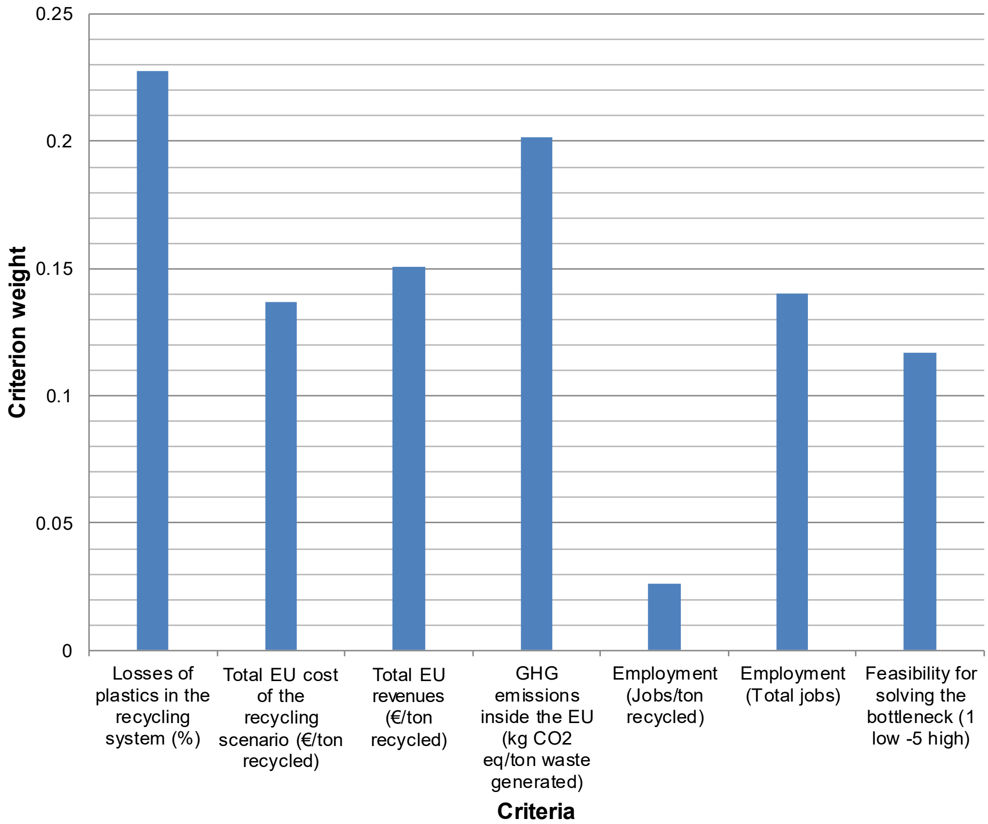


**FIGURE 4:** The criteria weights in the PPW chain. Adapted from (Bacher et al., 2016).

bottlenecks and their effects are more challenging which may have been reflected in the answers of the experts.

The answers received from the experts varied significantly, and this may at least partly depend of the background of the respondents. Commonly the background of an expert seemed to reflect the importance of the bottleneck. Therefore, it is important to have experts in the analysis from all stages of the value chain. In addition, a broader spectrum of the fields (academic, industry, etc.) of the experts would better take into account both the theoretical and practical point of views.

### 4.3 Discussion: Learnings from the case study

Despite the differences in the material compositions of the evaluated value chains, many similarities between the value chains and bottlenecks were identified. For example, among the important aspects were costs of recycling, unstable demand and low prices of the recovered materials (other than metals). Additionally, many of the bottlenecks related to the heterogenic composition of the input material that presents challenges for recovery and refining of the materials. Although recyclers have limited ability to affect product design, creating guidelines for product manufacturers and designers for taking into account the demands of material recycling could be an effective means in moving towards the more efficient and sustainable use of resources (Bacher et al., 2016). The importance of product design for improving recycling efficiencies underlines the need for a lifecycle approach that is central within the circular economy concept.

Due to the value chain focus and interlinked nature of the bottlenecks, evaluating their impacts within the value chains was rather difficult for the experts who participated in the workshops. Other challenges related to unclear definitions. Recent studies point out that the circular economy is a rapidly evolving concept and common agreements on necessary guiding principles for action are still missing (Merli et al., 2018). This highlights the importance of the goal definition for the MCDM, as it has to be clear for all participating experts. In the case of a blurred concept like the circular economy, a more case-specific definition of the circular economy should be created, including potential environmental, economic and social impacts.

Further challenges in applying the MCDM method in practice included lack of data related to the value chains, definition of comprehensive criteria that would be able to cover relevant economic, environmental and social aspects in quantitative terms, and engaging a balanced group of stakeholder representatives in the MCDM exercises. Since the bottlenecks were located along the value chain, there was a need for detailed economic, environmental and social data from each part of the lifecycle. In many cases this data was lacking, event hough great effort was made to collect it. Another important aspect is the balance between different aspects of sustainability within decision-making. Social implications are currently insufficiently presented in circular economy related literature, which is mostly dominated by environmental aspects (Merli et al., 2018).

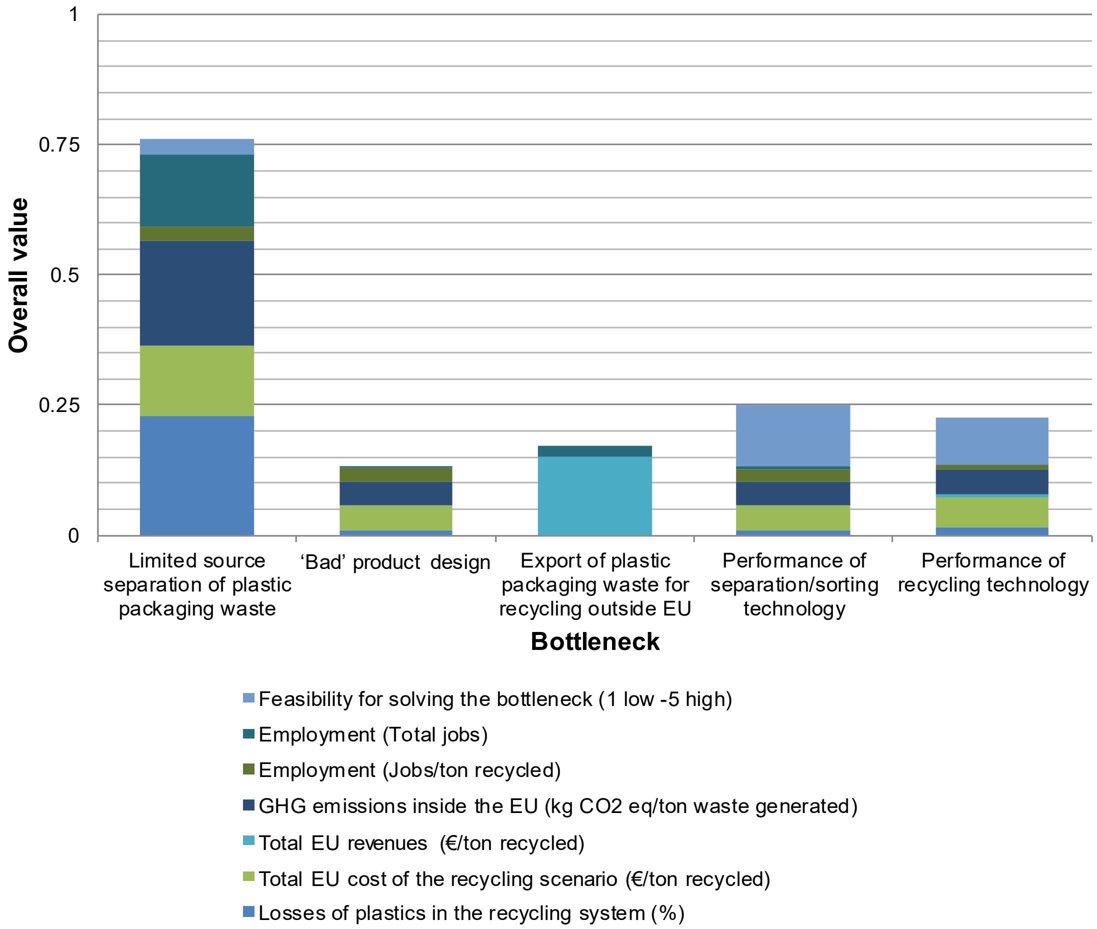


**FIGURE 5:** The results from MCDM in the PPW chain. Adapted from (Bacher et al., 2016).

Group MCDM methods may be helpful in addressing this shortage, since they in principle allow inclusion of economic, social and environmental criteria within the same decision-making context, although this might be challenging to apply in practice.

The findings from the case study highlight that transparent statistics and better information on the material flows within the value chains would be essential for successful use of the MCDM methods, and for advancing circular economy and related research. This would require more efficient monitoring activities and development of uniform waste statistics across the EU, together with new technological solutions that would allow tracking certain materials along their lifecycle. Ideally, addressing decision-making situations typical for the circular economy (covering the product lifecycle and including environmental, economic and social criteria), would require flexible methods capable of addressing both quantitative and qualitative data, and data of different quality. Similar findings have been obtained earlier by Kangas et al., (2001) when evaluating the potential of alternative MCDM methods for increasing consensus building among participants (in the context of forest management). Methods that allow using both low and high quality information in the same problem enable better use of all information that is available and enable more thorough decision support (Kangas et al., 2001).

Despite the challenging premises for decision-making faced at the EPWs, the participating experts considered the exercise and related discussions useful and interesting. Having participants with different backgrounds enabled addressing differing viewpoints while still trying to reach consensus. For example, while trade-off comparisons of criteria performances were perceived very difficult, the DMs became more comfortable after the first individual preference elicitations. The discussions within the group continued through the elicitation process. Once all preferences were elicited, the decision-makers felt satisfied with the results. This might have been assisted by two factors: the decision-makers were reassured by the discussions on their preferences, and the views on the meaningfulness of the criteria (based on data availability, limited scope etc.) were also shared by the participants.

### 4.4 Limitations of the study

The observations are based on a small sample of two empirical group MCDM exercises. There were differences in the composition of the expert panels but since the aim of this study was not to compare the results from the two workshops, this was not considered problematic. Furthermore, the participants in both expert panels represented a heterogenous group of stakeholders and thus differing points of view could be integrated in the process. Limita-

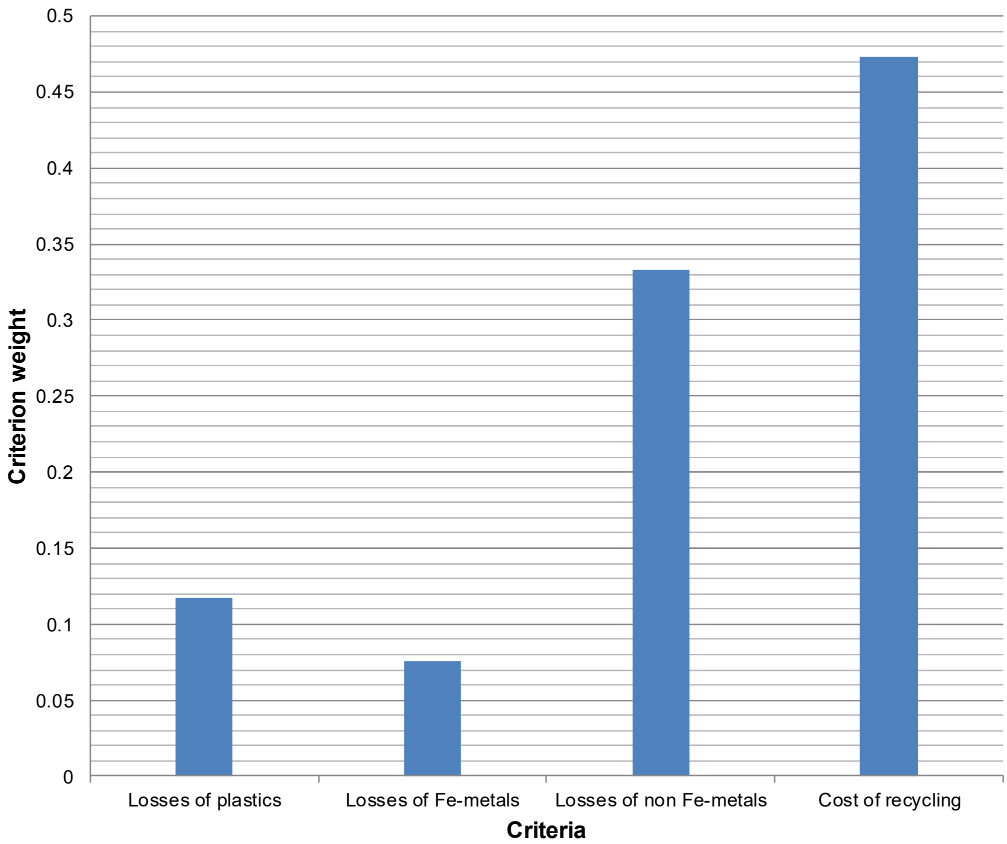


**FIGURE 6:** The criteria weights in the ELV chain. Adapted from (Bacher et al., 2016)

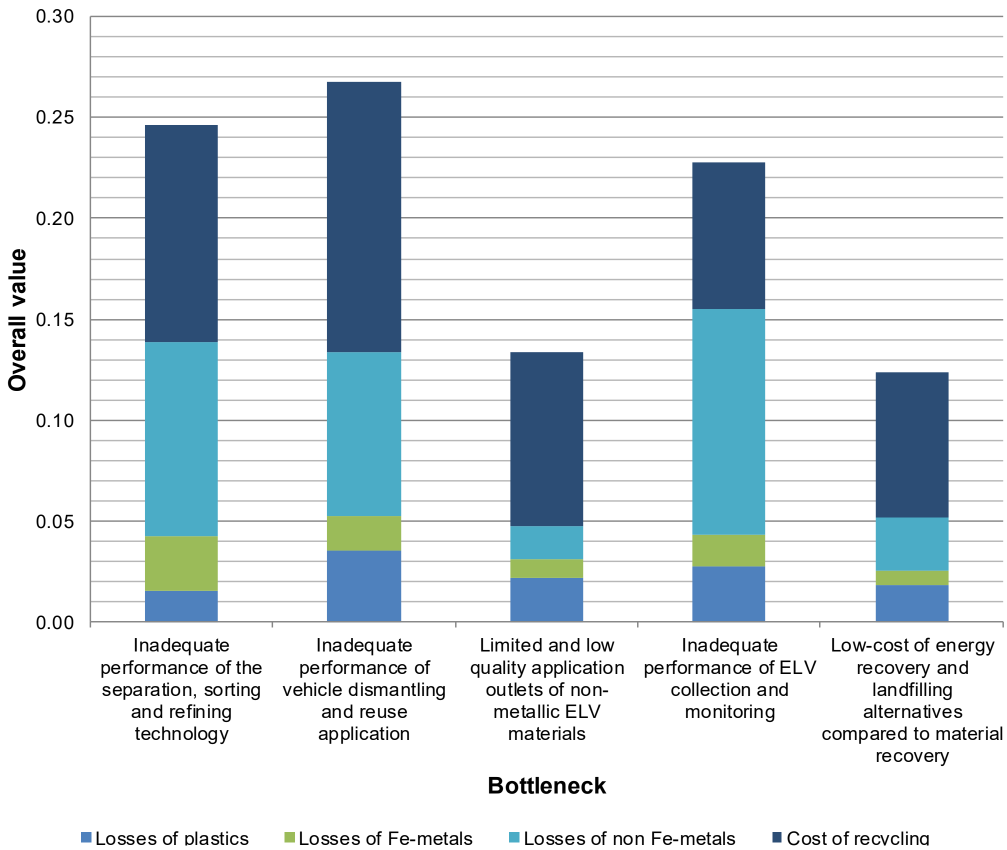


**FIGURE 7:** The results from the MCDM in the ELV chain. Adapted from (Bacher et al., 2016).

tions in availability of background data and ability to measure the impacts of some of the bottlenecks in quantitative terms resulted in reducing the number of bottlenecks and criteria that could be applied in the case study. While this can be considered as a limitation of the process, it simultaneously allowed analys of the challenges that can be faced in such a case study. Even with a reduced number of criteria and bottlenecks, the applicability of the methods could be tested. Most importantly, the participating stakeholders considered the process interesting and useful.

Using AHP allowed capitalizing on expert judgement in substituting for the lack of performance data of the alternatives in the selected criteria. In addition to evident issues on confidence in the results, such a decision-making process requires a considerable amount of time and is highly dependent on the selected expert group. Furthermore, the burdensome task in the empirical MCDM required leaving out the social and environmental criteria, other than material losses, which are all central to advancing the circular economy. As each alternative had to be contested against each other in fulfilling each criterion, also the number of alternatives were needed to be kept to a minimum in the empirical setting, thus affecting the results of the bottleneck prioritization.

Similarly, as comparable data is needed for all alternatives for each criterion using MAVT, the unavoidable shortcomings in data availability result in the rejection of criteria and alternatives prior to decision-making. This can lead to reduced transparency in the problem setting and the results. As a consequence, if the methods were applied in a real-life context, e.g. for prioritizing policy actions or implementation alternatives, considerable efforts would need to be dedicated to data collection and impact evaluation prior to the actual decision-making situation.

A methodological limitation to both MCDM cases was the use of linear value functions due to practical limitations. In other words, the non-linearities in the decision-makers' preferences over changes in the criteria performances of alternatives were not elicited. Regarding alternative approaches to AHP or MAVT, popular outranking methods, such as PROMETHEE (Brans et al., 1986) were not assessed. Application of these MCDM methods to group decision-making within CE remains an interesting subject for more research. Consideration of the generally known risks for shortcomings of the MCDM methodologies such as rank-reversals on the addition or removal of an alternative in AHP (Wang and Elhag, 2006) were left outside the scope of this study.

## 5. CONCLUSIONS AND RECOMMENDATIONS

The aim of the paper was to evaluate the potential of structured group decision-making methods to support co-operation between stakeholders and create learning that would be necessary in order to tackle the bottlenecks in the recycling value chains. The applicability of two different MCDM methods, AHP and MAVT, was tested for prioritizing bottlenecks in the PPW and ELV value chains. The empricial case study was limited in scope but learnings from the case study provide findings that could be of interest to the circular economy research community, and also for future policy development. While both MCDM methods and the circular economy have been extensively studied before, our understanding is that group decision-making methods have not yet been used in the context of circular economy studies. Thus the main contribution of this paper lies in combining these two approaches.

The experiences gained during the case study, together with the findings from the MCDM literature (see e.g. Kiker et al., 2005; Van den Hove 2006; Soltani et al., 2015) point out that structured group decision-making methods could be an effective means to increase co-operation and integrate views of different actors. In addition, they could promote openness of information and trust between actors in the value chain.

In the future, group decision-making methods could be applied as participatory methods to enable the collection and integration of stakeholder views within circular economy policy development and implementation activities. However, efficient use of the MCDM methods requires tackling some of the method-related challenges that include, for example, demands related to background data and criteria definition. Additionally, inclusion of qualitative data and criteria should be possible, in order to address different elements of the circular economy (environment, economy and society). This will also require that enough resources (time and money) are available to collect necessary background information and conduct the evaluations.

In order to support consensus-building, then applied methods should be capable of integrating the views of a large group of stakeholders, possibly having very different perspectives and backgrounds for evaluating the decision-making problem under study. Since the circular economy covers full product lifecycles, participating experts should represent all parts of the lifecycle, as well as different businesses, authorities, end-users and academia. This might be difficult to handle in practice, and will require commitment from the participating stakeholders. However, bringing together several actors from a supply chain (creating a life cycle view) is an essential element of the circular economy. The number of studies addressing whole supply chains in the circular economy is still limited (see Merli et al., 2018) and thus more research is needed on this topic in the future.

An important contribution from the use of group decision methods should be knowledge exchange and discussion among the participants. This is important since information exchange has been recognized as one of the major constraints to the success or circular economy initiatives (Winans et al., 2017). Structured group decision-making methods provide occasions for people with various backgrounds to exchange their views and learn how other persons think and what they consider important. Thus, in addition to creating a priority order for decision attributes and available alternatives, understanding the views of others may help us in understanding the impacts of our own actions within the value chain.

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# REUSE IN PRACTICE: THE UK'S CAR AND CLOTHING SECTORS

Peter Shaw [1,*] and Ian Williams [2]

[1] Centre for Environmental Science, University of Southampton, United Kingdom
[2] Faculty of Engineering and Physical Science, University of Southampton, United Kingdom

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**ABSTRACT**

Ongoing efforts to seek better resource efficiency have highlighted the role of reuse as a contributor to achieving circular economy objectives. In order to improve resource efficiency, the motives, means and opportunities for reuse need to be understood such that best practice can be identified and measures implemented to foster more effective and more extensive reuse. This study compares and contrasts reuse in the car and car components sector with the clothing sector as a means to identify commonalities and differences, and seek facets of effective practice. The car sector is found to align more with financial motives than the clothing sector, the latter providing more marked and apparent social benefits. Three key aspects appear common to both sectors. First, whole lifecycle – cradle to cradle – approaches to enhancing reuse are emerging and have considerable merit from a circular economy perspective. Secondly, the internet has become a key tool for the facilitation of reuse and is likely to grow further in this regard. Thirdly, decisions regarding the end-of-use of consumer products are critical and need to be better understood. Fourthly, for any reuse initiative to deliver positive outcomes, consumers must be fully engaged. We conclude that whilst some sector-specific adjustments may have to be implemented in future initiatives to promote and enhance reuse activities, the overarching principles and optimum methods of reuse facilitation may well be common for contrasting sectors.

## 1. INTRODUCTION

### 1.1 Background and context

It has been widely recognized that an urgent need exists for humankind to identify and apply means to achieve more effective, efficient and sustainable use of global resources. Approaches to resource use that recognize and respond to the call for due consideration of the environmental, social and economic facets of the production, provision and consumption of goods and services are widely supported. Indeed, the United Nations has called for "responsible production and consumption" at a global scale as their 12th Sustainable Development Goal (UN, 2015). The more efficient utilization of resources will likely be guided by key concepts, notably the waste hierarchy (EC, 2008; Williams, 2015) and the circular economy (Ellen MacArthur Foundation, 2017; EC, 2018), both of which highlight the role of reuse as a contributor to greater sustainability and advocate enhancement of reuse activities. In the case of the waste hierarchy, reuse is generally considered of lower preference than to "reduce" waste, but preferred to "recycle", "recover" or "dispose" (EC, 2008; Williams, 2015). In terms of circular economy concepts (Ellen MacArthur Foundation, 2017; EC, 2018; WRAP, 2018a), reuse offers a means to exploit more fully the utility of a product by extending its use beyond the point at which its owner considers it to have ceased providing them with the desired or needed function.

The aims and principles of reuse are well established (Williams and Shaw, 2017 and 2018). However, the practices of reuse in terms of what takes place, how it is achieved and what it achieves have been the subject of relatively little research (Cooper and Gutowski, 2017). In contrast, there has been considerable focus on recycling within the waste and resource management sector, allied with and driven by targets for recycling that are enshrined in policy and legislative frameworks on an extensive basis (e.g. EC, 2008). Likewise, targets for reduction of waste disposal to landfill are also enshrined in policy and law (e.g. EC, 1999 and 2008) and recycling is broadly recognized as a key factor in achieving reductions the quantities of waste that are landfilled (e.g. Farmer et al., 2015; UK Government, 2003).

It can be argued that ambitions to increase recycling and reduce disposal to landfill share a common feature. Provided that suitable infrastructure and recording methods are in place, quantities of recycled and landfilled materials are readily measurable. We suggest that, in addition to the overarching ambitions of targets for recycling enhance-

* Corresponding author:
Peter Shaw
email: ps@soton.ac.uk


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ment and landfill reduction, the opportunity for measuring quantities of recycled and landfilled materials renders such targets attractive for reasons of practicality. The preponderance of targets and initiatives orientated towards recycling enhancement (e.g. EC, 2008; UK Government, 2003) and landfill reduction (e.g. EC, 1999) are driven, of course, by a rational and well-founded ambition to achieve environmentally-desirable outcomes; the facility or opportunity to measure performance in relation to the specified targets is, arguably, a further motivation for their adoption. Reuse, in contrast, appears less readily measurable and is, we contend, consequently a less common feature of waste management or resource efficiency targets. The existence of extensive and common informal networks (Williams and Shaw, 2017 and 2018), for example, offers a wide range and high number of opportunities for reuse that are not routinely accompanied by formal or extensive recording mechanisms.

### 1.2 Aims of the study

Notwithstanding the inherent challenges in setting meaningful targets for reuse and measuring performance in this regard, it remains that reuse comprises an integral and critical contributor to actions orientated towards resource efficiency and associated benefits. In order for reuse to occur, however, there are key prerequisites. First, there must be a desire on the part of individuals to participate in reuse, either as the current owner or the future owner of an item. Unless this desire exists for both current and future owners, reuse will likely not take place. Secondly, there must be means by which reuse might occur, i.e. a mechanism or facility that enables what is no longer needed or wanted by one party to be made known and available to another, and with a method for payment, if needed, to be made. In the light of these prerequisites, this study seeks to explore, through exemplification, current practice in reuse with specific focus on:

- Motives: drivers and benefits of reuse;
- Means: structures and methods to facilitate reuse.

On the basis of this review, an appraisal will be made of opportunities to enhance reuse, and recommendations made for future research.

## 2. METHODS

### 2.1 Key definitions and terms

On a formal basis, reuse is considered to include both products and components thereof, and also operations or processes that might be conducted before reuse takes place (Table 1). Although this illustrative example is specific to the European Community (EC, 2008), inclusion of these terms highlights important aspects of reuse in that items that are viable for reuse are restricted neither to whole items and products, nor to items or products that are reusable in the condition or state at which the present owner decides that they are no longer needed or wanted. These definitions (Table 1) highlight a key difference between reuse and preparing for reuse. It is inferred (Table 1) that reuse applies to items that have not become waste as such, whilst preparing for reuse relates to items that have indeed become waste prior to preparing for reuse.

### 2.2 Case studies

In view of the stated realm of reuse (Table 1) and the aims of the study (Section 1.2), a review of studies and information in the public domain was undertaken, with the intention of exploring and exemplifying current reuse practice. Selection of the sectors on which the study would focus was made with reference to four requirements. First, there should be potential for reuse of goods purchased when no longer needed or wanted. Secondly, a sector was required that was notable in terms of consumer spending activity; consumer spending on such goods should comprise a substantial part of general household spending. Thirdly, we sought to elucidate how sectors differ in terms of reuse as a function of the frequency of opportunities for reuse and the potential economic value associated with reuse. This aspect was a particular focus for the study as the drivers and benefits of reuse could be expected to differ according to the value of reused items, and with more costly and more durable items generally being retained for longer by the purchaser. Fourthly, items reused within the sectors selected should offer opportunities for reuse and preparing for reuse (Table 1). UK householders' spending revealed that many key purposes of spending were for goods and services that did not offer opportunities for reuse (Figure 1), comprising, for example, items intended for immediate consumption and access to services. Of the four spending purposes identified as having potential for the goods purchased to be reused, transport (including purchase of cars; Figure 1) is clearly a major area of household expenditure. Car purchases may be considered high cost and low turnover as a generalisation, and with opportunities for both reuse and preparing for reuse (Table 1). Of the other areas of household expenditure with clear potential for recycling, purchase of clothing is characterised, in general, with relatively modest costs and high turnover, and with potential for both reuse and preparing for reuse (Paras et al., 2018; Table 1). Although other areas of

**TABLE 1:** Definitions of key terms appertaining to waste and reuse, as set out in the EC Waste Framework Directive 2008/98/EC (EC, 2008).

| Term | Definition |
|---|---|
| Waste | "…any substance or object which the holder discards or intends or is required to discard". |
| Reuse | "…any operation by which products or components that are not waste are used again for the same purpose for which they were conceived." |
| Preparing for reuse | "…checking, cleaning or repairing recovery operations, by which products or components of products that have become waste are prepared so that they can be re-used without any other pre-processing." |

household expenditure also include reusable items (e.g. "durable" households goods such as cookers, washing machines; ONS, 2018), the cost and turnover of such items are, in general, higher and less rapid than items of clothing.

A comparison of the car and clothing sectors was made, these two contrasting sectors meeting requirements as stated. The UK serves as an illustrative exemplar in this instance due to the availability of data for both sectors, and the opportunity for reuse of clothing through charity shops (Diop and Shaw, 2018; Osterley and Williams, 2018). The reach of some businesses brings some international facets to this review by virtue of cross-border initiatives and activities. The intention in this instance is to compare and contrast current practice rather than to present a comprehensive review or to scrutinize and report upon discrete reuse initiatives or case studies (e.g. Beasley and Georgeson, 2016; WRAP, 2011), providing a means to illustrate the extant motives and means for reuse, and thereby explore opportunities for enhancement of reuse in broader terms (§1.2). The sectors selected for focus are outlined in Table 2. The two sectors selected are both major contributors to consumer-orientated economic activity in the UK (Figure 1; Table 2), and both are substantial in terms of the volume of sales and consumption of primary resources. There is a contrast in that clothing consumption is characterised by high volume sales of relatively inexpensive items, whilst the car industry is characterised by quantitatively lower sales volumes but contributes substantially in economic terms due to the much higher purchase costs of individual units. In addition, these two sectors contrast in terms of the opportunities for reuse of components and preparing whole products or components for reuse (Table 1).

## 3. RESULTS

### 3.1 The automotive sector: cars and components

#### *3.1.1 Overview*

Global production of cars (Table 2) was in the region of 74.5 million in 2017 (Statistica, 2018a) and contributes at major scale to global economic activity. In general, the financial cost of cars is sufficiently high that owners (private individuals, organizations or traders) seek financial gain by selling or trading vehicles when they are no longer needed or wanted. In the case of traders, there is a clear profit motive to hand. For others, the money generated via reuse (i.e. sale or trade-in) is commonly less than was initially paid for the vehicle and presents an opportunity to recoup a part of the initial cost rather than to make a profit. There is a well-established market in "used" cars, driven by demand from, for example, private buyers who cannot afford or do not wish to pay for a new vehicle, and trade buyers who seek to profit through buying and selling used vehicles. For 2014, it has been estimated that sales of used (reused) cars in the UK amounted to a market value of £4.3 billion (Statistica, 2018a).

The importance and influence of legislation and associated requirements with respect to vehicle reuse and recycling have been highlighted (e.g. Bellemann and Khare, 1999; Wahab et al., 2008; Go et al., 2008). Motivated by the potential gains in energy and material productivity through increased reuse and recycling (McKenna et al., 2013), statutory requirements pose challenges to the automotive sector; statutory demands and the demands of customers both have to be met (McKenna et al., 2013). Resistance to reuse of components in new vehicles has been observed,

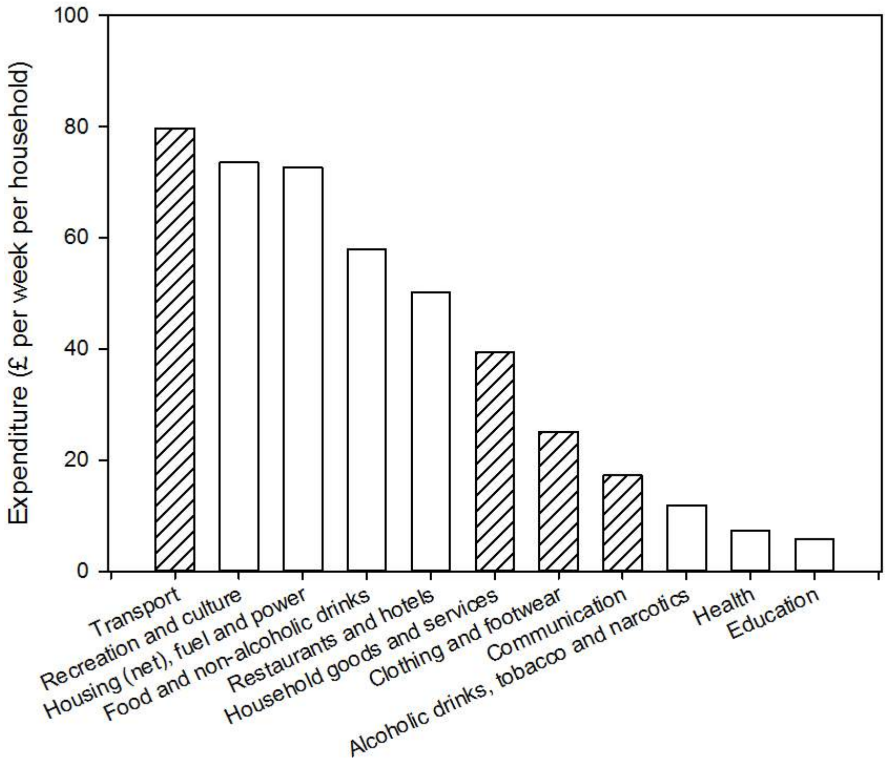


**FIGURE 1:** UK household expenditure in 2017. Data from ONS (2018). Shaded bars indicate sectors with opportunities for reuse.

**TABLE 2:** Profiles of the automotive and clothing industries in the UK.

| Sector | Notes |
|---|---|
| Cars and car components | In 2017 an estimated 74.5 million cars were produced globally (Statistica, 2018a). Reuse of automobiles (second-hand/used) and components ("spares") is common and extensive. In 2016-2017, 79% of UK households owned a car or van (ONS, 2018). In 2017, 2,540,617 new vehicles were registered (SMMT, 2018c) and 8,113,020 (re)used cars changed hands (SMMT, 2018d ) . In 2015, the total weight of end-of-life vehicles in the UK was an estimated 966,657 tonnes (Europa, 2018). |
| Clothing | The global clothing industry accounted for an estimated \$2.4 trillion in 2016 (MacKinsey and Co., 2017). Items of clothing may be deemed "end-of-use" for reasons other than loss of utility; options for reuse are numerous and offer opportunities for altruism (Diop and Shaw, 2018) and financial gain (Morley et al., 2009). Expenditure on clothing and shoes in the UK was estimated at £57.8 billion in 2017 (Statistica, 2018b); every year in the UK clothing worth an estimated an £140 million worth (ca. 350,000 tonnes) of used clothing is destined for landfill (WRAP, 2011). |

although opportunities for after-market reuse of components may arise (Amelia et al., 2009). Specific challenges for component reuse have been identified for tyres (Lebreton and Tuma, 2006) and plastics (Bellemann and Khare, 1999), for example. With regard to the necessary processes and facilities for direct secondary reuse of vehicle components (McKenna et al., 2013), the importance of approach(es) to disassembly appears critical. Identifying the optimal stage for disassembly is clearly important if the components recovered are to be of suitable economic value (Go et al., 2011), whilst designing components from a whole life-cycle perspective – including disassembly and reuse – is clearly a desirable step towards greater environmental sustainability in automotive manufacture (Wahab et al., 2008; Go et al., 2012;). With regard to disassembly, the question of "who does what?" is a further consideration and complication. As noted by Matsumoto (2009), options for reuse and remanufacture include both original motor manufacturers and independent reuse business companies.

*3.1.2 Reuse in practice: cars*

Selling and trading cars that have been previously owned by another party constitutes reuse (Table 1) and the means available to do so are numerous. Opportunities exist for exchanges between private individuals, between traders, and between private individuals and traders. Reused cars for sale may be advertised through online social media, web-based trading facilities, printed copy in local newspapers and magazines (specialist or general), whilst auctions, dealerships and associated marketing also offer information. Just over half of used car sales in the UK in 2014 were accounted for by sales from dealers (Statistica, 2018a).

Reuse of cars is clearly widespread and commonplace, if not routinely or widely considered as constituting "reuse" per se. In 2017, just over 2.5 million new cars were registered (SMMT, 2018a); over this same year, 8.1 million used (i.e. reused) cars changed ownership (SMMT, 2018b). Purchases of reused cars thus outnumber new car purchases at a ratio of ca. 3.25:1. When considered over a longer period, available data again illustrate the contribution of reused cars to the whole UK-wide stock of registered cars. By the end of 2016, some 7.8 million cars registered in the UK (24.5% of the total) were recorded as having had a single "keeper" (Figure 2). The remaining 75.5% of all registered cars had been previously registered to another keeper, implying that ¾ of UK-registered cars at the end of 2016 were reused vehicles (Figure 2). Around 3 out of 10 UK-registered cars at this time had had one former keeper (reused once since first purchase) and 2 out of 10 had had two former keepers (reused twice since first purchase). In extreme cases, usually for cars first registered before 1979, records show that some cars have been registered by 20 or more keepers, inferring that, albeit rarely, some cars had

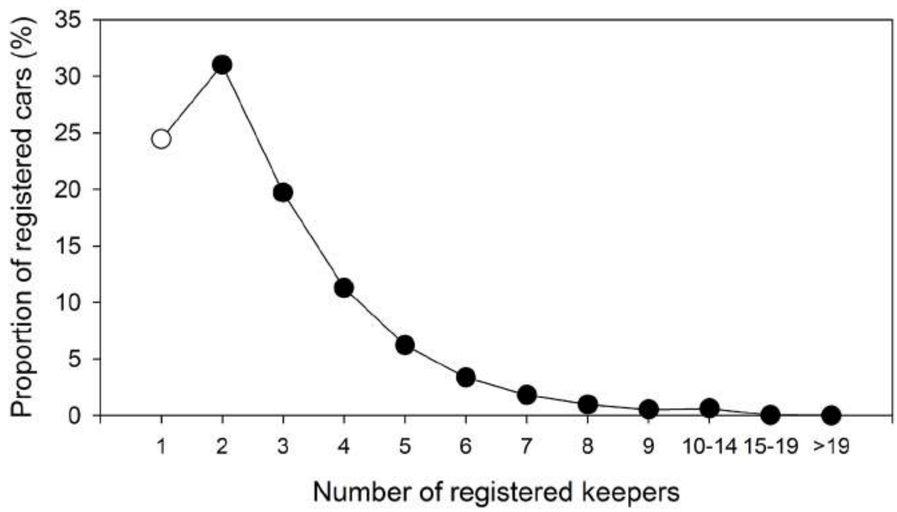


**FIGURE 2:** The proportions of UK-registered cars and corresponding number of keepers as recorded for 2016. The registered keeper is not necessarily the owner of the car; change of keeper usually signifies change of ownership. Open symbol represents new cars, closed symbols represent reused. Data from DfT (2017).

changed hands 19 times or more.

We note that measures are in place in the UK (and elsewhere) regarding the fitness-for-purpose of reused cars in terms of tests for their roadworthiness. UK MOT (Ministry of Transport) certification is provided for vehicles that meet the minimum standards as set (UK Government, 2018) which broadly concern (1) legalities regarding identification and registration of vehicles, and (2) checks and inspections to ensure that the vehicle is of a roadworthy condition. The MOT system also makes available some facets of a vehicle's history (e.g. MOT history, recorded mileage) to a potential purchaser but does not cover some major mechanical systems: engine, clutch and gearbox are excluded. Further assurance that a reused car is sound with respect to these mechanical systems may be gleaned by independent inspection, but usually at additional cost for the services of automotive industry organizations or other industry professionals. MOT certification does not provide any guarantee to a prospective purchaser or a valuation of the vehicle. Warranties are commonly provided by registered traders, purchases from whom are covered by consumer law (UK Government, 2015). Private sales of vehicles are not regulated under the same framework; it is common practice that such sales are made "without warranty or guarantee", the onus being on the purchaser to ascertain the status of the vehicle over and above the MOT certification. It should also be noted that the preponderance of reused cars (Figure 2) generates a need for replacement components and parts. Failure, breakage or damage inevitably become more likely as the age and use of a vehicle increase, leading to a demand from owners which may be met in part by reused components and parts.

There are multiple means by which used cars are exchanged. Historically, vehicles no longer wanted or needed by the present owner have been advertised to prospective buyers via hard copy publications. Well known publications in the UK are "Exchange and Mart" and "Autotrader" both of which have adopted and adapted to internet-based formats in the light of widespread access to and use of web-based technologies and devices. From a potential buyer's perspective, there exist similarities in hard copy and digital formats: details of the vehicles for sale are provided in concise form, including make, model, age, colour, recorded mileage, presence or absence of MOT certificate, and usually a seller-provided description of the vehicle's condition. Public access to digital databases, however, offers great choice to the potential buyer and access to highly specific search tools. In addition to the details provided in hard copy format, it is usual to provide several photographic images of the vehicle to assist the potential buyer. With regard to the search for a vehicle, potential buyers may readily filter their search according to their preferences and desires. In addition to make, model, age, colour, recorded mileage, presence or absence of MOT certificate, on-line search engines (e.g. www.autotrader.co.uk, www.ebay.com/motors/carsandtrucks; www.gumtree.com/cars) can include numerous additional filters. Distance from the searcher's address, minimum and maximum price, body type, engine size, $CO_2$ emissions, number of seats, gearbox and fuel consumption, plus other variables, can be specified. Access to such tools is clearly advantageous in terms of vehicle reuse, in making more streamlined and controllable the ways in which sellers and buyers may be linked. We note that in terms of cost, the wide range and number of vehicles that can be viewed has impacts on both sellers and potential buyers. The price of a vehicle as advertised is readily comparable with prices of similar vehicles; the quasi "market value" of a vehicle as observed will thus likely influence the expected price from both sellers' and purchasers' perspectives.

#### *3.1.3 Reuse in practice: car components and parts*

Whilst formal targets for the reuse of whole vehicles appear absent in the UK, there are policies intended to ensure that reuse is embedded in the fate of end-of-life vehicles (ELV). The European Community ELV Directive (EC, 2000) for example, challenged manufacturers "to factor in the dismantling, reuse and recovery of the vehicles when designing and producing their products." Two targets were set within the ELV Directive (EC, 2000) for new vehicles in terms of (1) the percentage by weight per vehicle that should be reusable and/or recyclable (85%), and (2) the percentage by weight per vehicle that should be reusable and/or recoverable (95%).

Innovations within the car manufacturing industry have included "easy to dismantle" vehicles, through which manufacturers claim to have made less onerous the process of dismantling a vehicle such that most of the value of components and materials is not lost. The Toyota Motor Manufacturing Company, for example, promote "the four Rs" (Reduce, Reuse, Recycle, Recover) and have implemented a range of associated initiatives. With specific regard to reuse of vehicle parts and materials, key areas of activity are highlighted (Table 3). In this instance, remanufacturing of parts constitutes "preparing for reuse" according to EC definitions (Table 1). The stated reuse of hybrid and traction batteries (Table 3), however, does not lead to reuse in the same way as the components were used in the vehicle as initially produced. In this instance, "reuse" does not therefore formally accord with the definition as stated in the EC Waste Framework Directive (Table 1; EC, 2008).

In practice, the requirements of the EU ELV Directive (EC, 2000) and industry reuse innovations (e.g. Table 3) represent a re-casting of well-established and long-practiced activities. In particular, the reuse of vehicle parts has long existed in the form of "scrap yards", "breakers' yards" or "auto dismantlers", i.e. businesses providing a source of replacement car parts from ELVs. Such businesses commonly operate in differing modes. A "self-service" approach operates by customers being able to access ELVs stored in open-air yards and remove desired parts from vehicles themselves. Other businesses dismantle vehicles and salvage reusable and saleable parts, creating a stock of off-the-shelf parts and components; some will remove parts from ELVs to order. Inspection and reconditioning of parts (cf. Table 3) may be necessary for some components obtained from dismantlers or scrap yards. Components of a cosmetic nature (e.g. decorative body trim) may require only visual inspection as a means to ascertain their condition, whilst the condition of mechanical or electrical parts

**TABLE 3:** Reuse initiatives promoted by the Toyota Motor Manufacturing Company (Toyota, 2018).

| Initiative | Notes |
|---|---|
| Remanufacturing parts | Remanufacturing programme for the European retailer network. Returned parts are inspected and reconditioned; worn elements are replaced, and the part reassembled and packaged for sale to customers. Major items remanufactured include air conditioning compressors, automatic transmissions, power steering racks, cylinder heads, engines and clutch kits. |
| Reusing hybrid batteries | Nickel and lithium-based batteries used in hybrids may outlast the vehicles they power. Ongoing investigation into how these might be remanufactured for repeat use in vehicles and/or be reused for other purposes; use for stationary or emergency power storage units is being considered. |
| Stationary storage; traction batteries | Traction batteries for end-of-life hybrid vehicles, if in good working order, can be removed and used as a stand-alone, stationary power storage unit. Applications may include co-ordinated energy-saving systems, or emergency back-up supply. |

may not be so readily assessed.

Issues of supply and demand present challenges in, and possible barriers to, reuse of ELV parts and components. In principle, a car owner seeking a specific replacement part for their own vehicle needs to locate a like-for-like part from an ELV "donor" vehicle that is suitably compatible (e.g. with respect to model, variant, age and potentially colour). Matching of parts may be easier in some instances than others. Where engines are used commonly for several different models of car from a single manufacturer, for example, components will more likely be engine-specific than model-specific. In contrast, other parts (e.g. body trim, interiors, lighting and body panels) will necessitate location of a more precise match of donor ELV.

The task of locating suitable parts for specific models has been made easier by the advent of searchable online databases of available parts and vehicles. Access to digital databases via the internet offers widespread access to vehicle dismantlers across broad geographic areas; web-based search tools provide access to national networks of auto dismantlers. Common practice in the past in the UK has been to locate a suitable donor vehicle was, by custom and practice, achieved by telephone conversations with yard operators, a practice rather limited by proximity of an individual to dismantling yards as well as the ELVs held in stock. Locating specific used parts to replace damaged or non-functioning car components requires, of course, a specific match. To this end, internet-based searching permits the customer to specify the necessary details and thereby locate a suitable part. There appear to be two modes by which such searches can be made. First, some dismantlers provide searchable lists of vehicles currently held in stock; in some instances the vehicles are routinely dismantled and parts held in stock; in other cases the details of vehicles are held to enable "self-service" to parts by customers. Networks of dismantlers offer a perhaps more effective approach enabling access to vehicle and parts held in stock by a large number of dismantlers. Such networks operate on an "enquiry" basis whereby the customer specifies the part(s) required. The enquiry is then circulated to network members, who respond to the query when a suitable part is available. The emails received by the enquirer then provide details of choice, availability, and cost. When the facility is available, entering the registration number for a vehicle permits the make, model and year to be readily identified and partner/network organizations may then be invited to respond via email to specific enquiries. This facility enhances opportunities to locate suitable parts by opening up access to a high number of dismantlers distributed over a wider geographical (National) area and, at the same time, for purchasers to compare prices.

### 3.2 The clothing sector

*3.2.1 Overview*

The retail textile industry is a major contributor to economic activity (Table 2). Estimates of the economic scale of the global clothing market range from \$1.2 trillion in 2014 (Resta and Dotti, 2015) to \$2.4 trillion in 2016 (MacKinsey and Co., 2017). In the UK, consumers spend in the region of £53 billion per year on around 1.1 million tonnes of clothing, accounting for 5% of household expenditure (data for 2014: WRAP, 2016). Volumes of textiles destined for landfill at their end-of-use have in the past exceeded a million tonnes per year (WRAP, 2013). Consumer demand for clothing products (largely orientated to fashion) is increasing (Pookulangara and Shepard, 2013) and is likely associated with increases in textile waste arisings (Birtwistle and Moore, 2007).

Clear benefits of using recovered (recycled) materials rather than virgin resources have been demonstrated (e.g. Woolridge et al., 2006; Dhalbo et al., 2017; Esteve-Turillas et al., 2017); the benefits of clothing reuse have likewise been demonstrated and are likely to exceed the benefits of textile recycling (Sandin and Peters, 2018). At the same time, complexities in understanding the reuse of clothing have been identified. Consumers' behaviours are complex, involving, for example, compulsive purchases and value-orientated hoarding (Joung, 2013). Behaviours relating to clothing and its reuse may also be orientated to the consumers' level of materialism (Joung, 2013) and their status as "fashion consumers" or "non-fashion consumers" (Weber et al., 2017). Factors influencing reuse may also be many and varied, including environmental concerns, economic concerns, concerns for charity, subjective norms (Joung and Park-Poaps, 2013), convenience (Laitala, 2014); and a personal preference for antique items, clothing from a past era and/or a particular "look" (e.g. steampunk, gothic, rock'n' roll, military, anime, etc).

*3.2.2 Reuse in practice: clothing*

The fate of end-of-use clothing is clearly of concern in terms of environmental impacts and sustainability, and yet determination of consumers' post-purchase behaviour with regard to clothing has in the past received relatively little attention in the research literature (Joung, 2013). If items of clothing that retain utility are destined for landfill or incineration, for example, the value of the resources used in their production (e.g. raw materials and energy) and delivery to the point of sale (e.g. packaging and transport) will not be fully realised.

Whilst there are uncertainties regarding the destinations and ultimate fates of end-of-use clothing with regard to their reuse or recycling, prior research has given some indication of the relative quantities in this regard (Figure 3). Substantial quantities of clothing are evidently exported from the UK for reuse or recycling. In 2007, for example, 368,000 tonnes of clothing were exported from the UK (Morley et al., 2009). Quantities of textiles intended for reuse within the UK are considerably higher than for recycling or export (Figure 3). Indeed, the ratio of reuse to recycling on a weight basis (Figure 3) is 4.6:1 for the UK and 6.1:1 for exported textiles. Uncertainties regarding the fate of exported items have been previously reported (Morley et al. 2009); exports are deemed beyond the scope of this study.

There are economic benefits in ensuring that clothing items are reused and their premature disposal is avoided. Estimates suggest that over 350,000 tonnes of potentially reusable textiles are disposed of or recycled per year; reuse of textiles in England could save the tax payer £35 million per year via avoided disposal costs, and the resale value of reusable items could be over £140 million (LGA, 2014). Recycling and reuse of clothing have differing merits. Whilst avoidance of disposal has benefits in economic and environmental terms, the motivation for promotion and enhancement of reuse of clothing is additionally orientated to social benefits (LGA, 2014). Funds may be raised for charitable purposes through the sale of reused items (Osterley and Williams, 2018) and, at the same time, low-cost clothing is made available to those with limited income (Diop and Shaw, 2018) or who wish to purchase reused items as a lifestyle choice (Williams and Shaw, 2017).

Opportunities for reuse of clothing in the UK mainly occur in two modes: donations or sales. The means for reuse thus contrast, although some (e.g. internet-based means) permit access on a broad geographical scale. It is noted that in all cases the expense of purchasing new items and the need for production of new products are reduced or avoided (Table 4).

**Donation to a personal acquaintance**

Donation and exchange of clothing items between friends, neighbours, family and other acquaintances are, for obvious reasons, not formally recorded. Estimates have been made, however: together with items sold or exchanged on-line, direct reuse of clothing in this form may account for around 100,000 tonnes per year (WRAP, 2016). Such figures are encouraging, but there are no doubt more opportunities in this regard: reusable items may be placed by householders into kerbside-collected refuse bins rather than given to others living in close proximity or with whom householders are in regular contact. Local authorities have noted the placement of reusable items in kerbside-collected refuse bins, including clothing, and taken steps to encourage householders to reuse rather than dispose of suitable items (e.g. SCC, 2018). Anecdotally, donation and exchange of clothing items appear well-developed amongst families with young children. In this context, two observations may be made: (1) clothing items are often outgrown by children before they become excessively worn and their utility is diminished, and (2) social networks and contacts to facilitate reuse are readily available in the form of social media and informal encounters through school-related activities. We note that donations in this manner comprise "giving" of clothing items as opposed to "gifting" via donations to a charity or similar organization (Diop and Shaw, 2018).

**Donation through a reuse group or event**

As with direct donations to known individuals, donation

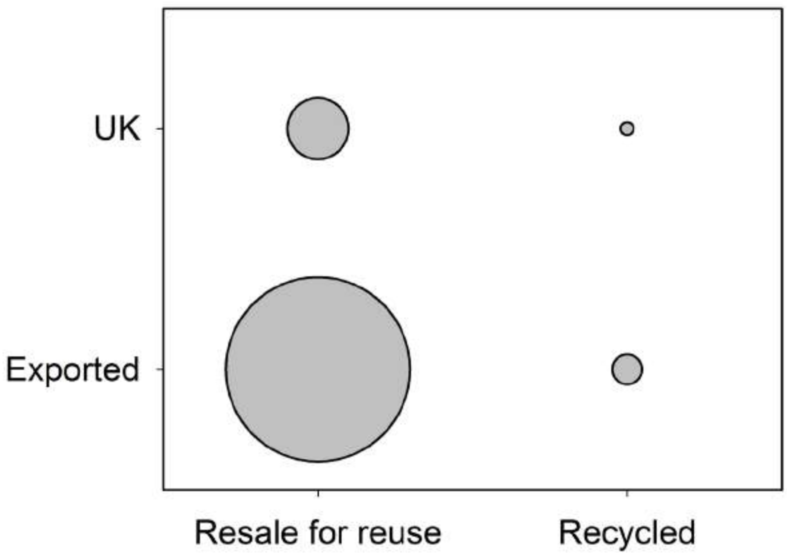


**FIGURE 3:** The relative quantities (by weight) of textiles sent for reuse or recycling in the UK in 2007. Data from Morley et al. (2009). Areas of symbols are proportional to total weight for 2007.

**TABLE 4:** Examples of main clothing reuse opportunities for consumers and their associated benefits.

| Initiative | Recipient | Benefit(s) – examples |
|---|---|---|
| Donate to a personal acquaintance. | Friend, neighbour, family, colleague etc. | Avoided expense of purchasing new items; avoided need for production of new items "feel-good" factor for donor; reinforcement of social relationships. |
| Donate to an unknown person through reuse group. | Reuse group member | Avoided expense of purchasing new items; avoided need for production of new items. |
| Donate to a charity or third sector organisation (TSO). | Charities with UK and/or international remit. | Avoided expense of purchasing new items; avoided need for production of new items "feel-good" factor for donor; provision of low cost items through charity shops/outlets; employment. |
| Direct resale via websites, local newspapers, car boot sales. | Purchaser | Avoided expense of purchasing new items; avoided need for production of new items; income for vendor. |

and exchange of clothing items through the use of reuse groups and events remains largely unrecorded. Reuse activity within this domain has largely arisen and expanded by virtue of the internet and associated facilities. Web-orientated organizations can offer, in principle, facility for the exchange of clothing items; perusal of currently-available goods and products tends to reveal bias towards, for example, household items, garden equipment and toys. There are web-orientated initiatives becoming available that are focusing more specifically upon clothing (e.g. myfamilyclub.co.uk).

"Swap Shops" for clothing offer a perhaps more suitable forum for exchange of clothing items (WRAP, 2018d). These events involve individuals taking unwanted items of clothing to a venue (e.g. community centre) and select unwanted items that have been brought by others. It is usual practice that direct payments for items of clothing are not involved. Since these events usually involve physical access to items, fuller evaluation of quality, size and suitability for a potential new owner is permitted. So-called "Swishing Events" generally follow the same format and functions as Swap Shops, in cases with emphasis on accessories and shoes as well as clothing (WRAP, 2018d). These are now well-established in the UK and are well-served by guidance and promotion (e.g. Love Your Clothes, 2018; WRAP, 2018b).

**Donation to a charity or third sector organization**

Donations of clothing to charities and third sector organizations (TSOs) are widely considered the main route for clothing reuse in the UK (WRAP, 2016). Such donations may be considered "gifting" (Diop and Shaw, 2018) since the items are gifted to an organization for resale. Donations to charity shops in the UK have been estimated to lead to around 310,000 tonnes of textiles per year being diverted from landfill (Osterley and Williams, 2018). Moreover, there are social and economic benefits. In the UK, activity in the charity/TSO sector is associated with the creation of employment (LGA, 2014); there are around 11,200 charity shops employing some 23,000 paid staff and 230,000 volunteers (Osterley and Williams, 2018). As noted (Table 4), the donor may well value the "feel-good" aspects of such donations through the expected benefit(s) for the receiving charity and those causes supported financially by the subsequent sale of donated items. At the same time, reused items are made available for purchase by those who seek reused clothing at low price, out of desire or necessity (Williams and Shaw, 2017). Both these outcomes are important social benefits of clothing reuse. As noted by the LGA (2014), the low price of secondhand items is in general the biggest motivator for their purchase. Donations to charities and TSOs are also actively promoted by third parties: local authority messages intended to dissuade householders from placing items of clothing in residual waste bins suggest reuse by using charity shops and textile banks, for example (SCC, 2018).

**Selling for personal financial gain**

Direct resale and reuse of personal possessions is far from new. So-called "small ads" (i.e. newspaper advertisements), notes on display in local shops and in specialist publications have for decades been commonly used to advertise a wide variety of possessions. When clothing has been advertised in this manner, focus has frequently been upon items of higher value (e.g. wedding attire, formal wear and men's suits) or "as new" (little worn or unworn), commonly with the intention of selling rather than giving away. Car boot sales have also offered opportunities for resale and reuse of clothing, amongst a wide range of household possessions (Gregson and Crewe, 1994). These events are commonly set according to a repeated schedule; goods are brought in cars or small vans and sold direct from the boot (trunk) of a car, or items are set out on display on folding tables or on tarpaulins on the ground (Williams and Shaw, 2018). There is usually a modest charge for vendors and occasionally a smaller charge for entry by the public.

Use of internet-based technologies is also recognized as a key means to enhance sale, donation or swapping of clothing (Morley et al., 2009). Sales of reused clothing have been particularly influenced by activities in this regard, notably through the launch of the "ebay" internet site in 1995. The advent and wide ownership of smart 'phones and internet-enabled handheld devices have no doubt amplified the role and impact of ebay and similar web-based entities by permitting highly flexible and mobile trading and exchange of reused clothing. The advent of other internet-based businesses has also offered individuals greater opportunities for resale of clothing: there is a plethora of such websites including preloved.co.uk, vinted.co.uk, and whowhatwear.com, for example. It is notable that in many instances the primary focus of such initiatives is personal financial gain, e.g. "6 ways to make money from your closet clear-out"

(Foreveramber, 2017). A small number of companies have started to view the provision of clothing as a service via leasing rather than as a product offering (e.g. MUD Jeans, 2018).

*3.2.3 Clothing reuse in the broader context*

Enhancement of reuse opportunities for clothing focuses not only on the donation and sale of used items for reuse (Table 4) but also adopts a broader view of reuse in the more general context of production, use, resale and recycling. The UK-based Sustainable Clothing Action Plan (SCAP: WRAP, 2018c), for example, advocates "… using collective action to minimise the environmental impact of our clothes." This initiative (Table 5) sets a wide-ranging agenda that recognises the interdependencies and connections to hand. Decisions regarding resources used, clothing designs and materials are critical and can, in principle, be made such that the durability of clothing products is enhanced. If combined with suitable adjustments of consumers' behaviour by persuading people to "make the most of their wardrobe", for example (WRAP, 2018b), the overall impact associated with the production, use, reuse and final destination of end-of-use clothing can be much reduced.

Whilst initiatives such as SCAP (WRAP, 2018c) offer the prospect of more and better with regard to clothing reuse, there remain challenges. There is the prospect that urban mines (Ongondo et al., 2015) represent an under-exploited source of reusable clothing. Estimates suggest that around 30% of the clothes in UK wardrobes have not been worn by their owner for at least one year, and around 80% of individuals own clothes that no longer fit them or need altering in order to be worn (WRAP, 2018b). At the same time, extending the useful lifespan of clothing can be achieved by design and fabric/fibre selection (Table 5); repair can also contribute in this regard, constituting "preparing for reuse" (Table 1). It is arguable, if speculative, that repair skills in the 21st Century are less widespread than hitherto, but there is at least some appetite for learning more about repairing clothes (WRAP, 2018b), albeit lower amongst men (25%) than amongst women (>50%). Skills with regard to the repair of clothing will likely contribute to extending the utility of items, but the role of repair in terms of reuse and its contributions to reuse activities remains largely unknown. Mending an article of clothing may, for example, extend its utility for the existing owner, which has obvious merit in terms of avoided expense of purchasing new items and avoided need for production of new items. Alternatively, repair skills could be employed to mend or alter second-hand items bought at low cost, inferring that repair-orientated skills offer opportunities for cost-saving - as long as there is a desire to purchase reused items and apply repair skills.

## 4. DISCUSSION

The clothing and car sectors clearly differ in terms of reuse practice in some respects. As a broad generalization, the reuse of cars and their components is orientated primarily to resale for financial gain or purchase at lower cost than the equivalent new product or component; some manufacturer-level initiatives (Toyota, 2018) align with motives of resource efficiency (Table 3). There are, however, known challenges regarding component (e.g. tyres; Lebreton and Tuma, 2006) and material (e.g. plastic; Bellemann and Khare, 1999) reuse and practice elsewhere indicates a lack of reuse (Ameilia et al., 2009). In contrast, reuse of clothing appears commonly orientated around acts of altruism, whether to organizations (charities or TSOs) or individuals (§3.2.2), although involvement of clothing-related businesses is clearly aligned with and complementary to existing reuse activities (WRAP, 2018a,c). It is likely that this difference in orientation reflects the relative financial value of clothes and cars; donating an item of clothing may well be financially viable for an individual whilst the value of a used (i.e. reused) car would be too high to be lost through its donation. The impact of this difference is that whilst donations of clothing contribute to the well-being of others (e.g. through income raised by charity shops and/or benefits of free or low cost clothing), the sale of cars or car components has a specific benefit in that reuse supports an extensive market in which cars and components are available at lower cost than the equivalent new products. The social value of car and components reuse is thus, arguably, lower than for clothing.

We note that some commonalities between car and clothing reuse are also apparent. Notably, lifecycle ("cradle to cradle"; Braungart and McDonough, 2009) approaches are evident. Within the car sector, the ambition to produce "easy to dismantle" vehicles (e.g. Toyota, 2018; Table 3) signifies the importance of design in enabling and enhancing the potential for reuse of components. Similarly, the Sustainable Clothing Action Plan (WRAP, 2018c) recognizes the role and importance of product design in improving durability of products as a means to support reuse (Table 5). In both these cases, ongoing initiatives accord well

**TABLE 5:** Initiatives promoted via the Sustainable Clothing Action Plan (SCAP: WRAP, 2018c).

| Initiative | Reuse context: notes |
|---|---|
| Resource efficient business models | Intended to help: create commercial value from sustainable business practices; develop new revenue streams and products from resources previously considered waste. |
| Design for extending clothing life | Improved durability of clothing to increase rates of reuse through extended life and increased desirability for consumers. |
| Fibre and fabric selection | |
| Consumer behaviour and sustainable clothing | Intended to provide practical tips to householders with respect to: reducing environmental impacts of clothing laundry, dealing with unwanted clothes, and making the most of their wardrobe (WRAP, 2018b). |
| Reuse and recycling | Voluntary agreement intended to enable re-use organisations, reprocessors and local authorities to increase collection rates through partnerships, schemes, and advice on good practice. |

with the aims and principles of the circular economy (Ellen MacArthur Foundation, 2017; WRAP, 2018a).

A further commonality is also evident in that the internet provides a means for reuse to take place for clothing, cars and car components. Whether allied with donations (to acquaintances or persons unknown) or sales, internet-based initiatives offer access to products at a scale that arguably surpasses all foregoing means and opportunities. For accessing reused products for sale, internet-based technologies provide searchable and conveniently-accessed databases for prospective buyers. This facility is broadly and commonly available for the purchase of reused cars and car components (§3.1), and for the sale or donation of clothing (§3.2.2). Searchable internet-based databases permit access to a wide range of items for sale over a large geographic area, providing insight to the range, variety and costs of items being sought. From a purchaser's perspective, the ease of access is perhaps counteracted in part by the inability to inspect closely items of interest. For clothing, size, fit, condition and texture will not be readily assessed. For cars, potential purchasers will frequently arrange a viewing prior to deciding whether to purchase a vehicle; for parts or components located on internet databases, scrutiny is likely less common or practicable, depending on the proximity of the vendor and potential buyer. In both cases, the prospect of "buying blind" incurs risks for the purchaser. Given the ongoing reliance of human society on digital and mobile communications, we suggest that it is likely that internet will continue to facilitate reuse activities and grow in this regard.

The decisions regarding why and how end-of-use decisions are made merits consideration. Whilst the foregoing review identifies and evaluates examples of current practice, it must be borne in mind that reuse is of lower preference than "reduce" in relation to the objectives of the waste hierarchy (EC, 2008; Williams, 2015). An individual's decision that a possession is no longer wanted or needed is critical in this regard; arguably, the differentiation between "want" and "need" is subjective but critical. Moreover, continued use of products – in the case of both cars and clothing – is contingent on maintaining their utility. In this respect, repair may be of particular importance. The issue of economic viability of repairs to cars is critical, i.e. the cost of repair(s) relative to the value of the vehicle must be favourable. Availability of reused components offers a means to reduce the cost of replacement parts and thereby favourably alters the balance of repair costs in relation to vehicle value, provided that the vehicle owner has necessary and sufficient skills to carry out the repair or can locate a technically-competent mechanic willing to work with reused replacement parts. For clothing, carrying out repairs and alterations relies on skills and, if carried out by the owner, may incur little in terms of cost whilst requiring time.

Finally, we observe that there remain challenges to the enhancement and expansion of reuse in both the sectors considered in the present study and more broadly within the consumer goods sector. Reuse has, until relatively recently, developed on a progressively evolving basis, led by opportunities that serve the needs, aspirations and desires of the population. The advent of whole lifecycle thinking (including aspirations to progress to a circular economy) introduces a need for deeper and wider understanding. Paras et al. (2018) proposed six primary drivers that need to be recognized in the implementation of a reuse-based value chain: the system at hand, redesignability and price of products, information, legislation, and consumer attitude. For the car components sector there is an additional challenge in that systems for disassembly must be considered and appropriately designed (Wahab et al., 2008; Go et al., 2011).

If achievable, a shift to a value chain approach could, in principle, be of considerable merit, but we note that the reuse practices presented in this study all involve, in some shape or form, a necessity for consumers to align their motives and actions with the aims and facilities of systems for reuse. The advancement in technology may hinder progress, but in different ways for cars and clothing. There is potential for tension between increasing technological complexity (e.g. of cars and their mechanical systems) that may well render repair or replacement the domain of technical experts. For clothing, the durability of products may be reduced by both fashion considerations and by a shift to cheaper materials and lower production costs. It is difficult how to envisage how such changes might be accommodated within a "cradle-to-cradle" lifecycle design approach as set out in the SCAP (Table 5; WRAP, 2018c). Such changes and differences between product sectors need to be taken into account when designing systems for reuse (Tables 3 and 5); this comprises an area in which sector-specific aspects will need to be taken into account and fully recognised.

## 5. CONCLUSIONS

As noted by Williams and Shaw (2017), it is clear that reuse has a critical and central role in progressing towards more sustainable use of resources orientated to circular economy thinking; it is crucial that reuse continues to serve and contribute to our ambitions to achieve responsible production and consumption (UN, 2015). If we as a society are to achieve this ambition, sector-specific adjustments may have to be implemented in future initiatives to promote and enhance reuse activities. Overarching principles and optimum methods of reuse facilitation may well, however, be common for different sectors. In this regard, the shift towards full lifecycle product design, incorporating all stages from "cradle to cradle" (Braungart and McDonough, 2009; WRAP, 2018c) has, we believe, considerable merit as a means to enhance and expand reuse activity. Such approaches, however, are unlikely to succeed in their aims unless consumers are fully engaged in reuse. For their engagement to occur, the internet in particular provides opportunities for reuse, offering access to and choice of reused products that appears independent of sector in the case of cars and clothing and may indeed provide a tool for more and better reuse in other sectors.

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# IN THE SEARCH FOR EFFECTIVE WASTE POLICY: ALIGNMENT OF UK WASTE STRATEGY WITH THE CIRCULAR ECONOMY

Carly A. Fletcher * and Rachel M. Dunk

*School of Science and the Environment, Manchester Metropolitan University, Manchester, United Kingdom*

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**ABSTRACT**

Over-consumption within a linear economy has been recognised internationally as a barrier to sustainability and a major cause of environmental degradation and economic disparity. To address these issues, the transition towards a circular economy (CE) has been advocated. A broad resource efficiency concept, the CE seeks to reduce consumption, encourages the reuse and recycling of materials and products, and encompasses the three pillars of sustainable development; economic prosperity, environmental protection, and social equity. Efforts to implement the CE have seen the introduction of various hierarchies that prioritise the implementation of R-imperatives (such as the '3Rs' of reduce, reuse, recycle). One such example is the waste hierarchy, originally introduced to encourage sustainable waste management and more recently reiterated by the EU Circular Economy Package as a means to stimulate the transition to the CE. Following the development of a CE Framework, this study presents a content analysis of the waste strategies of the four devolved nations of the United Kingdom. Key differences and similarities in the strategies of the four devolved nations are identified and discussed in light of CE aims, core concepts and principles (with particular focus on promotion of the waste hierarchy), enablers, and stakeholder engagement, where Scotland and Wales were found to have the most progressive strategies. This study also considers the potential impact of Brexit, where it is recommended an overarching UK-wide strategy that provides consistent and collaborative long-term objectives is required to replace the overarching objectives previously supplied by the EU policy.

## 1. INTRODUCTION

Rising global population and a growing trend towards higher living standards have led to increased depletion of natural resources and environmental degradation due to the linear 'take-make-dispose' model on which economic growth has been built (Moreno et al.,, 2016; Wysokinska, 2016). The need to overcome this unsustainable pattern of consumption has been acknowledged internationally, most notably through the prioritisation of Sustainable Consumption and Production (SCP) within the United Nations 10-year Framework of Programmes (UNEP, 2015), where a key component of SCP is the transition to a Circular Economy (CE).

At its core, the CE is a broad resource efficiency concept (Su et al.,, 2013) that seeks to mimic natural biological systems by continuously recirculating and reprocessing materials and energy (Lieder and Rashid, 2016). As discussed by Winans et al., (2017), the CE model has evolved continuously since the 1970s, building on and encompassing a number of preceding ideas. It is deeply rooted in resource efficiency concepts that advocate moving from end-of-pipe solutions to life cycle and systems thinking. For example, Stahel and Reday-Mulvay's (1976) vision of a 'loop economy' that returns durable products from cradle-to-cradle, and Pearce and Turner's (1990) argument for a shift from a 'resources-products-pollution' to a 'resources-products-regenerated resources' mode. Additionally, it includes the recognition that there are limits to growth (Meadows et al., 1972), and that human industry relies on resources and services provided by the biosphere (industrial ecology) and cannot therefore be considered in isolation from it (Erkman, 1997). Indeed, Murray et al., (2017) argues that rather than promoting biomimicry, the CE should aim to "bio-participate" where actions take place within the existing biosphere.

Despite widespread agreement for the need to transition to a CE, a standardised definition of the CE (or understanding of what transition entails) has been lacking, where this has been attributed to both the evolving nature of the concept and the use of the concept by stakeholders from different disciplinary or industrial backgrounds (Kirchherr

* Corresponding author:
Carly A. Fletcher
email: carly.fletcher@stu.mmu.ac.uk


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et al., 2017). Analysis of 114 definitions found that while the CE is often defined using hierarchies of R-imperatives (e.g. the 3Rs of reduce, reuse and recycle), the systemic change needed to implement the CE is frequently overlooked (Kirchherr et al., 2017). Furthermore, definitions are limited by insufficient linkages to other aspects of sustainable development, primarily focusing on economic prosperity followed by environmental protection, and with very limited consideration of social equity or future generations (Kirchherr et al., 2017). To address this shortcoming, Kirchherr et al., (2017) proposed a standardised definition of the CE as "an economic system that is based on business models which replace the 'end-of-life' concept with reducing, alternatively reusing, recycling and recovering materials in production/distribution and consumption processes, thus operating at the micro level (products, companies, consumers), meso level (eco-industrial parks) and macro level (city, region, nation and beyond), with the aim to accomplish sustainable development, which implies creating environmental quality, economic prosperity and social equity, to the benefit of current and future generations."

Intended to function as a fully regenerative closed ecological-economic system (Lieder and Rashid, 2016), resource use should be reduced by embedding R- imperatives at all stages of design, production, distribution and consumption (Su et al., 2013; Wysokinska, 2016). Thus, in addition to clean production techniques, maintaining the flow of materials and energy within the CE requires a combination of innovative product design, extended producer responsibility (EPR), new business models, and consumer behaviour change, and hence necessitates stakeholder engagement across the supply chain (EMF, 2015; Lieder and Rashid, 2016; Stahel, 2016; Su et al., 2016; Tukker, 2015; Wysokinska, 2016).

Eco-design and regenerative-design approaches play a critical role, where resources are either designed out through dematerialisation or can be readily regenerated at end of life, and where product and component lifespan is extended through increased durability, repairability, and the standardisation of components (Leider and Rashid, 2016; Wysokinska, 2016). Reuse and regeneration can be further enhanced through EPR and the implementation of reverse cycles within the supply chain, whereby products and materials are returned to the producer to be reused or reprocessed (EMF, 2015; Wysokinska, 2016). Product utility can also be increased by changing the business model in which products are sold and consumed or through consumer reuse (Stahel, 2016). Providing products through service agreements, such as pay-per-use, sees the producer retain responsibility and therefore incentivises resource efficiency and product utility above unit sales (Stahel, 2016; Tukker, 2015). With respect to consumer reuse, the emergence of the 'sharing economy', in which underutilised assets are shared (or re-sold) through peer-to-peer interactions within community-based (online) services, may not only enable more efficient use of products but also deliver economic and social benefits (Cherry and Pidgeon, 2018; Martin, 2016). However, the extent to which these benefits are realised is unclear, with concerns that the sharing economy may lead to increased overall consumption (Cherry and Pidgeon, 2018; Martin, 2016).

The transition to a CE is often viewed as synonymous with or requiring a movement towards 'zero-waste' (e.g. Ghisellini et al., 2016). Zero-waste (ZW) can be defined in various ways including ZW to landfill and ZW emissions to land, sea and air, but generally requires sustainable waste management and increased resource utility (Cole et al., 2014). However, while the two concepts are clearly complementary, they can be viewed as subtly different (Veleva et al., 2017), with implications for policy development for appropriate emphasis. For example, Veleva et al., (2017) argue that ZW approaches focus primarily on recapturing resources from waste streams, reducing consumption, and applying a life cycle approach to product design, whilst the CE extends beyond this by designing out waste and introducing innovative business models and collaborative platforms to continuously reuse materials. The CE also emphasises use of renewable materials and energy and places a stronger emphasis on return of biological nutrients to nature. As such, pursuing ZW might be viewed as incremental continuous improvement, whilst in comparison the CE could be seen as transformative.

Although there is strong agreement regarding the urgent need to shift to more sustainable patterns of consumption and production, limitations of the CE (as described above) and barriers to transition have also been identified. Jawahir and Bradley (2016) argue that while the socio-political dimensions and opportunities of the CE are being pursued and promoted, the technological challenges to implementation are often overshadowed, where they highlight the need to promote innovation and expand thinking to consider multiple, intersecting lifecycles. Similarly, Murray et al., (2017) highlight the risk of unintended consequences, where impacts are transferred due to over simplistic goals that are based on reductionist thinking and mathematical models. Andrews (2015) suggests that the transition towards the CE may be limited due to the presence of materials and products that are difficult to reuse or recycle and a lack of knowledge and understanding of relevant stakeholders. Likewise, Kirchherr et al., (2017) highlight that consumers, and their role as key enablers of the CE, are frequently neglected.

Thus, to aid transition to the CE, the challenge for policy makers is to engage with all stakeholders to reduce consumption, enable and develop new markets, encourage innovation, and promote resource efficiency (Price, 2001; EMF, 2015; Lieder and Rashid, 2016). Three levels of contributing stakeholders should be acknowledged in successful CE policy; micro-level such as individual consumers, designers and producers; meso-level, including community groups, individual sectors and industrial parks; and, macro-level, encompassing cities and regions (including local authorities, national government and regional administrations), co-operative networks and multi-national businesses (Su et al., 2016; Kirchherr et al., 2017). Nevertheless, some have argued that the most important role of overarching strategies and policies is to engage and inspire individuals to consume less, reuse goods, and present high quality recyclate when waste is unavoidable (Price, 2001; EMF, 2015).

As previously noted, the R-imperatives are recognised as a building block of the CE, where ranking these imperatives (in order of preference for value retention) within R-hierarchies is viewed as necessary to provide guidance and promote effective implementation of the CE (Kirchherr et al., 2017; Murray et al., 2017). However a recent review of CE literature found significant variation in the number of R-imperatives used (between 3 and 10), the combination of imperatives, choice of terminology, and assigned meanings (38 different R-imperatives were identified with varying definitions), whether or not the imperatives were ranked, and in cases where they were ranked, their relative position (Reike et al., 2018). While efforts have been made to develop nuanced hierarchies employing a high number of R-imperatives, thereby providing an operationalisation principle than maximises resource value retention, inconsistencies remain (Potting et al.„ 2017; Reike et al.„ 2018). Nonetheless, as argued by Kirchher et al., (2017), without the use of R-hierarchies that explicitly identify waste prevention imperatives as the highest priority, the concept of CE could be subverted, resulting in limited and minimal changes when implemented. Furthermore, the introduction of strategies that address current (lower) priorities can lead to 'lock-in', where they lack the flexibility to change in the future and so the emergence of more sustainable strategies is restricted (Foxon, 2002). Perhaps one of the most consistent expressions of an R-hierarchy is the "waste-hierarchy" (4R-imperatives of reduce, reuse, recycle, recover, followed by dispose), introduced as a tool to promote sustainable waste management (Van Ewijk and Stegemann, 2014) and referenced in the definition of the CE proposed by Kirchherr et al., (2017). This hierarchy has been particularly visible over the last ten years, where continual development of European Union (EU) waste policy has repeatedly reiterated the waste hierarchy, including re-casting it as a tool to promote the CE within the EU Circular Economy Package (CEP).

This paper examines the circularity of current waste strategy within the UK. As the UK is currently a member of the EU, a brief review of EU waste policy is presented first, followed by current waste policy in the UK and the potential impact of Brexit. Using an adapted CE-framework, a content analysis is then conducted to assess and compare the four devolved nations in terms of CE aims, core concepts and principles (with a focus on use of the waste hierarchy as an operationalisation principle), enablers, and stakeholder engagement.

### 1.1 Abbreviations

CE: Circular Economy
CEP: Circular Economy Package (EU)
EFTA: European Free Trade Association
EPR: Extended producer responsibility
EWSR: European Waste Shipment Regulations
EU: European Union
SCP: Sustainable Consumption and Production
ZW: Zero waste

## 2. CONTEXT

### 2.1 EU Circular Economy Package

Already a leader in environmental policy (Wysokinska, 2016), the adoption of the CEP by the EU will introduce new priorities that advocate resource efficiency and initiate the transition towards a CE (EC, 2017). While previous strategies such as the 'Roadmap to a resource efficient Europe' (2011-2013) and 'Towards a circular economy: a zero-waste programme for Europe' (2014-2015) promoted the CE, their emphasis remained on the efficient use and management of waste. In contrast, the CEP aims to prioritise the CE and address inherent limitations of previous policy initiatives, including a shift in focus toward full product lifecycle thinking (EC, 2017). Although the CEP does encourage industrial symbiosis and the development of secondary materials markets, it also retains an emphasis on waste management strategies such as reiterating the need to implement the waste hierarchy and revising targets for landfill diversion and recycling (Table 1; EC, 2017; Pomberger et al., 2016).

While full implementation of the waste hierarchy would align with CE ideals, Van Ewijk and Stegemann (2014) and Gharfalkar et al., (2015) argue that the limited specification of prevention, the absence of a distinction between open- and closed- loop recycling, and the lack of inclusion of other sectors could constrain dematerialisation and resource effectiveness. Other authors have also argued that too little emphasis is placed on the higher priority R-imperatives. For example, while reducing waste at source is the most effective and efficient CE strategy, the absence of quantitative targets for reduction or reuse can create a perceived policy bias towards recycling and disposal (Mazzanti and Zoboli, 2009; Fischer, 2011). However, it is noted that the CEP is supported by other initiatives such as the "Thematic Strategy on the Sustainable Use of Natural Resources" (EC, 2005), the "SCP Action Plan" (EC, 2008) and the "Integrated Product Policy", which includes elements such as eco-design, eco-labelling, and green public procurement (EC, 2016). In combination, these approaches aim to reduce the environmental impact of resource use while promoting economic growth through improving the environmental performance of goods and services across the full lifecycle and creating sustainable business opportunities (EC, 2005, 2008, 2016). This provides a framework of strategies and policy objectives that individual member states should adhere to, to aid their transition toward a CE.

### 2.2 Current UK waste policy

The multilevel governance character of the EU sees overarching objectives published centrally, with decisions regarding the approaches and instruments used to achieve these objectives resting with the individual member states (Nilsson et al., 2012). There are several reported techniques by which EU policy is transposed into national policy including "copy-out" (using the exact words and phrasing of the EU directive), "gold-plating" (going beyond the minimum stated requirements), and "no gold-plating" (consists of only the minimum requirements; Anker et al., 2015). This degree of member state discretion has led to significant

**TABLE 1:** Current EU waste management targets and the proposed amendments set out in the Circular Economy Package (CEP).

| Waste Stream | Existing Policies and Targets | | Circular Economy Package Proposals | |
|---|---|---|---|---|
| Municipal Solid Waste | **Landfill Directive (EC, 1999)** When compared to 1995 base year, the share of biodegradable municipal waste going to landfill may not be greater than 75% by 2006, 50% by 2009, and 35% by 2016. | | **Proposed Amendment (EC, 2018b)** Bans disposal to landfill of separately collected wastes and extends landfill diversion target to all municipal waste, where the share of municipal waste sent to landfill is limited to 10% by 2035. | |
| | **Waste Framework Directive (EC, 2008)** By 2015, separate collection shall be set up for at least paper, metal, plastic and glass. Preparing for re-use and recycling of 50% of at least paper, metal, plastic and glass from household and similar sources by 2020. | | **Proposed Amendment (EC, 2018d)** Extends preparation for re-use and recycling to all municipal waste, with targets of 55% by 2025, 60% by 2030, and 65% by 2035. | |
| Construction & Demolition Waste | **Waste Framework Directive (EC, 2008)** Preparing for reuse, recycling and other recovery such as backfilling of 70% of non-hazardous construction and demolition waste by 2020. | | **Proposed Amendment (EC, 2018d)** No extension to existing target, but requires introduction of measures to promote selective demolition and removal of materials, and to establish sorting systems for at least wood, mineral fractions (concrete, bricks, tiles and ceramics, stones), metal, glass, plastics and plaster. | |
| Packaging Waste | **Packaging Waste Directive (EC, 1994)** By 2008 60% of packaging waste to be recovered, with a minimum of 55% and maximum of 80% to be recycled, and minimum recycling rates for specific materials as follows: | | **Proposed Amendment (EC, 2018c)** Removes the maximum and extends the minimum recycling rates for all packaging waste to 65% by 2025 and 70% by 2030, and extends the targets for specific materials as follows: | |
| | wood: | 15% | wood: | 25% and 30% |
| | plastics: | 22.5% | plastics: | 50% and 55% |
| | metals: | 50% | ferrous metals: | 70% and 80% |
| | | | aluminium: | 50% and 60% |
| | glass: | 60% | glass: | 70% and 75% |
| | paper and board: | 60% | paper and board: | 75% and 85% |

differences in national implementation of resource and waste policy (Garcia Quesada, 2014).

Over the last two decades, UK environmental legislation has been largely shaped by EU directives, where it is a notable feature of UK waste policy that secondary legislation is used extensively to transpose EU law into domestic law (Scotford and Robinson, 2013). Indeed, EU legislation has provided momentum to improve waste management in the UK, lifting it above the national party politics that previously hindered the development and implementation of a long-term strategy (UKELA, 2016; BP Collins, 2016). During this time, the UK has introduced fiscal instruments such as the landfill tax, extended separate recyclate collections, and increased exports of refuse derived fuel, all of which have aided a transition away from high landfill dependency (Pomberger et al., 2016). However, due to a plateau in progress potentially caused by the "no gold-plating" approach of transposition, the development of new measures that manage resources rather than waste are now required to maintain the momentum of positive change.

## 2.3 Impact of Brexit

Although the UK is currently negotiating its withdrawal from the EU (termed "Brexit"), it is expected that the CEP will be transposed into UK law. Once the UK has fully withdrawn from EU membership it will no longer be obligated to transpose or adhere to EU directives. However, while the official withdrawal date is the 29th March 2019, a transition period extending to 31st December 2020 has recently been agreed, during which EU law "shall be applicable to and in the UK" (EC, 2018a). Hence, as the amendments to existing directives proposed under the CEP (EC, 2018b-d) are expected to enter into force in 2018 and require transposition within eighteen months, the UK will be obligated to transpose them. As noted above, current UK environmental law is highly dependent on that of the EU, where the UK will convert the existing body of EU environmental law into domestic law on 'exit day' through a blanket transposition under the Withdrawal Bill (European Union (Withdrawal) HL Bill (2017-19) 79). However, after the end of the transition period the UK would not be obligated to adhere to the CEP, where UK governments could act to repeal or amend the transposed domestic law (BP Collins, 2016; UKELA, 2017). This leads to the question of how UK waste and resource management will develop in the absence of the long-term vision and strategy provided by the EU. Current commentary on post-Brexit waste policy suggests that in the short term the UK would continue to apply existing EU legislation and strategy (Burges Salmon, 2016; BP Collins, 2016). However, in the medium to long term it is difficult to predict whether successive UK governments would maintain compliance with current and successive EU legislation, look to go beyond them, or maintain the current status quo, with the risk of being left behind (Burges Salmon, 2016).

Other potential implications of Brexit for waste management in the UK and for other EU member states have also been highlighted, particularly in relation to cross border movement of wastes (House of Lords, 2017; UKELA, 2016, 2017). Gibraltar (a British overseas territory) is completely reliant on Spain for its waste management (both collection and treatment) and the Republic of Ireland exports 40% of its hazardous waste to the UK due to the lack of capacity in local treatment facilities (McGlone, 2018). The UK also exports a significant tonnage of waste derived materials to other EU member states. Indeed, exports of waste derived fuel to European countries have increased from zero in 2010 to over 3 million tonnes in 2016 (DEFRA, 2017; UKELA, 2016). Likewise, due to limited domestic processing capacity, exports of recyclable materials have risen from around 8 million tonnes in 2002 to around 14 million tonnes in 2015 (DEFRA, 2017), where around a quarter of sorted waste materials are sent to northern European countries which have an overcapacity in processing facilities (House of Lords, 2017).

Post-Brexit, the movement of waste between the UK and EU countries must adhere to the European Waste Shipment Regulations (EWSR) (EC, 2006). Under the EWSR, the import of waste is allowed from a third (non-EU) country that is a party to the Basel Convention on the Control of Transboundary Movement of Hazardous Wastes and their Disposal (the Basel Convention) (UNEP, 1989). However, export of waste for disposal or mixed municipal waste for recovery to a third country is prohibited, unless it is both a party to the Basel Convention and a member of the European Free Trade Association (EFTA). Furthermore, imports and exports of waste between the UK and the EU will most likely become subject to border checks and depending on the outcome of negotiations could become subject to tariffs (EC, 2018a), with the risk that such shipments become financially unviable (House of Lords, 2017).

The future status of the UK with respect to the Basel Convention (an international agreement ratified jointly by the EU and the UK) is uncertain. Analysis indicates that the effect of Brexit on such "mixed agreements" is somewhat ambiguous, with some analysts concluding that they will have to be renegotiated, and others adopting the position that the UK will remain bound by them post-Brexit (UKELA, 2017). Nonetheless, while the status of mixed agreements remains to be clarified, the UK government has expressed the view that the UK is a party in its own right and will continue to be bound by such agreements post-Brexit (House of Lords, 2017).

The UK joining the EFTA post-Brexit has been posited as a potential option, in which case waste exports from the EU to the UK could continue with respect to EWSR, however access to the single market (so as to avoid import/export tariffs) would require the UK to continue to adopt the relevant evolving EU acquis. Furthermore, for any recovery of waste generated by EU member states and exported to the UK, the EU member state will only be able to count that waste towards fulfilment of EU targets if the treatment conditions are equivalent to the requirements of applicable EU directives (EC, 2018d).

All of the above Brexit related uncertainties regarding the future of waste management in the UK are further complicated by the differing positions of the devolved nations. The devolution of power in the UK allows the four home nations (England, Scotland, Wales and Northern Ireland) to manage waste and resources within their own boundaries while contributing to overall UK objectives. This has led to the introduction of different strategies by the four nations. Indeed, based on an evaluation of primary and secondary environmental legislation, Scotford and Robinson (2013) argue that Wales and Scotland are providing the most innovative legislation developments within the UK.

## 3. METHODS

A content analysis was used to assess the current waste management strategies of the four devolved administrations of the UK home nations. Based on a CE framework adapted from Kirchherr et al., (2017) in light of the literature reviewed above, the main themes explored within the analysis were; CE aims, core concepts and principles, enablers, and stakeholder engagement, with a particular focus on the promotion of the waste hierarchy as an operationalisation principle and the inclusion of stakeholders. The analytical framework is presented in Figure 1, where the correspondence between the waste hierarchy and a more nuanced hierarchy of R-imperatives is presented in Figure 2. Here the R-hierarchy is synthesised from Potting et al., (2017) and Reike et al., (2018), and modified to align with the EU waste-hierarchy, such that repair without change in ownership (by a consumer or under a product-service agreement) to extend product life is viewed as a waste prevention measure (as the product is not discarded and has not become a waste). The role of re-servitisation and re-modelling business and actions that can be undertaken by consumers (italic text) as enablers of high priority R-imperatives are also highlighted.

Content analysis has been widely employed as both a qualitative and a quantitative method across a range of policy areas including: health (e.g. Lemiengre et al.,, 2008), environment (e.g. Maczka et al.,, 2016), serious crime (e.g. Paoli et al.,, 2017), procurement (e.g. Testa et al.,, 2016), and cleaner production (e.g. Peng and Liu, 2016). It provides a simple yet flexible method to describe and quantify phenomena, analyse written, verbal or visual communication, and enhance the understanding of data through the exploration of theoretical ideas (Elo and Kyngäs, 2008). It also allows the inclusion, comparison and corroboration of large volumes of textual data from different sources (Elo and Kyngäs, 2008). To do this and ensure reliability, analysis should be objective, systematic and quantitative whereby categories of analysis are precisely defined, and the inclusion/exclusion of documents is based on consistent rules (Testa et al.,, 2016).

Taking these factors into account, the most recent waste management strategies published by each of the home nations were selected for inclusion in this study:

- England: "Waste Management Plan for England" (DEFRA, 2013);
- Scotland: "Scotland's Zero Waste Plan" (Natural Scotland, 2010);
- Wales: "Towards Zero Waste - One Wales: One Planet" (WAG, 2010);
- Northern Ireland: "Delivering Resource Efficiency" (DoE, 2013).

Only main body text was analysed with all other text (front matter, legends, footnotes, etc.) excluded. To ensure rigour, two researchers assessed all documents, with points of ambiguity or disagreement discussed and clari-

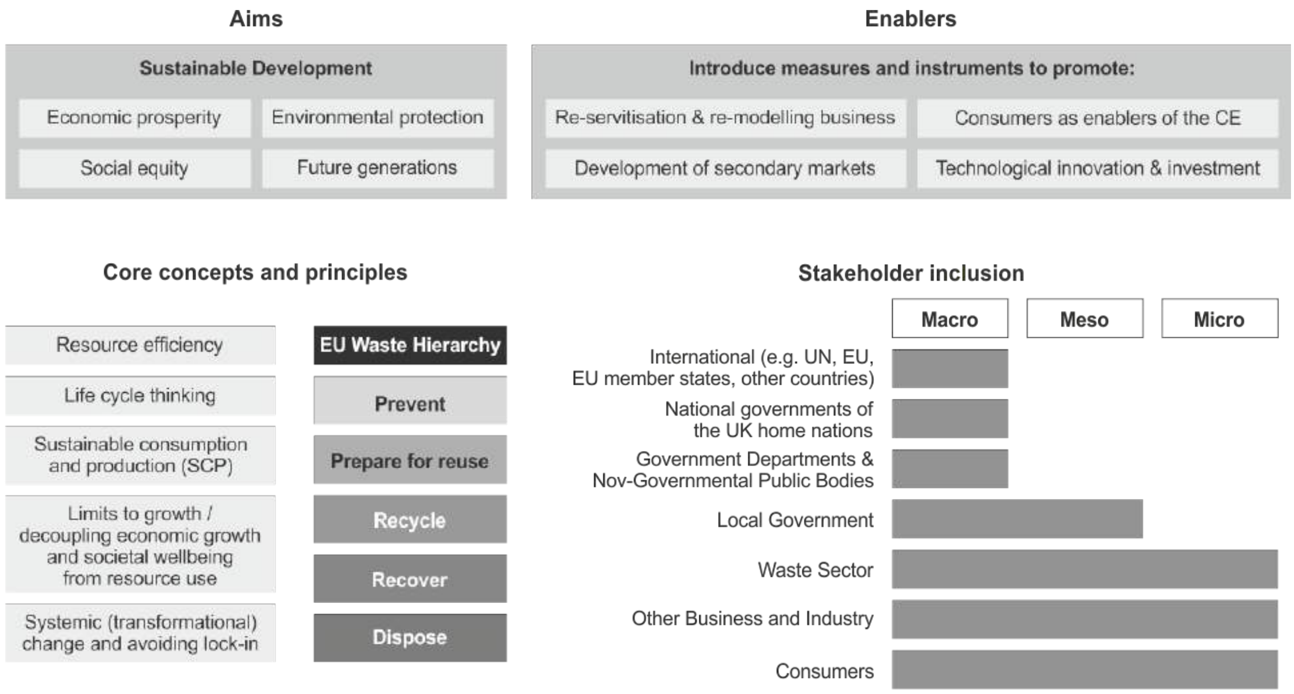


**FIGURE 1:** Circular Economy Framework (adapted from Kirchherr et al.,, 2017).

fied. The use of CE or ZW terminology, the broader context within which waste management was positioned, and the overarching approach of each strategy document was first explored. This was supported by the compilation of national statistics regarding population and rates of waste generation, recycling and landfilling (based on DEFRA, 2018). Using a basic automated keyword search, and manual analysis to ensure complete coverage, the inclusion of CE aims, core concepts and principles, and enablers was evaluated (Corbin and Strauss, 2008; Welsh, 2002). Waste hierarchy R-imperatives and stakeholder terms were also quantified on both a total document and per paragraph basis, and documents ranked (based on per paragraph counts) to compare incorporation of waste hierarchy R-imperatives and stakeholder engagement. Additionally, the responsibilities of each stakeholder group were noted and compared.

## 4. RESULTS AND DISCUSSION

### 4.1 Context and overarching vision

Table 2 presents a summary of the waste strategy document for each home nation, including the volume of text analysed, the context, and the overall vision, alongside population and waste statistics for the document year and for 2016 (Defra, 2018).

*4.1.1 England*

The stated aim of the Waste Management Plan for England is to work towards a ZW economy as part of the transition to a sustainable economy. Here, a ZW economy is defined as one within which material resources are reused, recycled or recovered wherever possible and only disposed of as the option of last resort, where the need to reduce waste generation and ensure all materials are fully valued during their productive life (in addition to at end of life) are also recognised. However, the substance of the plan focuses primarily on minimising the environmental and human health impact of waste generation and management, where this is achieved by supporting local authorities (and waste management companies) to prioritise recycling and recovery of waste materials. While it highlights the role of ZW initiatives and advocates lifecycle thinking and closed loop approaches, it provides little more than rhetoric regarding these ideas. For example, although it does imply that resources should be used efficiently, rather than introducing governmental drivers to achieve this, it places the responsibility on business and industry for creating more goods and services with fewer resources.

*4.1.2 Scotland*

Scotland's Zero Waste Plan defines a ZW Scotland as one that makes the most efficient use of resources by minimising demand on primary resources and maximising the reuse, recycling and recovery of resources instead of treating them as wastes. It frames waste management strategy within the context of economic growth and climate change, where resources are managed efficiently, economic opportunities are sought (and capitalised upon), waste materials are given a value, and greenhouse gas emissions are reduced. To do this, it advocates a transition away from a linear economy, long-term policy stability, and effective resource use. It also acknowledges the role of consumer behaviour, asking individuals and businesses to recognise and take responsibility for their actions. It recognises the need for continued waste management strategies for the foreseeable future and promotes the reuse, recycling and recovery of resources from waste in line with the waste hierarchy.

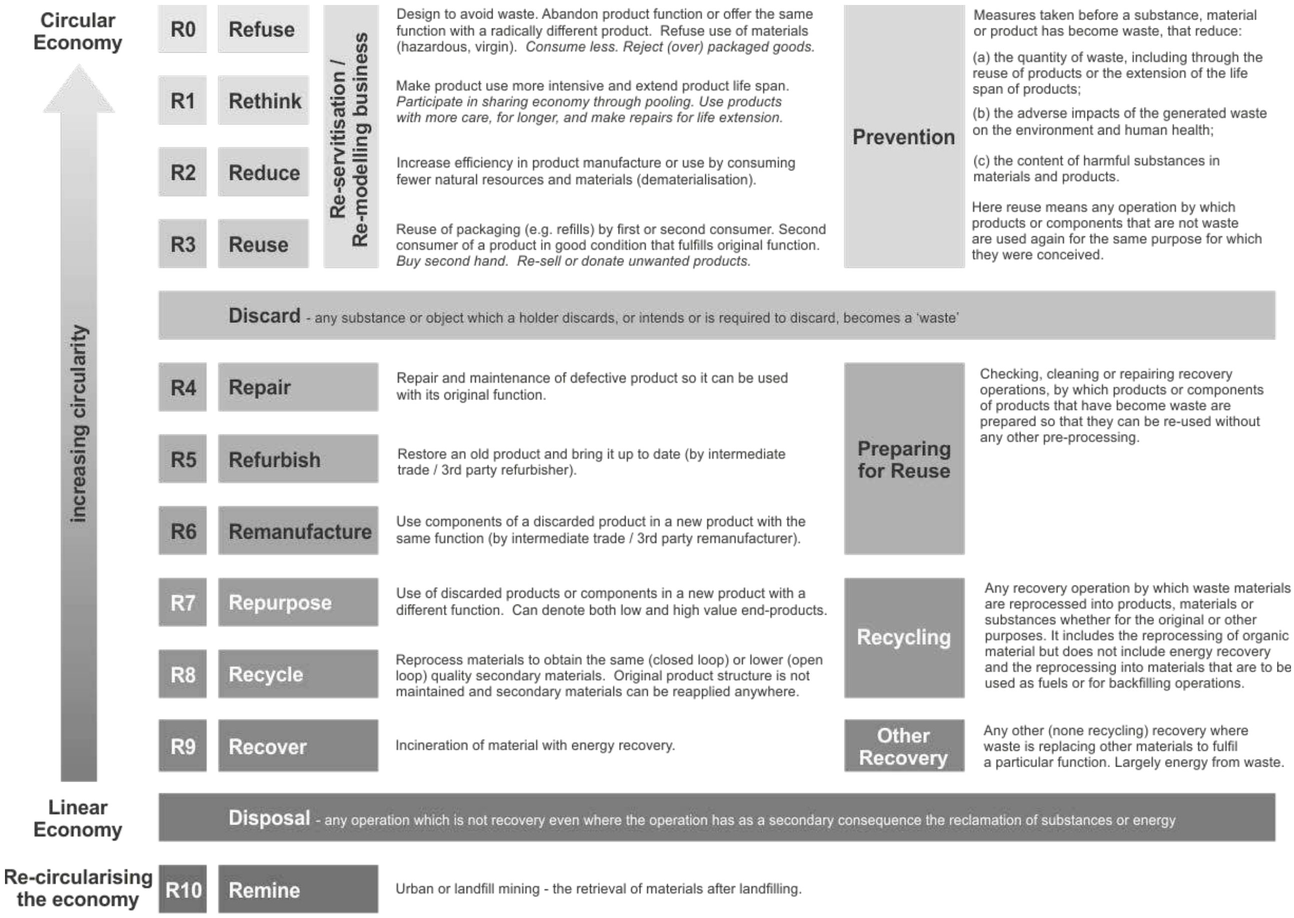


**FIGURE 2:** Alignment of the EU Waste Hierarchy with the R-Imperatives (R0 – R10) needed in the transition to the circular economy (R-imperatives synthesised from Potting et al.,, 2017 and Reike et al.,, 2018).

### *4.1.3 Wales*

Towards Zero Waste – One Wales: One Planet defines ZW as an aspirational end-point where all waste that is produced is reused or recycled as a resource, without the need for any landfill or energy recovery. It frames waste management strategy in the broader context of social justice, cultural legacy, climate change and limited resources. It aims to create a pathway to where resource use is within environmental limits, society and culture prosper, and human well-being is maximised. To do this, it advocates SCP, optimisation of material utilisation, and reduced dependence on primary resources. It promotes a long-term framework that requires the engagement of citizens, business and industry. Citizens are asked to rethink and reconsider consumption patterns, and to become a recycling society, whilst business and industry are asked to use alternative materials, employ Integrated Product Policy and reduce associated emissions. It acknowledges the continued production of some waste and so advocates enhanced action on waste prevention, maximised recycling and near ZW to landfill. It also notes the requirement to manage legacy wastes.

### *4.1.4 Northern Ireland*

The Delivering Resource Efficiency strategy aims to set a direction towards treating waste as a resource and using it more efficiently. This is positioned within the EU objective of moving towards a CE, and although no definition of a CE is given, it is noted that it requires a greater focus on waste prevention followed by an increase in recycling. The strategy is positioned in the context of economic growth, whereby sustainable waste management can promote green jobs, maximise opportunities, and contribute to a low carbon, CE It identifies the need for both socially responsible economic growth and global economic transformation to address depletion of finite natural resources and climate change. To do this, it advocates the implementation of the waste hierarchy, recognition of waste as a resource, use of environmentally friendly technology and behaviours, and increased integrated support across sectors and between stakeholders.

## 4.2 CE aims, core concepts and principles, enablers and stakeholders

Development of the CE framework allowed the systematic, yet simple, assessment of documents. When CE aims, core concepts and principles (including promotion of the waste hierarchy), enablers, and the inclusion of stakeholders were considered, both similarities and substantial differences were found between the waste strategies of the UK home nations.

**TABLE 2:** Summary of UK home nations population, waste generation and management statistics, and national waste strategy documents.

| | England | | Scotland | | Wales | | Northern Ireland | |
|---|---|---|---|---|---|---|---|---|
| **Year of statistics** | **2013** | **2016** | **2010** | **2016** | **2010** | **2016** | **2013** | **2016** |
| Population | 53.9m | 55.3m | 5.3m | 5.4m | 3.0m | 3.1m | 1.8m | 1.9m |
| Waste generation(i) | 400 kg | 410 kg | 490 kg | 440 kg | 450 kg | 420 kg | 430 kg | 450 kg |
| Recycling rate | 44.2% | 44.2% | 32.5% | 42.8% | 44.0% | 56.7% | 41.5% | 43.0% |
| Landfill rate(ii) | 25% | 21% | 41% | 30% | 33% | 16% | 24% | 27% |
| **Strategy document** | **Waste Management Plan for England** | | **Scotland's Zero Waste Plan** | | **Towards Zero Waste One Wales: One Planet** | | **Delivering Resource Efficiency** | |
| Total pages | 42 | | 59 | | 92 | | 68 | |
| Pages analysed | 38 | | 46 | | 59 | | 51 | |
| Paragraphs analysed | 194 | | 288 | | 357 | | 374 | |
| Words analysed | 10,943 | | 13,746 | | 13,768 | | 19,604 | |
| CE Terminology | zero waste | | zero waste | | zero waste | | circular economy | |
| Context | Minimise environmental & human health impacts | | Economic growth & addressing climate change | | Social & cultural justice, climate change & limited resources | | Economic growth | |
| Approach | Supports local authorities, highlights zero waste initiatives, and advocates lifecycle thinking | | Advocates long-term policy stability and effective resource use, acknowledges role of consumer behaviour and notes need for continued waste management | | Highlights that resource use should be within environmental limits. Engages citizens, business & industry, and notes legacy wastes | | Advocates implementation of waste hierarchy, recognises waste as a resource, and calls for increased integration and support across sectors and stakeholders | |

(i) *Municipal waste generation per capita per year (ii) Biodegradable municipal waste disposed to landfill as a % of the 1995 baseline.*

*4.2.1 CE aims*

All four documents made reference to economic prosperity combined with some other dimension(s) of sustainable development, variously referring to a 'zero waste economy' and a 'sustainable economy' (England, Scotland and Wales), a 'low carbon economy' and a 'green economy' (Scotland, NI) and a 'prosperous society' characterised by full employment and high value green jobs (Wales). However, the extent to which environmental quality, social equity, and future generations were considered varied significantly.

With respect to environmental issues, all four documents referred to environmental protection, with a strong emphasis on reducing climate change impacts. Regarding environmental targets and ongoing assessment of strategies, Scotland and Wales were the most progressive, going beyond the weight-based indices used within EU policy by adopting more challenging targets measured a carbon footprint based metric (Scotland) and ecological footprinting (Wales). While NI mentioned carbon footprinting, like England it did not introduce any new targets or metrics to measure improvements.

While all four documents referred to safeguarding human health, and Scotland and NI made some reference to social benefits and well-being, the emphasis was less than that placed on environmental protection. Wales was the only exception to this, with directly comparable prominence of environmental and social aspects of the CE, , linking economic and social development with environmental quality, well-being, social justice and equality of opportunity.

All four documents made some reference to shaping the future (through decisions made now) and/or future waste management needs, where Scotland, Wales and NI also made specific reference to future generations. Wales had the strongest consideration of future societal needs (as indicated by the title of the strategy document), where the concept of living within environmental limits explicitly incorporates the time dimension so as to ensure sufficient resources are available to achieve a better quality of life for both present and future generations.

*4.2.2 Core concepts and principles*

All four documents included multiple references to resource efficiency, where the emphasis placed on this concept was comparable across Scotland, Wales and NI, but significantly weaker for England. Scotland and Wales clearly identified the need for large-scale changes to achieve their objectives (including changes to attitudes and behaviours, and acceptance of change), highlighting the role of policy and the public sector in driving this change. In comparison, NI made limited reference to the scale of change (although the need for behavioural change and the role of Government leadership in maintaining the pace of change were touched on), while England made no reference to the scale or type of change needed. Inclusion of other core concepts was variable and limited. Only England and NI made explicit reference to decoupling economic growth from resource use, only Wales recognised limits to growth, and only Wales and NI cited the need for SCP. While Wales and NI made multiple references to the need for life cycle thinking and approaches, England and Scotland made only one reference each. In the case of England this was simply to note that departure from the waste hierarchy could be justified by lifecycle thinking (rather than advocating lifecycle thinking as an underpinning concept to delivering resource efficiency).

*4.2.3 Promotion of the Waste Hierarchy as an operationalisation principle*

Figure 3 presents the occurrence of terms associated with waste hierarchy categories within the waste strategy documents of the UK home nations on both an absolute and per paragraph basis.

While occurrence of the waste hierarchy categories differed widely between the four documents on an absolute basis, frequency counts were more comparable on a per paragraph basis. Overall, the implementation of the full waste hierarchy across all documents is considered to reflect EU waste policy, with some differences in relative emphasis relating to the approach to transposition adopted by England and NI on one hand (“no gold-plating”, reactive) and Scotland and Wales on the other (“gold-plating”, proactive).

Recycling strategies (material recovery, anaerobic digestion, and composting) were dominant within all four documents, where this national policy emphasis on recycling is likely driven by EU policy and targets that focus on recycling and landfill diversion (Mazzanti and Zoboli, 2009; Fisher, 2011).

Prevention strategies were the second most frequently cited in all documents. It is noted that differences in the counts of prevention terms will to some extent reflect the scope of the waste strategies, where England and Scotland both elected to develop separate waste prevention plans and therefore provided only an overview of intended prevention activities within the analysed documents. Nonetheless, inferences can be drawn from the presence or absence of any reference to different prevention imperatives and activities. Furthermore, it is noted that the separate consideration of waste prevention strategies may have unintended consequences arising from a lack of joined up thinking between waste prevention and waste management activities.

The majority of the prevention terms counted made general reference to the need to reduce waste and mirrored the terminology employed by EU policy. While all four documents made some reference to activities associated with R0-R2 (Refuse, Rethink, Reduce), there was a much stronger emphasis on these imperatives in the Welsh document (particularly with respect to product design and the use of recycled materials), and this was also the only strategy to note the role of consumers (in buying less). Likewise, only Wales and Scotland included R3 (Reuse), and only Wales included re-servitising and re-modelling business. .

The least priority was given to “Recover” terms in all documents except Scotland (where it ranked fourth ahead of disposal). However, reference to incineration within the Scottish document was found to be in conjunction with a potential ban on incineration, where the context was to ensure strategies were moved further up the waste hierarchy (not just from disposal to incineration).

The use of continued disposal was found to be a higher priority for the English document (ranked third within this document) when compared with Wales and NI, where it ranked fourth and Scotland where it was given least priority. Interestingly, it is noted that when counts included reference to landfill diversion, the majority of mentions in the Scotland (77%), Wales (56%) and NI (59%) documents were with respect to the latter, whilst in the English document the majority of mentions (66%) were concerned the continued use of landfill.

As noted by Reike et al (2018) it is common to find within CE literature the use of identical terms with different meanings. In this analysis, particularly when considering the waste hierarchy, terms were found to have unclear

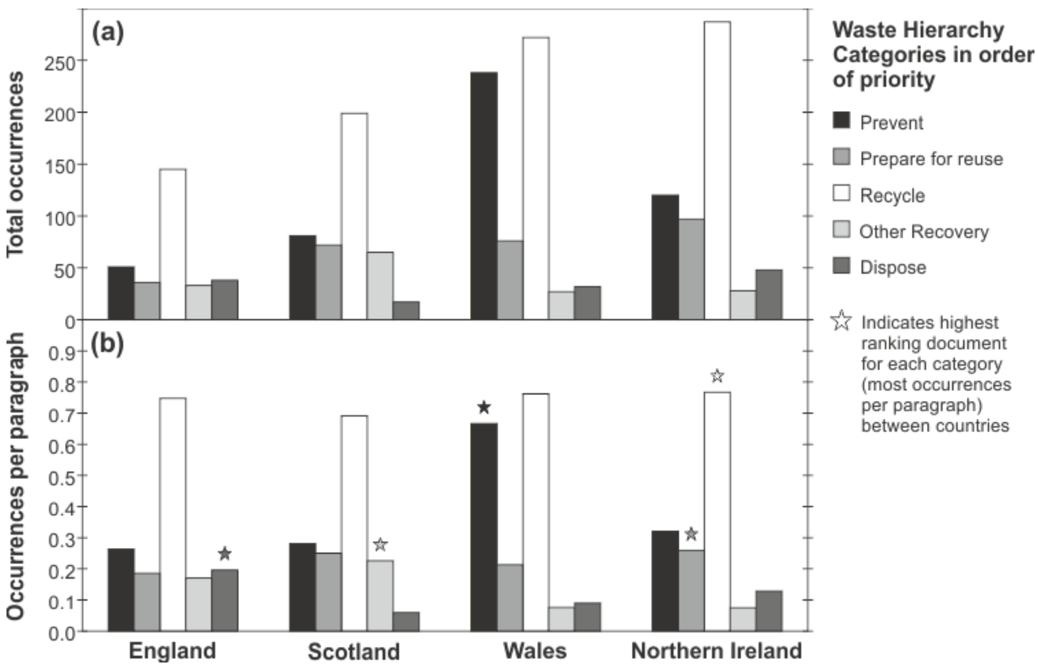


**FIGURE 3:** Representation of waste hierarchy categories in the waste strategy documents of the UK home nations on (a) a total occurrences basis and (b) an occurrences per paragraph basis.

meanings. For example, incineration was often referred to without specifying whether it was "with energy recovery" or "without energy recovery" with the former being classified as a recovery term and the latter a disposal term. Other terms were found to cross the boundaries of R-imperatives, for example reuse could be classified under "Reduce" or "Preparation for reuse". While efforts were made to decipher the correct meaning of terms from their context and / or position within the text, this has been acknowledged as a limitation of the framework.

*4.2.4 Enablers and stakeholder engagement*

Comparison of the four documents found variation in the dominant types of enabling measures and instruments employed to drive market changes. While all four documents made some reference to investment, other fiscal incentives/disincentives, green procurement, extended producer responsibility, and the use of voluntary agreements and standards, the relative emphasis differed. Scotland had a strong emphasis on investment, England dominantly referred to EPR followed by investment, Wales promoted the use of green procurement followed by EPR, while NI focused on voluntary agreements / standards and EPR. Furthermore, Wales, and to a lesser extent Scotland and NI, encouraged the development of markets for recyclates and reuse. With respect to measures that addressed consumer behaviour, England was found to be severely lacking. In comparison, Scotland, Wales and NI all promoted the use of education, communication, and consumer engagement and awareness campaigns to change attitudes. These strategies also incorporated measures that required the involvement of other sectors as well as the waste management industry.

Figure 4 presents the occurrence of terms associated with stakeholder categories within the waste strategy documents of the UK home nations on both an absolute and per paragraph basis, where the responsibilities identified for each stakeholder group with respect to policy instruments and feedback mechanisms are summarised in Tables 3-6 for each of the home nations.

Substantive differences were found between the four documents with respect to the engagement of different stakeholder groups. While, England and NI tended to focus on Macro-level stakeholders, particularly those concerned with cities and regions, Wales and Scotland also placed equal emphasis on micro-level (e.g. consumers, producers, designers) and meso-level stakeholders (e.g. sectors, community groups). In light of the argument made by Su et al (2013), Wales and Scotland would be the most successful in implementation of the CE as they include all three levels of stakeholders.

Notable comparisons include the similar prominence of national stakeholders in all four documents. This is expected given the nature of the documents (i.e. published by the devolved governments and being primarily concerned with domestic strategy). While there was differing prominence, the responsibilities of GD/NGPB and International stakeholders were similar, reflecting the former's role as regulators to ensure compliance and issue sanction where necessary and the latter's role to provides and enforce overarching objectives and targets. With respect to international stakeholders, England and NI were found most likely to engage, this being due to existing waste export routes (England) and the presence of a land border with the Republic of Ireland together with ambitions of an all-island waste strategy in NI. Scotland and Wales also referred to using their influence with national and international stakeholders to shape future goals.

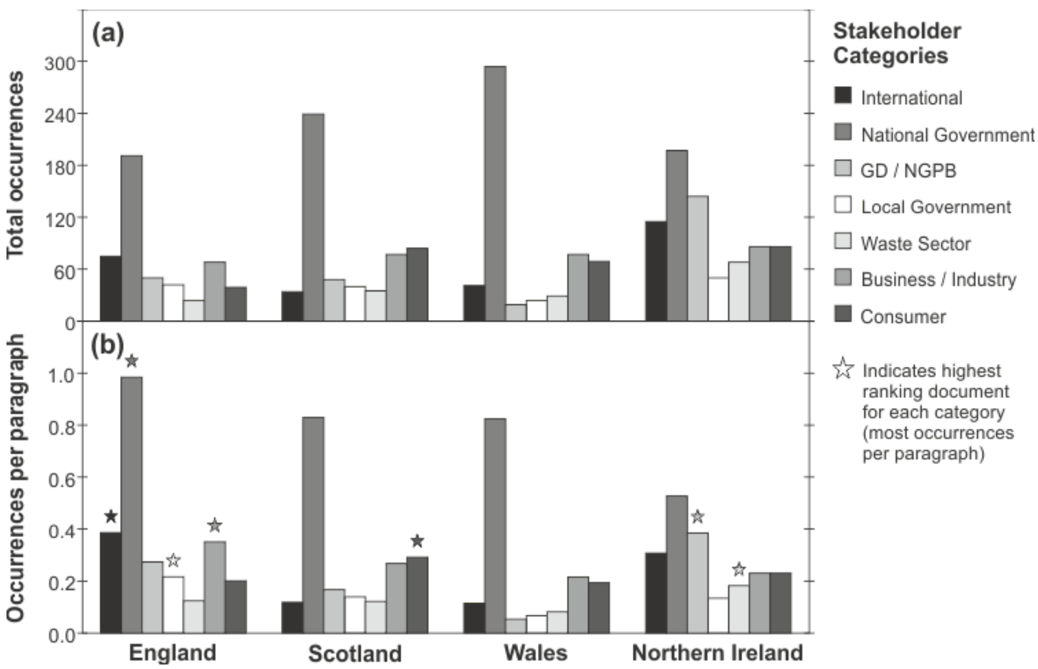


**FIGURE 4:** Representation of stakeholder categories in the waste strategy documents of the UK home nations on (a) a total occurrences basis and (b) an occurrences per paragraph basis.

**TABLE 3:** Stakeholder responsibilities within the Waste Management Plan for England.

| Stakeholder | Responsibilities |
|---|---|
| International | Set overarching legislation and objectives. Introduce broad programmes to assist with meeting objectives. Require the collection of data to assess progress. |
| National | Transpose international legislation into national objectives. Set targets, provide support and guidance. Encourage sustainable thinking within resource and waste management. Produce quality standards for recycled materials. Identify suitable locations for future facilities. Monitor and review progress. Provide information and data to other stakeholders. Drive behaviour change |
| Government Departments & Non-Gov. Public Bodies | Implement international and national legislation and policy. Provide funding for schemes. Organise voluntary sector agreements. Distribute environmental permits. Conduct routine inspections. Provide advice and guidance on the of waste hierarchy strategies and support inter-stakeholder collaboration. Provide data and evidence regarding current and future waste management activities. Initiate and/or respond to consultations. |
| Regional | Obligated to implement national legislation, provide waste collection services, and support businesses in meeting their responsibilities. Work in partnership with the waste sector to ensure full and efficient waste services. Record and report waste data and illegal activity. Provide evidence to consultations. |
| Waste Sector | Adhere to national and international legislation and relevant environmental permit conditions. Where appropriate, develop actions to meet quality standards and change behaviours to contribute to national objectives. Obligated to provide waste collection services that are regular, efficient and affordable, working in partnership with local authorities and other regional stakeholders. Contribute to future waste strategy by providing evidence regarding current activities and responding to consultations. |
| Other Business & Industry | Adhere to national and international legislation, meet sector specific targets, participate in voluntary agreements, and provide private financial initiatives. Supported in recognising and capitalising on resource efficiency opportunities and encouraged to incorporate sustainable thinking into product/service design. Contribute to future waste strategy by providing evidence regarding current activities and responding to consultations. |
| Consumers | Provide evidence on current waste management activities and can respond to consultations. It is acknowledged that consumers are the main contributors to waste generation and that a change in behaviour would contribute to national objectives; however, they are not held responsible or accountable by any policy mechanism. |

**TABLE 4:** Stakeholder responsibilities within Scotland's Zero Waste Plan.

| Stakeholder | Responsibilities |
|---|---|
| International | Set overarching legislation and objectives. Introduce broad programmes to assist with meeting objectives. Promote the waste hierarchy and high-quality recycling. Require the collection of data to assess progress. |
| National | Introduce policies, targets and strategies to address the requirements of international legislation. Develop programmes, promote the waste hierarchy and best available techniques, introduce measures that value resources, and develop secondary materials markets. Provide guidance, tools and support to encourage good practice, and promote long-term stability, eco-design and investment. Stimulate behaviour change by strengthening market confidence, developing measures to influence behaviour, and providing reliable information. Information is collected and reviewed to measure progress with respect to targets and the success of implemented measures and initiatives. |
| Government Departments & Non-Gov. Public Bodies | Enforce regulatory frameworks and provide other regulatory functions to control relevant activities, develop programmes and tools, and provide guidance for the delivery of zero waste plans and policies. Enable efficient resource use. Encourage investment in innovative technologies. Contribute to the design of non-waste facilities / activities. Provide evidence to consultations and macro level studies. |
| Regional | Adhere to regulatory frameworks. Develop programmes and strategic waste infrastructure plans with neighbouring regions. Provide leadership in areas of influence and to achieve value for money with respect to procurement. Provide evidence for consultations, adhere to audits, report data, and contribute to relevant planning applications. |
| Waste Sector | Adheres to regulatory frameworks. Partial responsibility for compliance. Responsibility regarding investment in capacity and infrastructure considering national policy. Develop good practice commitments. Adhere to audits, and report information concerning compositional data, services provided, and voluntary opportunities. Increase workplace skills. Public engagement. |
| Other Business & Industry | Adhere to regulatory frameworks. Responsibility for investment in capacity and infrastructure considering national policy. Subject to sector-specific programmes. Adhere to good practice commitments. Develop innovative technologies. Responsibility for reducing waste generated under their control through resource efficiency opportunities and the incorporation of sustainable thinking into product/service design. Provide evidence to consultations. Participate in awareness campaigns. Improve understanding and usage of resources. |
| Consumers | Active participation in programmes and initiatives. Provide evidence to consultations. Involvement in waste infrastructure planning process. Increase understanding of consumption and waste generation. Recognise and take responsibility for the waste generated. Implored to be enthusiastic and take action. |

Perhaps the starkest difference between the four documents was the inclusion of consumers, or lack thereof, where they held no responsibilities within the English document other than to receive waste management services and potentially participate in initiatives and information collection schemes. This contrasts with the Welsh and Scottish documents that, to varying degrees, hold the consumer responsible for their level of consumption and waste generation, and asks them to actively engage and participate in waste reduction programmes.

With respect to industry and business groups, Scotland and Wales encouraged greater engagement with CE ideals when compared to England and NI. Within the former, industry and business were asked to be innovative, and were encouraged to develop and take opportunities that would incorporate CE thinking into their business models. In contrast, in England and NI engagement with industry and business was limited to providing policy, regulation and voluntary agreements (these were present in all documents) to which business and industry should adhere. Interestingly, NI placed an emphasis on the role of business and the implementation of environmental management systems to

**TABLE 5:** Stakeholder responsibilities within Towards Zero Waste, One Wales: One Planet.

| Stakeholder | Responsibilities |
|---|---|
| International | Set overarching legislation and objectives. Introduce broad programmes to assist with meeting objectives. |
| National | Transpose international legislation and objectives. Provide a long-term vision to reduce Wales' ecological footprint to within environmental limits. Apply key principles (precautionary principle, polluter pays principle, proximity principle, waste hierarchy, and equality of opportunity). Set domestic targets and sector-specific objectives. Introduce penalties for non-compliance. Grant powers to regulators for enforcement. Explore initiatives. Develop sector plans (including voluntary targets). Raise awareness. Provide advice and support regarding secondary materials markets, IPP, and waste infrastructure. Promote broader themes of zero-waste, sustainable development and citizen empowerment. Collect and publish data. Monitor indicators of progress (ecological footprint of waste, provision of recycling services, destination of recyclates, outcomes of eco-design programmes, wellbeing, employment, and skills). |
| Government Departments & Non-Gov. Public Bodies | Ensure and enforce compliance. Develop and implement campaigns. Support local capacity/infrastructure plans and skills development. Provide information on technical requirements. Assess skills gaps. Consult on legislation. Encouraged to adopt sustainable waste management practises and drive change through procurement. |
| Regional | Provide waste collection services and implement engagement campaigns. Support alternatives to landfill and encourage systems that treat waste as a resource to ensure greater consistency in recycled materials. Collect and report data to evaluate progress towards waste prevention goals, best practice, and value for money. |
| Waste Sector | Adhere to legislation. Implement waste strategy. Provide waste collection services. Introduce programmes/initiatives that promote closed loop recycling. Assess infrastructure requirements. Establish integrated networks of waste facilities. Address skills gaps and increase the number of green jobs. |
| Other Business & Industry | Implement waste strategy. Adhere to sector specific plans (and achieve sector-specific targets). Develop and implement voluntary arrangements that consider the polluter pays principle, extended producer responsibility and IPP. Exert influence through procurement activity. Employ eco-design to reduce product impacts (including use of recycled/alternative materials and avoiding the generation of legacy wastes). Contribute to feedback mechanism by recording and submitting data. Assessing skills gaps within their own sector. Share responsibility for waste generated and future proof against future resource competition. |
| Consumers | Encouraged to develop local exchange schemes and participate in national educational and engagement schemes. Workers are encouraged to recognise and rethink their influence within the workplace and at home regarding procurement and consumption. Contribute to the well-being of Wales, resource efficiency and waste reduction. |

**TABLE 6:** Stakeholder responsibilities within the Northern Ireland Delivering Resource Efficiency plan.

| Stakeholder | Responsibilities |
|---|---|
| International | Set overarching legislation and objectives. Introduce broad programmes to assist with meeting objectives. Provide access to officials to support implementation of programmes and objectives. Identify financial and non-financial opportunities. |
| National | Ensure compliance with international policy. Develop (all-island) compatible and complementary policy. Participate in international and UK initiatives. Propose sector-specific targets. Develop domestic re-use and voluntary quality assurance schemes. Reduce burdens on business and support resource efficiency. Collect and publish information on waste flows, commodity prices, and legislative proposals. |
| Government Departments & Non-Gov. Public Bodies | Develop, monitor and enforce waste management strategy and accompanying policies and regulations. Use a suite of penalties and sanctions to ensure compliance. Grant funds for schemes and initiatives. Develop programmes and educational campaigns. Explore and exploit economies of scales. Support market development. Promote collaboration. Provide information. Instrumental in consulting on strategies, legislation and spatial aspects. |
| Regional | Adhere to national and international legislation. Use powers to improve the quality of the environment. Responsible for planning aspects of waste management strategies. Work in partnership with regulators, other regional stakeholders and the third sector to tackle poor compliance, develop schemes and initiatives, and provide advice. Collect and report data. Provide evidence to consultations and participate in studies, campaigns and inspections. |
| Waste Sector | Adhere to national and international legislation and permit/ licence conditions. Deliver domestic targets and actions. Develop and utilise programmes and investment schemes to introduce innovative waste collection schemes and integrate facilities on an all-island basis. Implement codes of practice. Support local authorities and communities to adhere to the waste hierarchy. Contribute to consultations. Collect and report data regarding specific waste streams. |
| Other Business & Industry | Adhere to national and international legislation, and sector specific domestic targets. Develop and participate in voluntary initiatives. Build market confidence. Consider best available techniques. |
| Consumers | Participate in campaigns. Promote social enterprise along with green jobs. Instigate improvement through public engagement and social acceptance. |

improve environmental performance, where this consideration did not feature in the other strategies.

## 4.3 Implementation of EU policy and future implications

Analysis of these four documents illustrates point made by Garcia Quesada (2014) that the amount of discretion given to member states to implement EU objectives can lead to significant differences (and success) in national implementation. Where England has transposed EU policy with "no gold-plating" (minimum requirements), Wales in particular can be argued to have had more success in using the "gold-plating" (going beyond minimum requirements) approach (Anker et al.„ 2015). Indeed, it is noted that the English document incorporates and combines existing policy into one document without introducing new approaches. This is fundamentally different to Scotland, Wales and NI who all aim to set a strategic direction. Having said that, while NI does set a strategic direction, like England, its emphasis remains on meeting the requirements set out by the EU. In comparison, Scotland and Wales appear much more proactive, extending their strategies beyond EU requirements, influencing policy not in their direct control to achieve their individual goals, and understanding the need for, and instigating, change. This observation agrees with Winans et al (2017) and Scotford and Robinson (2013), regarding the superiority of Welsh and Scottish environmental policy within the UK, in that the strategies they promote are more progressive, but like England and NI they continue to refer to overarching objectives set by the EU.

Differences in approach may have contributed to differing levels of success with respect to EU targets. This disagrees with Andrews and Martin (2010) who found no variation in waste management services between the four devolved administrations, attributing this to objectives being set at a supranational level (i.e. by the EU). Conversely, these findings agree with Falmer et al (2013) who noted marked differences in the management strategies employed by the four devolved nations, connecting this to a lack of clarity and direction within overarching waste policy. This analysis found that in the period since strategy publication (2010 for Scotland and Wales; 2013 for England and NI), both Scotland and Wales have implemented strategy that has reduced waste generation, increased recycling rates and reduced landfilling of BMW, with Wales achieving a landfill rate reduction of over 50%. In comparison, waste generation in England and NI has increased and varying results are reported for recycling and landfilling. In England, while the landfill rate has been reduced, the rate of recycling has plateaued, remaining at 44.2%. Whereas in NI, both recycling and landfill rates have increased. With respect to EU targets, all four nations have achieved the landfill directive of no more than 55% BWM landfilled by 2016, and Wales has already surpassed the recycling rate target set by WFD of at least 50% by 2020. While it could be suggested that Scotland and NI are progressing towards meeting this target, the plateauing of England recycling rate could suggest its current strategy may struggle.

Overall, limitations for all of the strategies are a continued focus on waste management rather than resource utilisation, and the reliance on EU targets and objectives to set national priorities. This issue may become more pertinent after Brexit due to an absence of overarching UK strategy, which would have previously been supplied by the EU. While it appears that Wales and Scotland do have long-term policy objectives (including to future proof and avoid 'lock in') and have started the process of incorporating waste management strategy into the broader context of resource management and sustainable development, this is generally absent from the English (and therefore overall UK) strategy. This lack of coherence in objectives and enforcement across the four devolved nations may lead to further complications in the future. As suggested by Scotford and Robinson (2013), diverging amendments enacted by devolved administrations may lead to increased fragmentation and disparity of UK environmental policy.

## 5. CONCLUSIONS

An alternative to the linear economy model, the CE has been advocated internationally as a solution to current unsustainable consumption patterns. It aims to reduce consumption, recirculate products and materials, and prevent environmental degradation. In response, the EU has developed the forthcoming CEP to provide more stringent objectives and targets, reiterate the waste hierarchy, promote industrial symbiosis and elevate the role of resource efficiency. As with previous EU strategies, member states will be required to transpose the CEP into national strategy and achieve its targets and objectives. While the CEP does provide the correct direction for member states to initiate a transition toward the CE, it has also been criticised for its continued focus on waste management with too little emphasis on high priority waste hierarchy categories such as reduce and reuse.

This study developed a framework based on CE-related literature in which an overall CE definition was identified, along with the importance of R-imperatives (in particular the waste hierarchy) and stakeholder engagement. The framework was used to assess the current waste strategies of the four devolved UK nations (England, Scotland, Wales and NI). Differences in interpretation and implementation of current EU objectives were identified across the devolved nations, with Wales and Scotland promoting more progressive strategies and showing greater improvement regarding EU waste targets. This confirms the conclusion of previous studies that Wales and Scotland currently have the most progressive waste management strategy of the four devolved nations.

The future of waste management strategy in the UK, will be shaped by the CEP and potential ramifications of Brexit. In the short to medium term, adoption of the CEP will provide overarching objectives and targets for the UK due to transposition into national policy. Long term objectives will depend on changes implemented by the UK government. In addition, enforcement that has previously been supplied by the EU to ensure objectives and targets are met may not be present unless a UK wide enforcement system is adopted. This may become an area of contention if Scotland and Wales, who already promote progressive waste strategies, were to diverge further. To address this issue, it is imperative that strong cross-party support is gained for long-term CE objectives both within each devolved parliament and across the UK. This would prevent the return of waste strategy politicisation that was successfully overcome on joining the EU due to the primacy of European law.


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# RECOVERY OF BY-PRODUCTS FROM THE OLIVE OIL PRODUCTION AND THE VEGETABLE OIL REFINING FOR BIODIESEL PRODUCTION

Mariana Cruz [1], Emanuel Costa [1], Manuel Fonseca Almeida [1], Maria da Conceição Alvim-Ferraz [2] and Joana Maia Dias [1,*]

[1] *LEPABE, Departamento de Engenharia Metalúrgica e de Materiais, Faculdade de Engenharia, Universidade do Porto, R. Dr. Roberto Frias, 4200-465 Porto, Portugal*
[2] *LEPABE, Departamento de Engenharia Química, Faculdade de Engenharia, Universidade do Porto, R. Dr. Roberto Frias, 4200-465 Porto, Portugal*

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**ABSTRACT**

The by-products acid oil from soapstock of vegetable oil refining and olive pomace oil were evaluated for biodiesel production. Enzymatic hydroesterification was studied to convert the acid oil (~34 wt.% free fatty acids) into methyl esters; due to the low free fatty acid content of the fresh olive pomace oil (~2 wt.%), alkaline transesterification was conducted. The results from the enzymatic hydrolysis (35˚C, 24 h, 200 rpm) showed a clear influence of enzyme concentration (0.1 – 5 wt.%, relative to oil) and water:oil ratio (1:0.25 and 1:0.5 w:w) towards free fatty acid production. After applying the best established conditions (3 wt.% of enzyme and 1:0.5 water: oil ratio, w:w), enzymatic esterification was performed (35˚C, 7 h, 200 rpm, 2 wt.% of enzyme and 2:1 molar ratio of methanol to acid). Hydroesterification led to a product with a methyl esters content of about 84 wt.% whereas the esterification alone allowed reaching only around 65 wt.%. The olive pomace oil was obtained from chemical extraction of fresh olive pomace (~18 wt.% of oil). By performing direct alkaline transesterification (65˚C, 1 wt.% NaOH, 1 h and 6:1 molar ratio of methanol to oil) a product with a purity of 90 wt.% was obtained. The olive pomace storage in the air during 2 weeks led to an increase in the oil free fatty acid content of almost 2 fold showing the relevance of developing storage and conservation strategies to ensure a sustainable recovery of this by-product. Both by-products showed potential for biodiesel production.

## 1. INTRODUCTION

Biodiesel is considered a very promising biofuel for fossil fuels replacement and might have an important role to reduce the global energy demand due to its environment-friendly and renewable properties (Mahmudul et al., 2017).

There are several methods for biodiesel production such as blending, microemulsification, pyrolysis and transesterification, using edible and non edible oils as well as waste raw materials. Alkaline transesterification is one of the most common and attractive processes to produce biodiesel, being widely accepted (Baskar and Aiswarya, 2016; Verdugo et al., 2011). Despite the difficulties during purification, associated with loss of product yield and management of wastewaters (Machado et al., 2016), it is still the most employed route for economic reasons. However, the homogeneous alkaline catalysed process cannot be applied to a material with a high free fatty acid (FFA) content due to soap formation; the conventional reported limit is 1 wt.% FFA content, corresponding roughly to 2 mg KOH $g^{-1}$ in terms of acid value (Dias et al., 2009). The pretreatment of raw materials with the objective of decrease the FFA content can be achieved by performing different types of reactions, namely homogeneous acid esterification, enzymatic hydroesterification and glycerolysis, which recover the acids present (Živković et al., 2017).

The hydroesterification represents a new alternative for the production of esters, since it allows the use of raw materials with high water and FFA contents. The FFA produced by hydrolysis can be further esterified by a short chain alcohol, producing esters and water. Firstly, glycerides are hydrolysed to FFA and then esterified into esters using methanol or ethanol and different catalysts (Zenevicz et al., 2016). Recently, the use of enzymes as an effective biocatalyst has been an emerging contribution for

* Corresponding author:
Joana Maia Dias
email: jmdias@fe.up.pt

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biodiesel production, since they have tolerance to FFA and water, require mild temperatures for the reaction and show versatility to catalyze hydrolysis, esterification and transesterification, thus enabling hydroesterification (Avhad and Marchetti, 2015).

Most of the biodiesel production costs (around 80%) are related to the raw material, so exploring low-cost raw materials for this process is still of high relevance (Knothe and Razon, 2017). Non-edible oils can be seen as the future sources of biodiesel, since they do not compete with food supply; however, they have as disadvantage the need of soil support for its growth.

By-products such as soapstocks are seen as potential low cost feedstocks for biodiesel production (Piloto-Rodríguez et al., 2014). The soapstock, which results from the neutralization of the raw vegetable oils, is normally acidified with a strong mineral acid, which allows the release of FFA. The process generates a fraction which is generally dark in colour, known as acid oil, that usually contains water (0.8-3.1%), FFA (39-79%), acylglycerols (18-30%) and unsaponifiable matter (0.4-4.2%) (Echim et al., 2009). Soapstock is generated at a rate of about 6 %Vol. of refined oil (Park et al., 2008). Another by-product from vegetable oil refining, the fatty acid distillate or deodorizer distillate, is obtained in the final deodorization stage and can also contain high amounts of FFA (Piloto-Rodríguez et al., 2014).

In the Mediterranean region there is a large production of olive oil for human consumption and a by-product of this industry is formed, the olive-pomace, which might still present a relevant oil content (on average 5-8%, wet basis) (Göğüş and Maskan, 2006). In Portugal, the olive production forecast for 2016 was around 476 003 t (INE, 2018). For the traditional olive press process, Azbar et al. (2004) indicates that in the olive oil production, 1 t of olives leads to around 400 kg of solid waste (olive pomace), 200 kg of olive oil and 600 kg of wastewater containing residual solids and oil. It is therefore advisable to develop studies regarding the recovery of such material, in particular, for biofuels production.

The present work evaluated the use of two by-products for biodiesel production (acid oil from soapstock from vegetable oil refining and olive pomace oil). Taking into account the FFA content associated with each raw material, for biodiesel production the enzymatic hydroesterification was evaluated from the acid oil from soapstock and the alkaline transesterification was performed directly to the olive pomace oil and to the best product of the hydroesterification aiming maximum biodiesel purity.

## 2. MATERIALS AND METHODS

### 2.1 Materials

The acid oil from soapstock of vegetable oil refining (mixture of sunflower and soybean seeds) was provided by the company Nature Light, S.A. Olive pomace was supplied from an olive oil company of Northern Portugal (DouroSol Company). Petroleum ether (LabChem ≥ 90%) was used for chemical oil extraction (soxhlet).

Methanol (Fischer Scientific ≥ 99%) was used as the acyl acceptor. The catalysts used were sodium hydroxide powder 97% (reagent grade, Aldrich) and the lipase from *Thermomyces lanuginosus* (Lipolase 100L, activity ≥ 100,000 U/g) purchased from Sigma-Aldrich. All the other reagents were of analytical grade.

### 2.2 Extraction of olive pomace oil

The oil extraction was performed using a Soxhlet extractor (1 L). Firstly, the raw material was crushed in a mortar and placed inside the thimble. Then, the thimble was dipped in petroleum ether solvent for 6 h (equivalent to 14 turns of the solvent in the extractor). After the extraction, the solvent was removed in a rotary evaporator at close to 70°C. Several extraction cycles were conducted until reaching the necessary amount of oil for the study. The procedure performed was based in NP EN ISO 659 (2002).

In order to evaluate the effect of the storage in the FFA content of the oil, the extraction was carried out with the fresh olive pomace and after two weeks of storage at the air (room temperature).

### 2.3 Analytical procedures

The physicochemical properties determined in the acid oil from soapstock were the FFA content and the water content. The FFA content was determined according to NP EN ISO 660 (tritimetric method) and the results are expressed as the weight percentage in terms of oleic acid equivalents (molar mass of 282 g $mol^{-1}$).

Taking into account the expected values for water content, it was determined by weight loss at T = 105°C ± 2°C (oven method), until constant weight, according to EN 12880 (2000) for both by-products; results are expressed as weight percentage, in wet basis.

For the olive pomace, the oil content was determined according to NP EN ISO 659. Oxidation stability and FFA content were determined in the extracted olive pomace oil. The oxidation stability was measured in agreement with the EN 14112 by accelerated oxidation using a Rancimat equipment (Metrohm).

The methyl ester content of the final products was determined according to EN 14103 (2003) by gas chromatography, using a Dani Master GC with a DN-WAX capillary column of 30 m, 0.25 mm internal diameter and 0.25 μm of film thickness. The temperature program used was as follows: 120°C was initially selected as the starting temperature, followed by a temperature rise at 4°C per minute, up to 220°C, holding time of 10 minutes.

### 2.4 Recovery of acid oil from soapstock

#### *2.4.1 Enzymatic hydroesterification*

The hydroesterification process occurs in two-steps. Firstly, the glycerides are hydrolysed to FFA and glycerol and after, the FFA are esterified using an alcohol to produce biodiesel (esters) and water.The hydrolysis reactions were carried out in 100 mL Erlenmeyer flasks in an orbital shaking incubator (Agitorb 200IC), with constant stirring of 200 rpm, during 24 h at 35°C according to the literature (Aguieiras et al., 2014; Cavalcanti-Oliveira et al., 2011; Kabbashi et al., 2015; Watanabe et al., 2007).

For the hydrolysis reaction, 25.0 g of the acid oil were used and the defined amount of water was added (mass ratio of oil:water 1:0.25 or 1:0.5). After reaching the reaction temperature, the established amount of enzyme (0.2, 1, 3 and 5 wt.%) was also added to the reactional mixture.

The hydrolysis reaction generates two phases: oil and water/glycerol that are essentially immiscible at mild temperatures, and can be separated by centrifugation (Machado et al., 2016; Vescovi et al., 2016). Thus, at the end of the reaction, the final product was centrifuged (Hermile Z200A) at 3500 rpm during 12 minutes and two distinct phases were formed and separated. The final FFA content of the oily phase was determined.

Esterification reactions were carried out under batch conditions using 100 mL Erlenmeyer flasks in an orbital shaking incubator (Agitorb 200IC). The present study was performed using the best conditions (35°C, 200 r.p.m, molar ratio of acid:methanol 1:2 and 2 wt.% of enzyme) by adjusting, in preliminary studies, those obtained from a previous study (Cruz et al., 2017).

The enzymatic esterification was carried with the best result in terms of FFA production obtained by the hydrolysis reaction, monitored for 7 h, based on the literature (Cruz et al., 2017) and compared with the enzymatic esterification of untreated oil, under the same conditions. Samples of around 0.4 mL were collected at different time intervals to measure the FFA content. All experiments were performed in duplicate and the variation was evaluated in terms of the relative percentage difference to the mean (RPD). In all cases, RPD was less than 10%.

### 2.5 Transesterification

Synthesis of the final biodiesel product by alkaline transesterification was conducted with the product from the hydroesterification obtained under the best established conditions. In the case of olive pomace oil, the transesterification was performed directly.

The alkaline transesterification was performed in a batch reactor at 65°C during 1 h; the amount of catalyst was 1 wt.% NaOH and the methanol:oil molar ratio was 6:1; purification was conducted by acid/water washing and drying procedures in agreement to previous studies (Dias et al., 2009).

## 3. RESULTS AND DISCUSSION

### 3.1 Characterization of by-products

*3.1.1 Acid oil from soapstock*

In spite of raw material being from the same source than that previously studied by Cruz et al. (2017), the FFA content was 34 ± 1 wt.%, possibly due to sampling in a different season of the year. This FFA content is quite low in comparison with values from the literature. Irandoust et al. (2012) reported 67.4% of FFA content for an acid oil from soapstock of soybean oil refining, whereas Chiplunkar and Pratap (2016) studied an acid oil from soapstock of sunflower oil refining with 65% of FFA content.

The oil water content was 4.8 ± 0.1 wt.%, in the range reported by Cruz et al. (2017). Pérez-Bonilla et al. (2011) reported a water content slightly lower (2.2 wt.%) for an acid oil from soapstock of vegetable oil refining.

*3.1.2 Olive pomace oil*

Olive pomace oil is reported in the literature as having a high content of oil and water (Barbanera et al., 2016; Missaoui et al., 2017). In the present work, the following results were obtained: 18 ± 2 wt.% lipid content; moisture content of 26.8 ± 0.2 wt.%

The result of the lipid content shows the recovery potential of this by-product aiming biodiesel production. In terms of management and application, the high moisture content of olive pomace can affect the costs of transport and promote its degradation.

The FFA content determined in the olive pomace oil was 2.2 wt.%. The value obtained was much lower than that reported by Che et al. (2012) which was around 22 wt.%; Rajaeifar et al. (2016) studied an olive pomace oil with 5.75 wt.% of FFA content. The low acid value obtained is attributed to the fact that it results from fresh olive pomace which was collected right after the mechanical extraction of olive oil.

Taking into account alkaline transesterification, the FFA content exceeds the reported limit of 1 wt.% (Dias et al., 2009). For this reason, pretreatments to decrease the FFA content might be required prior to the transesterification reaction. However, because of the low yields previously reported for two steps processes and the low FFA content obtained, the alkaline transesterification was still evaluated as a single process (Dias et al., 2009).

In order to evaluate the impact of storage, olive pomace oil was extracted after two weeks in the air at room temperature; the extracted oil presented around 4 wt.% of FFA content, corresponding to an increase in almost 2 fold of this parameter. It is therefore expected that the FFA content continues to increase with the time of the storage at the facilities where the olive oil is produced, until the end of the campaign. Such values will clearly impair the use of the conventional homogeneous alkaline transesterification process and therefore such conditions of storage should be revised aiming at recovering a high quality product from the olive pomace. Go et al. (2016) reported that FFA contents in agricultural biomass and residues increased significantly with increasing storage time. Treatments such as the storage below 0 °C or stabilization through drying or heating to remove moisture and deactivate the enzymes, are suggested to avoid FFA formation.

The oxidation stability of the oil was 12.7 ± 0.5 h, very high when compared with other raw materials used for biodiesel production like canola oil or soybean oil (Atabani et al., 2013) and should be related with its fatty acid profile and content in natural antioxidants (phenolic compounds) (Maurizio Servili and Montedoro, 2002). In fact, the low content in polyunsaturated fatty acids (14.4% wt.%, includes C18:2 and others presented in Table 1) is possibly one of the reasons for the high oxidation stability found for oil. The unsaponifiable matter might also be an indicator of the stability of this oil since bioactive compounds such as natural antioxidants found in this raw material are present in this fraction. In the present work it was not possible to perform this quantification; however, results of revised studies report that unsaponifiable matter represents around 2%

**TABLE 1:** Fatty Acid Methyl Ester profile of biodiesel produced from olive pomace oil.

| Methyl ester of the following fatty acids | wt.% |
|---|---|
| C16:0 | 9.90 |
| C18:0 | 3.20 |
| C18:1 | 71.3 |
| C18:2 | 12.7 |
| Others (< 1 wt.% each) | 2.90 |
| Mean molecular weight (g $mol^{-1}$)* | 880 |

*Calculated taking into account the profile*

of the total weight (Bulotta et al., 2014; Chanioti and Tzia, 2017; Orozco et al., 2011).

## 3.2 Enzymatic hydroesterification

### *3.2.1 Enzymatic hydrolysis*

As mentioned previously, enzymes are promising catalysts for biodiesel production by hydroesterification, since they work very well under mild conditions of temperature and pressure, and particularly, because lipases exhibit high activity for both hydrolysis and esterification reactions (Sousa et al., 2010).

The hydrolysis reaction was monitored by acid value determination during 24 h (Figure 1) and the effect of water:oil mass ratio (1:0.25 and 1:0.5 w:w) and enzyme concentration (0.2-5 wt.%, relative to oil) were studied. The evolution with reaction time shows generally a gradual increase in FFA content, achieving a final FFA content of almost double of the initial in almost all conditions after 24 h (1440 min). It should be highlighted that the FFA content in the Figure 1 relates to the reaction mixture (there is a dilution effect) whereas the final FFA content was measured in the recovered oil after phase separation according to the methodology presented at the 2.3.1 section.

It is known that the water content affects the equilibrium conversion of the reactions at the esterification/hydrolysis and it can also affect the products distribution in the reaction medium (Foresti et al., 2007). After 24 h, using 1:0.25 oil:water mass ratio, the final FFA content of the oil ranged from 58.1 wt.% to 78.3 wt.%, from the lowest to highest catalyst concentration of 0.2 wt.% and 5 wt.% respectively. Using 1:0.5 oil:water mass ratio, the final FFA content of the oil ranged 47.9 wt.% to 84.8 wt.% from 0.2 wt.% to 5 wt.%, respectively.

At the lowest catalyst concentration (0.2 wt.%) and water:oil mass ratio (1:0.25), the results are clearly different, with lower conversions, than at the other concentrations studied (1-5 wt.%), as expected. The results for the 1:0.5 oil:water mass ratio using 0.2 wt.% of catalyst seem interesting, although an increase in the reaction time more than 1440 minutes might be unfeasible towards industrial scale application.

The conversion increased with the increase of catalyst concentration, with the best results being obtained using higher water concentration. The highest FFA content was obtained using 5 wt.% of enzyme and the highest oil:water ratio, which afforded a final FFA content of around 1.5 times the initial. Increasing the catalyst dose further resulted in a decrease in the FFA produced, though insignificantly (data not shown), a drawback which can be attributed to inability of the excess catalyst to reach the aqueous phase of the substrate (Kabbashi et al., 2015).

Aguieiras et al. (2014), by studying the hydrolysis of an acid oil (10.5 wt.% FFA content) from macauba (*Acrocomia aculeata*), achieved 99.6% of FFA after 6 h at 30°C. The reaction medium was composed of macauba acid oil and buffer (50% v/ v), using a 0.1 mol $L^{-1}$ sodium acetate buffer of pH 4.0 and 2.5% (w/v) of biocatalyst; Rodrigues and Ayub (2011) obtained a conversion of the FFA of about 95% after 10 h of enzymatic hydrolysis of refined soybean oil (25.0 g, 30°C, 200 r.p.m., 3:1 water:soybean oil molar ratio; 25 wt.% enzyme (mixture of 65% *Thermomyces lanuginosus* and 35% *Rhizomucor miehei*); Vescovi et al., (2016)

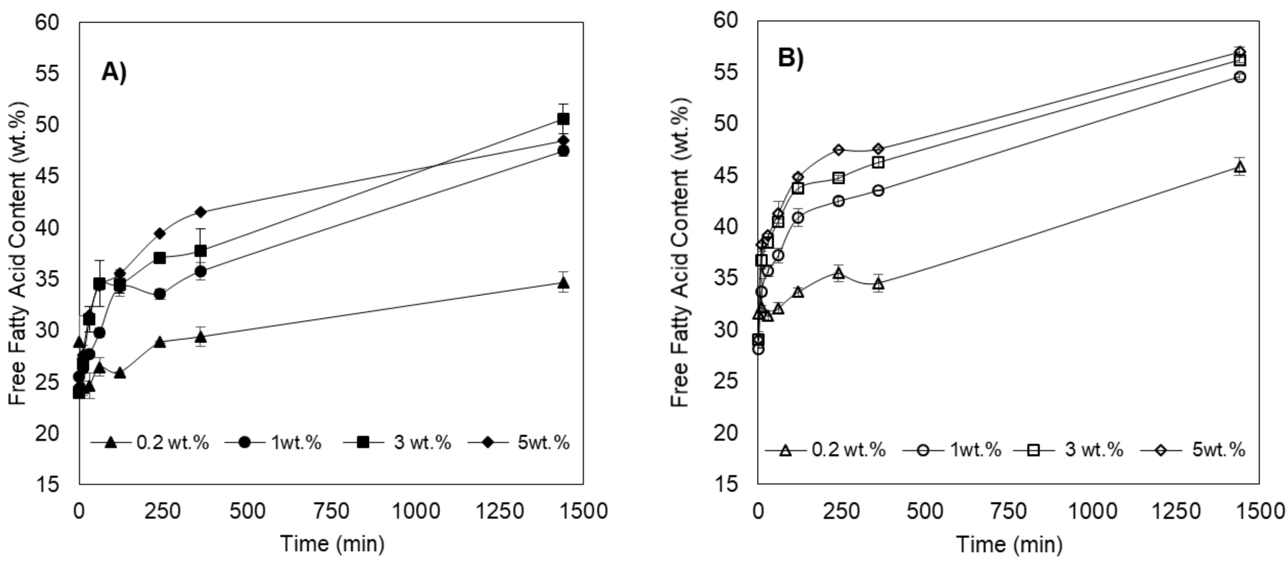


**FIGURE 1:** Release of fatty acids during hydrolysis of acid oil from soapstock using different concentrations of enzyme and two oil:water mass ratios. A) 1:0.25 and B) 1:0.5 (25 g of oil, 35°C, 200 rpm).

achieved about 90% of conversion after 9 h of reaction using waste cooking oil (WCO) (~1.4 mg KOH $g^{-1}$), reaching around 100% after 24 h and the hydrolysis reaction was carried out at 30°C in a salt-free aqueous medium (WCO/water ratio of 1:4, v/v) using lipase from *Thermomyces lanuginosus* (enzyme/WCO ratio of 1:5.6, w/w). The differences to the present study should be related with the different characteristics of the oils used and the variations of the conditions employed.

### 3.2.2 Enzymatic esterification

The esterification of hydrolysed acid oil was performed and compared with the esterification of acid oil without any pretreatment (Figure 2). Without the hydrolysis pretreatment the FFA content reduction was 53.4% and after hydroesterification the acid oil FFA content was reduced in 87.8%.

Vescovi et al., (2016) using an hydrolysed waste cooking oil (acid value of 197.92 mg KOH $g^{-1}$) achieved around 90% of FFA content reduction at 6 h of enzymatic esterification, and there was not significant increase by increasing time up to 12 h of reaction (reaction conditions: 15 g, 40°C; 10 wt.% enzyme, acid:ethanol molar ratio of 1:7).

The methyl ester content was determined for the final products and the GC analysis showed about 65 wt.% ± 1 and 84 wt.% ± 1 of esters at the esterification and hydroesterification product, respectively. Since in the first case the initial FFA content was 34 wt.% and in the second case was 81% (after hydrolysis), it was verified that not only FFA esterification but also the transesterification of mono-, di- and/or triglycerides present occurred (Cruz et al., 2017). This, because taking into account the reaction stoichiometry, the mass of esters should not differ greatly from the mass of the acids. However, it would be extremely important for future work to analyse the mono-, di- and triglycerides profile of the acid oil and throughout the reactions (not possible under the present work), because it would lead to the clear understanding and consequently an accurate quantification of their role.

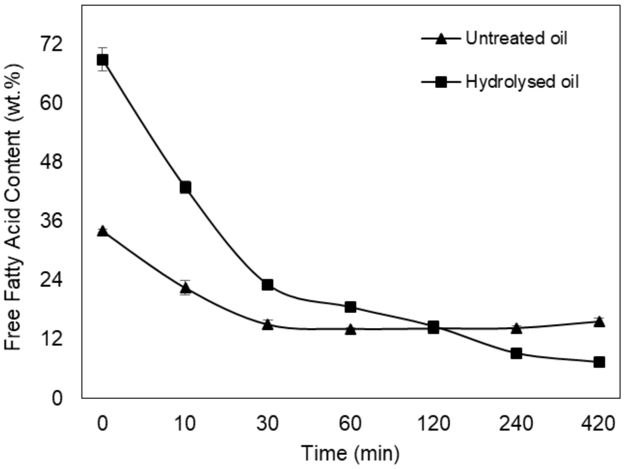


**FIGURE 2:** Reduction of free fatty acid content during enzyme catalysed esterification of untreated and hydrolysed acid oil (25 g of oil, 35°C, 200 rpm, molar ratio of FFA:methanol 1:2, 2 wt.% enzyme).

The homogeneous alkaline transesterification was performed on the best pretreated product aiming to obtain a high purity (convert the remaining glycerides into methyl esters), in agreement with EN 14214 (> 96.5 wt.% of methyl esters); however, biodiesel with around 90 wt.% of methyl ester content was obtained. Thus, it would not be advantageous its application (transesterification) under the conditions studied. The analysis of the methyl ester product showed predominance of C18:2 and C18:1 methyl esters (51 and 27 wt.%, respectively), similar to that reported Haas (2005) for an acid oil from soapstock of soybean oil (55 wt.% and 17 wt.% for linoleic and oleic acid, respectively).

### 3.2.3 Alkaline transesterification of olive pomace oil

Biodiesel production was performed using olive pomace oil without any pretreatment. The production of biodiesel directly from olive pomace oil was possible and led to a biodiesel with a purity close to 90%, slightly lower than that required according to EN14214. López et al. (2014) obtained a biodiesel with 98 wt.% methyl esters from direct transesterification of olive pomace oil (40 min, 60˚C, 6:1 molar ratio of methanol:oil, 1.2 wt.% of KOH). The oil presented however a very low FFA content (0.3 wt.%).

The fatty acid profile for this oil is in agreement with that reported by Che et al. (2012) and Costa et al. (2013) for such oil type and shows the predominance of the oleic acid.

Taking into account the verified increase of the FFA content resulting from storage, the conventional alkaline transesterification should be inviable for such by-product. To avoid increased production costs, alternative storage conditions of olive pomace should be evaluated to ensure its higher quality.

## 4. CONCLUSIONS

The enzymatic hydroesterification of acid oil from soapstock presenting an FFA content of 34 wt.% led to a product with a final methyl esters content of about 84%, a value higher than 65%, obtained from the esterification alone. In agreement, such process seems to be promising towards the biocatalysed production of methyl esters from low cost raw materials.

The olive pomace oil obtained from fresh olive pomace showed recovery potential as raw material for biodiesel production. The original FFA content of about 2 wt.% verified for this oil was low, but still higher than the threshold value of 1 wt.% generally agreeded for alkaline transesterification. In spite of that, the biodiesel produced directly from such oil without pretreatement led to a product with a methyl ester content of 90 wt.%. Due to the fast increase of the oil FFA content resulting from the storage of the olive pomace during olive oil production campaigns, storage conditions should be revised aiming to have a by-product with higher quality.

The studied by-products show potential as sustainable raw materials for biodiesel production.

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## ABBREVIATIONS

FFA: Free Fatty Acid(s)
RPD: Relative percentage difference to the mean
WCO: Waste cooking oil

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# TREATMENT OF SMUGGLED CIGARETTE TOBACCO AND FOOD SOLID WASTE IN A 2000 L FACULTATIVE REACTOR

Karine Marcondes da Cunha [1,2,3,*], Rosimara Zittel [1], Cleber Pinto da Silva [1], Gislaine Vieira Damiani [2], Thainá Aparecida da Silva de Souza [2], João Vitor Gregório dos Santos [2] and Sandro Xavier de Campos [1]

[1] Research Group on Environmental and Sanitary Analytical Chemistry (QAAS), Ponta Grossa State University (UEPG), PO Box: 992, Av. General Carlos Cavalcanti, 4748 Ponta Grossa, PR 84030-900, Brazil
[2] Federal Institute of Paraná, Biotechnology Sciences Department, CEP 82530-230 Jaguariaíva, PR, Brazil
[3] Federal Institute of Santa Catarina, Food Sciences Department, Av. dos Expedicionários, 2150, Canoinhas, SC, 89460-000, Brazil

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**ABSTRACT**

Organic waste is among the solid waste produced worldwide. When such residue is improperly accumulated and disposed of, it generates environmental impacts, polluting soils, rivers and the air. The objective of the present study was to investigate the use of a 2000 liter capacity reactor for composting domestic organic waste, wood chips and smuggled cigarette tobacco. Physicochemical analyzes (pH, temperature, humidity and C/N ratio), biological (germination test and pathogens) and spectroscopic (UV-Vis and FTIR) analyzes were performed to monitor the process. In addition, the influence of two different C/N ratios was investigated. C/N ratio analyses and phytotoxicity tests showed that the compound reached maturity over a period of 120 days. Spectroscopic analyzes of UV-Vis and FTIR showed efficiency, indicating the degradation of compounds of simpler structure and the formation of humified compounds. The microbiological and heavy metal analyses of the final compounds revealed that the results are in accordance with the legislation. The different C/N ratios showed very close results, not affecting the composting process. Thus, the proposed treatment of domestic organic waste and smuggled cigarette tobacco in facultative reactors of 2000 liters was seen to be efficient and produced mature compound in the different C/N ratios studied.

## 1. INTRODUCTION

The accumulation of solid residues due to the growth of urban populations has become evident. Globally, 46% of the volume of solid organic solid waste generated is sent to landfills, and this increases to 51% in Brazil, according to the Institute of Applied Economic Research at Brazil IPEA (2012). The disposal of this kind of waste in landfills or dumps entails the waste of nutrients and organic matter that in the natural cycle have the role of fertilizing and maintaining soil life (Abreu, 2017).

In recent years a residue that has been gaining attention in Brazil is the residue from the seizures of smuggled cigarettes. Data from the Brazilian Internal Revenue Service report that approximately 70% of smuggled goods entering in Brazil are cigarettes (Pegoraro, 2016). Currently, cigarettes seized by the IRS are destroyed by incineration. Such destination uses a large amount of energy and still releases polluting gases such as carbon dioxide, nitrogen oxides, dioxins, among others (Yang et al., 2016). Domestic organic waste (DOW) and smuggled cigarette tobacco (SCT) have the advantage that they can be recycled through composting (Abreu, 2017).

Windrow composting is the most common form of composting, and at all scales it has been extensively developed and evaluated, with numerous reports in the literature (Campos et al., 2014). Recent research has demonstrated success in the cigarette tobacco composting process in a reactor, which is an efficient technology to reduce tobacco toxicity and produce matured compost (Kopcic et al., 2014; Zittel et al., 2018).

Different models of reactors were developed and adapted for the treatment of different organic residues as vertical and horizontal, aerobic, anaerobic and facultative (Karnchanawong; Suriyanon, 2011; Kopcic et al., 2014; Jeonga et al., 2017; Zittel et al., 2018). Reactors are systems that usually protect the composting process from environmental factors such as humidity and temperature. Research has evidenced that these biodegradation pro-

* Corresponding author:
Karine Marcondes da Cunha
email: karine.marcondes@ifsc.edu.br


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cesses are efficient, using reactors systems in pilot and domestic scales, with forced and natural aeration system (Karnchanawong; Suriyanon, 2011; Jeonga et al., 2017; Zittel et al., 2018).

The facultative reactor is considered a promising technology when compared to conventional techniques. Facultative reactors have two-microorganism phases in mutualism life, the aerobic phase at the top, and the anaerobic phase at the bottom, bringing great advantages to the process (Campos et al., 2017). The treatment does not require the composting compound to be stirred and provides sufficient ventilation for the process, it does not produce unpleasant odors, leaching or even the formation of pollutants. In addition, it provides physical-chemical parameter control and can be used in different weather conditions (Campos et al., 2017; Zittel et al., 2018). Studies about stability and maturity of composts produced by physicochemical analyzes (temperature, humidity, pH, C/N ratio) associated with spectroscopic techniques (FTIR and UV-Vis) have been demonstrated that this process is efficient. In addition, the quality of the final compost can be verified through germination index (GI), concentration of heavy metals and pathogenic microorganisms (Campos et al., 2017).

Researchers have shown that DOW and SCT composting in 200 L volume facultative reactors from different initial C/N ratios yielded matured, stabilized and non-toxic composts (Zittel et al., 2018). The initial C/N ratio is an important point for the composting process, since these elements are sources of nutrients for decomposing microorganisms, which throughout the composting process tend to reduce. Recent studies have indicated that the initial C/N ratio is dependent on the chemical and physical properties of the residues used and have shown that studies with different initial ratios can contribute to the verification of the efficiency of the process in the treatment of a larger quantity of residue with higher toxic power (Silva et al., 2014).

Studies on reactor composting have been developed from a technical and scientific point of view for pilot and domestic scales, but no investigations were found in the literature presenting facultative reactors with different configurations for the treatment of larger volumes of waste.

Therefore, this search investigated the use of a 2000 L facultative reactor for the treatment of DOW and SCT. In addition, we attempted to verify the influence of different initial C/N ratios. It evaluated the degradation of the residues obtained by physicochemical and spectroscopic techniques and the compost quality using GI, microbiological test and metal concentration.

### 1.1 Nomenclature list

| | |
|---|---|
| DOW | Domestic Organic Waste |
| FTIR | Fourrier Transformation Infra Red |
| GI | Germination Index |
| IR | Infrared |
| R1 | Reactor 1 |
| R2 | Reactor 2 |
| SCT | Smuggled Cigarette Tobacco |
| UV-Vis | Ultraviolet- Visible |
| WC | Wood Chips |

## 2. MATERIALS AND METHODS

### 2.1 Assembling facultative reactors

The reactor presents a passive ventilation system on the top, which is responsible for the aerobic phase gaseous exchange (top microbial phase), the anaerobic phase occurs naturally as a result of the compaction due to the weight of the top phase, reducing the flow of gases.

The reactor also has a slurry draining system, with a filter and a tap in its base. Figure 1 shows the details of the facultative reactor designed in real scale.

### 2.2 Experiments

Two reactors were assembled. Reactor 1(R1) was loaded with DOW (64 kg) which was homogenized by cutting

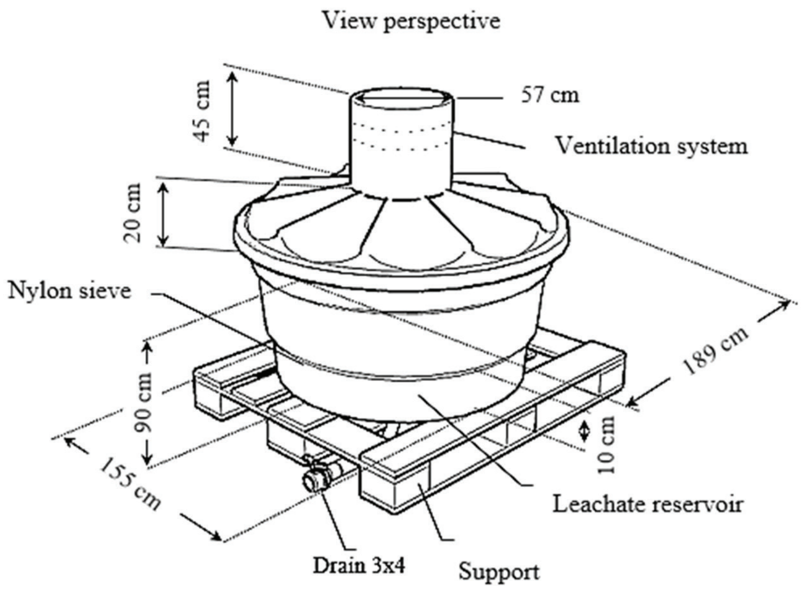


FIGURE 1: 2000 L facultative reactor.

the material into pieces, approximately 8-10 cm long, and mixed with WC wood chips (168 kg) into pieces of about 1-3 cm and SCT (64 kg).

Reactor 2 (R2) was loaded with DOW (120 kg) which was homogenized by cutting the material into pieces, approximately 8-10 cm long, and mixed with WC (120 kg) in pieces of about 1-3 cm and SCT (120 kg).

The influence of two different C/N ratios was investigated with different proportion residues of reactors started the process C/N ratio different, with R1 C/N ratio 28 and R2 C/N ratio 21 The mixture was homogenized and analyzed in several stages for the essential parameters (temperature, pH, C/N ratio, moisture content).

The proposal was to have a reactor working in hypoxic conditions, with top and bottom in mutualism. The top would contribute by capturing odors, if its layers were aerobic. Compounds and also to accelerate the compost stability via nitrification of the system (Campos et al., 2017).

### 2.3 Temperature, moisture, pH and C/N ratio

Temperature analyzes were performed daily, the analysis of pH, and moisture were carried out according to Fialho et al. (2010). The elemental chemical analysis to determine the C/N ratio used a TruSpec CN Analyzer (LECO® brand, St. Joseph, Mi, USA). During each collection, about 100 g of sample was collected from different points at the top and bottom of the reactor. These samples were homogenized and 20 g was used for analysis.

### 2.4 UV/Vis spectroscopy

The analyses were performed with samples of the composting process at 01, 30, 60, 90 and 120 days according to the modified method described by (Sellami et al., 2008). Were 10 mg of dry, comminuted compound was dissolved in 25 mL of 0.5 mol.L$^{-1}$ sodium hydroxide (NaOH) solution. The mixture was stirred for 1 hour and then centrifuged (7 min at 7000 rpm). The supernatant was diluted 2:1 and the pH of the solutions adjusted to a range from 8.3 to 8.6 with 2 mol.L$^{-1}$ HCl solution. UV-Vis absorption readings, wavelength 200 nm to 800 nm (were performed on the Varian Cary 50 BIO ® apparatus). The $E_4/E_6$ ratio was determined by the absorbance ratio in the range from 465 nm to 665 nm (Chen; Senesi; Schnitzer, 1977).

### 2.5 IR spectroscopy FTIR

The FTIR analyzes were performed with samples at 01, 30, 60, 90 and 120 days of process according to the method described by (Campos et al., 2017), using a spectrophotometer (brand Shimadzu ®, model IR PRESTIGE 21). The spectra were obtained in the range from 400 cm$^{-1}$ to 4000 cm$^{-1}$ from pellets prepared with 1.0 mg dry sample and 100 mg KBr.

### 2.6 Germination Index (IG)

For the germination test, the extraction was performed from the dissolution of the compost in distilled water and stirring for 30 min. After filtration in a qualitative filter paper, 5 mL of the extract was placed on petri dishes with filter paper and the seeds of *Lepidium sativum* (water cress) were added (Zittel et al., 2018). The incubation took place in a darkroom for 120 hours at 26°C. After this period, the number of germinated seeds and the root size were measured. The GI was calculated on the basis of eq. 1.

$$GI(\%) = \frac{seed\ germination\ of\ treatment\ (\%) - root\ length\ treatment}{seed\ germination\ of\ control\ (\%) - root\ lenght\ of\ control} x100 \quad (1)$$

### 2.7 Determination of metals in the final compost

The total concentration of the metals nickel (Ni), cobalt (Co), cadmium (Cd), chrome (Cr), lead (Pb), copper (Cu), zinc (Zn) and manganese (Mn) in the final compost was determined. The digestion process was carried out according to the method 3050B, United States Environmental Protection Agency US.EPA (1996), using a flame atomic absorption spectrometer – FAAS (brand Varian, model 240 FS).

### 2.8 Pathogenic microorganisms

For the analysis of the presence of Salmonella spp., 10 g. of the sample was diluted in 10 mL of sterilized water, the solution was seeded with SS agar (*Salmonella Shiguella*) and incubated in a bacteriological oven at 36°C for 24 h. Black colonies indicate that the sample is contaminated with Salmonella spp., method proposed by the National Sanitary Surveillance Agency/Brazil ANVISA (2004).

Coliforms were analyzed according to the method proposed by the FDA - Food and Drug Administration, bacteriological and analytical manual. The samples were diluted in 5% saline solution, $10^{-1}$, $10^{-2}$ and $10^{-3}$, and analyzed using the multiple tube technique (Feng et al., 2002).

Analyses of helminth viable eggs were carried out by the method of Hoffman et al (1934). The technique consists in dissolving about 10 g of compound in 10 ml of sterilized $H_2O$. It was filtered in gauze folded in four using a sedimentation cup. The beaker was filled with water and homogenized with a glass rod and allowed to stand for 24 hours.

With a Pasteur pipette, a sample was taken from the bottom of the vessel and the material was deposited on a glass slide with one drop of lugol solution covered with coverslip and examined under a 10x and 40x fold magnification optical microscope to investigate the presence of viable eggs.

## 3. RESULTS

### 3.1 Moisture, temperature, pH and C/N ratio

Regarding pH, R1 showed a variation from 5.0 to 6.9 on the first few days of the process and rose gradually up to 9.0 at the end of the 120 days. R2 started the process at pH 5.5, the top part had a gradual increase up to pH 8.7 while the lower part showed acidification at 4.9 in 15 days of the beginning of the process and then a gradual increase up to 8.8 in 120 days.

The temperature was monitored in the top and bottom regions of the reactors for a period of 120 days, as shown in Figure 2. Both had similar behavior.

After the 60-day period, the top and bottom parts showed similar behavior until reaching the ambient temperature in 120 days, revealing decrease in the microbial activity throughout the process.

Different initial C/N ratios for the reactors were evaluated. The initial ratio C/N in R1 was 28 and for R2 the initial C/N ratio was 21.

This work showed through the results of the spectroscopic analyzes that both initial C/N ratios evaluated presented the formation of matured and stabilized humic substances. The physicochemical analyses demonstrated that the stability of the microbial load responsible for the biodegradation of the organic matter was reached. The two ratios used had favorable results for biodegradation.

### 3.2 UV/Vis spectroscopy

The UV/Vis spectroscopic technique made it possible to obtain correlated data with aromaticity and also provided information of the molecular structure of the humic substances obtained after the composting processes (He et al., 2013).

Figure 3 shows the $E_4/E_6$ ratios obtained for R1 and R2.

Figure 3 reveals that during the composting process the $E_4/E_6$ ratios presented variations in their values due to the degradation of the organic matter and the formation of more condensed clusters. After 90 days of processing, these ratios were seen to decrease to values close to 5, indicating the formation of humified compounds. These results coincided with the temperature stability and consequent decrease in the activity of the decomposers microorganisms.

### 3.3 IR spectroscopy - FTIR

The FTIR spectra of the compound samples of the two reactors at different stages of the process are shown in Figure 4 and Figure 5.

No significant difference was observed between the FTIR spectra of the top and bottom regions of the reactors. There was a difference in the spectrum during the composting time.

The spectra of Figures 3 and 4 showed that they presented broadband in the 3450-3270 $cm^{-1}$ band, referring to O-H, N-H of alcohols, phenols and organic acids; 2924 – 2926 $cm^{-1}$, indicating the presence of aliphatic groups C-H ($CH_2$, $CH_3$); 1650 – 1660 $cm^{-1}$ C=C aromatic rings and C=O; 1519 – 1510 $cm^{-1}$, indicating the presence of C=C of aromatic groups, quinones attached to carboxyl and ketones, 1420 - 1410 $cm^{-1}$, C-O, C-O-H; carboxyl groups and 1060-1030 $cm^{-1}$ C-O polysaccharides (Fels et al., 2014; Campos et al., 2017).

### 3.4 Germination Index GI

GI is used as a mature compound indicator because it is a direct phytotoxicity meter of the compost produced on

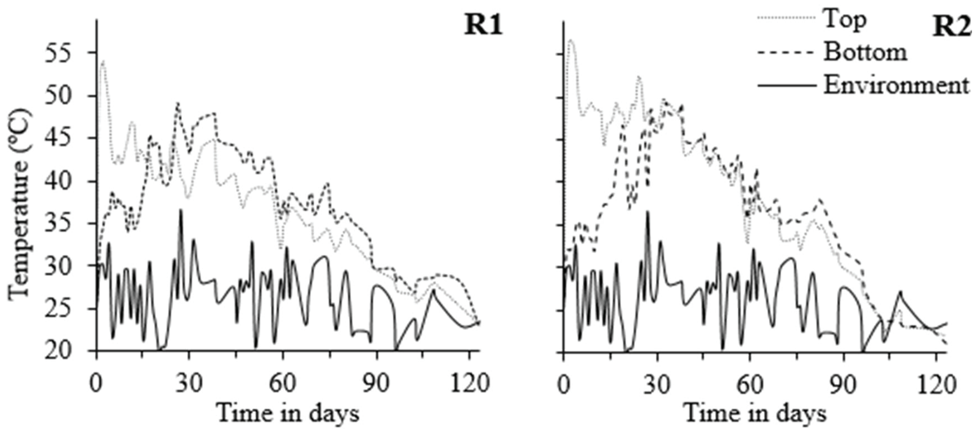


**FIGURE 2:** Reactors temperature during the composting process.

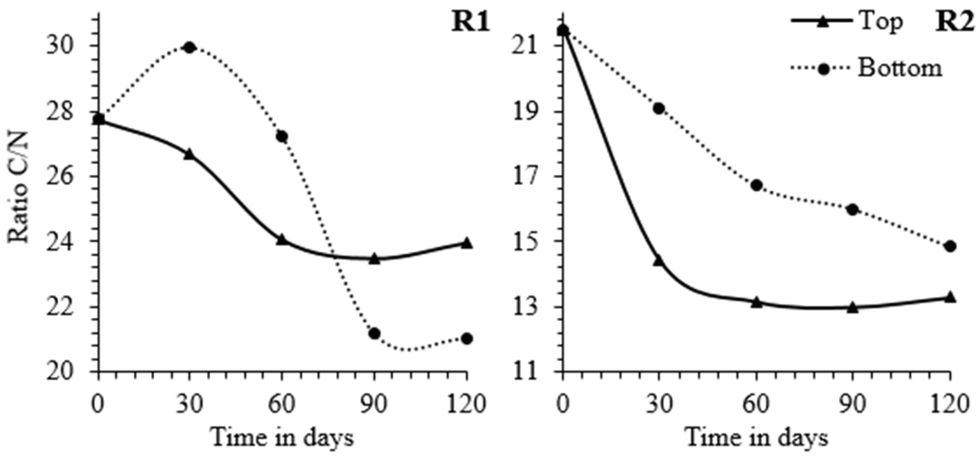


**FIGURE 3:** R1 and R2 ratio E4/E6; top and bottom.

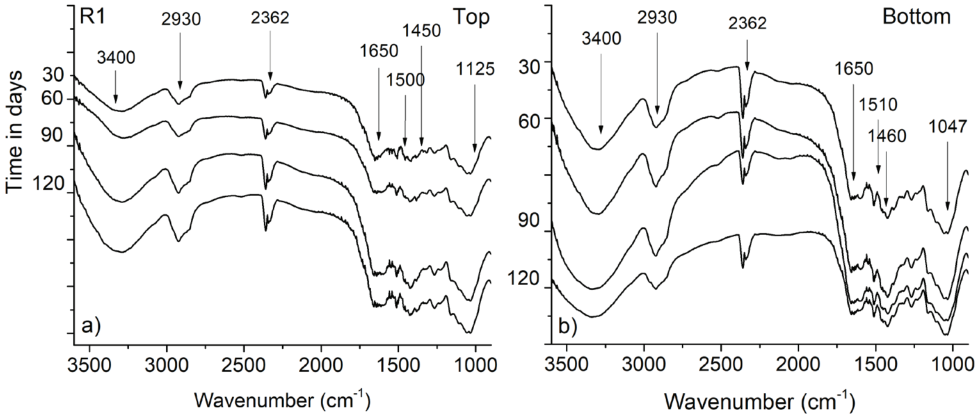


**FIGURE 4:** FTIR spectrum of the reactor 1.

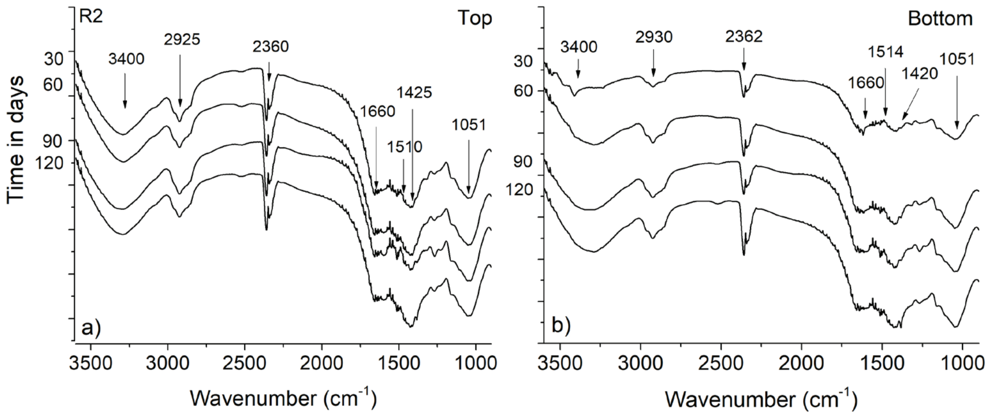


**FIGURE 5:** FTIR spectrum of the reactor 2.

seed germination (Wang et al., 2017b). For some authors, GI above 80 indicates a mature compost, free of phytotoxicity (Belo, 2011; Guo et al., 2012; Yang et al., 2016; Wang et al., 2017a).

In the composting process studied, R1 obtained GI results of approximately 80 in 90 days the process which was kept up to day 120. In 90 days of the process R2 showed GI close to 50, and reached the GI result 82 in 120 days.

### 3.5 Heavy metals

Table 1 presents the results of the metal analyzes in the final compost of the reactors.

Table 1 shows that the metal concentrations in the final compost of reactors 1 and 2 were lower than the maximum limits for heavy metals defined by the agencies – CCME Guidelines for Compost Quality 2005 and USDA Report and recommendation on organic farming 1999 Department of Agriculture.

### 3.6 Pathogenic microorganisms

Table 2 shows results of the microbiological analyses in relation to pathogens of reactors 1 and 2.

Table 2 shows that the analyses of the reactors revealed absence of salmonella and helminths viable eggs of and the result of thermotolerant coliforms in R1 was <3 and while R2 presented 23MLN/g.

The table also provides the maximum limit of contaminants required by the legislations, according to the Ministry of Agriculture, Livestock and Supply/ Brazil MAPA (2014), CCME (2005) and US.EPA (2003).

The final compost analyzed is in accordance with the required microbiological quality standard for MAPA (2014), CCME (2005) and US.EPA (2003), identified in Table 2, providing adequate final compost for pathogenic contaminants.

## 4. DISCUSSION

### 4.1 Moisture, temperature, pH and C/N ratio

Moisture is an important parameter to be evaluated in composting, as it ensures an adequate environment for the microbial development of the decompose. The initial moisture in the two reactors evaluated was close to 70% and at the end of the process the humidity reached approximately 40%.

Moisture above 70% hinders the gas exchange in the composting environment and below 40% prevents the microbial growth responsible for the degradation of organic matter (Richard et al., 2002; Khalid et al., 2011; Campos et al., 2017). In both reactors the evaluated moisture content was suitable for the development of decomposing

**TABLE 1:** Heavy metal concentrations in R1 e R2 and maximum limits allowed.

| | R1 (mg $kg^{-1}$±SD) | R2 (mg $kg^{-1}$±SD) | USDA (mg $kg^{-1}$±SD) | CCME (mg $kg^{-1}$±SD) |
|---|---|---|---|---|
| Cu | 16.4±1.2 | 20.5±0.7 | 1000.0 | 400.0 |
| Mn | 285.4±11.3 | 359.7±9.8 | - | - |
| Fe | 775.0±68.9 | 790.5±72.8 | - | - |
| Ni | 9.2±1.2 | 13.6±1.8 | 200.0 | 62.0 |
| Cd | 2.1±0.1 | 1.9±0.1 | 10.0 | 3.0 |
| Pb | 9.1±0.6 | 9.7±0.3 | 250.0 | 150.0 |
| Cr | <LQ | <LQ | 1000.0 | 210.0 |
| Zn | 35.7±2.2 | 56.8±7.4 | 2500.0 | 700.0 |

*SD - standard deviation; CCME - Guidelines for Compost Quality, Canada (2005); USDA - American Department of Agriculture. Limit of metals in the composts produced by aerobic digestion (1999).*

**TABLE 2:** Concentration of pathogens in R1 and R2 and maximum limits allowed.

| | R1 | R2 | MAPA (2014) | CCME (2005) | US.EPA (2003) |
|---|---|---|---|---|---|
| Thermo-tolerant coliforms (MLN/1g DM) | <3 | 23 | 1000 | 1000 | 1000 |
| Salmonella spp. (MLN/10g DM) | < 1 | < 1 | < 1 | < 1 | < 1 |
| Helmints viable eggs (MLN/4g of TS) | < 1 | < 1 | 1 | 1 | 1 |

*MLN = most likely number; DM = dry matter; TS = total solids.*

microorganisms.

Studies in aerobic, anaerobic and facultative reactors found that pH decreases during the first weeks of composting due to the formation of organic acids (amino acids and other volatile fatty acids). After this period, neutral pH values are reached due to the conversion of these acids into carbon dioxide by the action of microorganisms (Iyengar; Bhave, 2006; Campos et al., 2017; Razaa, Munirb, Nazb, Ahmedb, & Ameen, 2017; Zittel et al., 2018) .

The evaluation of composts with TCC in reactors showed that an increase in pH might occur due to the evolution of the ammonia and loss of nitrogen of the substrate. In view of this, the matured compost can promote the immobilization of nitrogen in the soil, reducing $NH_4^+$ and increasing $NO_2$ and $NO_3$, useful compounds as plant nutrients (Kopcic et al. 2014, Oviedo-Ocaña et al., 2015; Zittel et al., 2018).

Temperature around 40-60°C indicates that the ecosystem is well balanced and the microbiological activity will favor the degradation of organic matter (Xie et al., 2016).

The temperature can also be affected by the process porosity (Chowdhury et al., 2014). As the bottom supports the entire weight of the compost above it, the process then takes place under lower porosity and higher compaction. At the bottom of the reactor there is no such natural exchange of gases and, in addition, there is natural compression by the top mass (Zittel et al., 2018).

The time/temperature relationship is closely linked to the inhibition or growth of pathogenic microorganisms (Jones; Martin, 2003; Campos et al., 2017). For elimination of pathogens the temperature should remain above 40ºC for a minimum of 5 days, during which time the temperature exceeds 55°C for at least 4 hours US.EPA (2003). In both reactors the temperatures defined as appropriate for pathogen elimination were reached.

The carbon present in the waste is used by the microorganisms as a source of energy and nitrogen is used for cell synthesis. There is consensus in the literature that the ideal C/N ratio is between 25/1 and 30/1(Wu et al., 2015). However, lower initial C/N ratios mean treating larger amounts of pollutant residues.

Studies show that the initial C/N ratio depends on the compost residue, it is possible to start the composting process with ratios below 20 and to obtain a stabilized and matured humified compost (Yen; Brune, 2007; Silva et al., 2014; Wu et al., 2015) .

## 4.2 UV/Vis spectroscopy

In the analysis of the initial and final values of the ratio for samples from reactors 1 and 2, there was a decrease in the ratio with the increase in time of the composting process. This decrease in ratio may have occurred due to the mineralization of carbohydrates and quinones, oxidation of phenolic compounds and the presence of methoxy groups and/or aliphatic side chains attached to the humic substances (Sellami et al., 2008).

The $E_4/E_6$ ratio is inversely proportional to the degree of condensation of the carbon-carbon condensation structures, so that high $E_4/E_6$ values indicate a low degree of condensation of carbon-carbon instabilities and the presence of various aliphatic structures. Values below 5 for $E_4/E_6$ reflect a high degradation and formation of condensed aromatic chains, demonstrating a high degree of humification of the organic matter (Campos et al., 2017; Campos; Ressetti e Zittel, 2014).

The relationship $E_4/E_6$ has so far been scarcely used to evaluate composting processes in reactors. In this study, the results indicated that composting in the 2000 L facultative reactor produced final composts with a high degree of humic acids present in the structures of humic substances.

## 4.3 IR spectroscopy - FTIR

The spectra exhibited absorbance in similar regions, however, they differed in the intensity of some peaks, observing the decrease in peaks in the region of 2920 $cm^{-1}$ and 1100 $cm^{-1}$, indicating the biodegradation of aliphatic groups (Ouaqoudi et al., 2014).

The peak near 2315 $cm^{-1}$ occurred because of the $CO_2$ trapped in the KBr matrix (used in the pellets for the IR analysis), apparently from the decarboxylation of the COOH group (Stevenson, 1994)(STEVENSON, 1994).

In the 1660 $cm^{-1}$ region the intensification of the characteristic carbon-carbon unsaturation band, ascribed to the C=C stretching of the aromatic ring, can be observed for the aromatic ring vibrations of carbon-carbon unsaturation together with C=O and/or $COO^-$, indicating the condensation of the carbonic chains (Campos; Ressetti; Zittel, 2014) (Campos; Ressetti; Zittel, 2014).

The quinones attached to carboxylates and ketones presented intensification in the band near 1508 $cm^{-1}$, which indicates the elongation of carbon-carbon from aromatic groups (Droussia et al., 2009).

The increase in C of aromatic and the decrease in C of aliphatic chains are considered indicators of increased organic matter degradation. For the composting process, these changes may be associated with the stability and maturity of the composts with their highly humified substrate transformations (Droussia et al., 2009).

Recent work on composting in reactors has demonstrated the use of FTIR, which presents absorption peaks, indicating organic matter transformation and formation of mature compounds, characterizing the increase in humic acids during the process (Fels et al., 2014; Campos et al., 2017; Zittel et al., 2018).

The results presented in the FTIR evidenced that the two reactors had the same effect, but there was no difference in the intensity of the peaks. Both reactors indicated that tobacco degradation mixed with DOW and WC occurred. The evaluation of a large volume of waste improved the process with higher organic matter degradation and formation of humified compost.

### 4.4 Germination Index GI

The increase in GI is attributed to the transformation of toxic compounds such as ammonia, volatile fatty acids, phenolic compounds, among others into mature compounds, free of toxicity (Guo et al., 2012).

For different authors, a GI above 80 indicates maturate compound, free of phytotoxicoses, which can be considered not toxic above 50 (Belo, 2011; Guo et al., 2012; Yang et al., 2016; Wang et al., 2017a). Thus, R1 required 90 days, while R2 needed 120 days to reach maturity.

Higher proportion of SCT in R2 may have caused the longer time to reach GI, since tobacco is considered toxic, however, some authors argue that GI above 50 is already considered to be toxicity free (Zittel et al., 2018).

These results indicate that the degradation of a large amount of tobacco mixed with DOW and WC occurred and therefore can be treated in a reactor system.

### 4.5 Heavy metals

Even SCT presenting high concentrations of toxic metals, the facultative reactor showed efficiency in its treatment, this must be due to the dilution that occurs with the use and mixing of different residues (Fels et al., 2014; Zittel et al., 2018). The results showed that the use of large volumes of SCT in the composting process produced a compost with a concentration below the established limits.

### 4.6 Pathogenic microorganisms

According to the results, the final compound for the two reactors was observed to show absence of pathogens, regarding the studied groups. The absence of pathogens occurred due to temperature variation throughout the process, which remained above 40°C for more than five days in the two reactors, besides reaching peaks of 55°C, temperatures considered by the US.EPA (2003) sufficient for disposal or reduction.

Several mechanisms of pathogenic inactivation are possible, including thermal inactivation, microbial competition, toxicity (produced by the composting process itself, such as ammonia, sulphites, organic acids and phenolic compounds) and enzymatic rupture (Wichuk; Tewari; Mccartney, 2011). The time/temperature relationship is closely linked to the inhibition of growth of these microorganisms or the proliferation thereof and are the most used factors (Jones; Martin, 2003).

## 5. CONCLUSION

Due to the results obtained, residue degradation success was demonstrated for the proposed SWO cigarette mixtures in the two different C/N ratios evaluated. The physicochemical and spectroscopic analyzes (UV-Vis and FTIR) showed the stability and maturity of the final compost. And its quality was confirmed by the absence of phytotoxicity (obtained by GI), potentially toxic metals and pathogenic microorganisms.

Therefore, the 2000 L facultative reactor is a suitable technology for treating large volumes of cigarettes and SWC, and unlike the reports already found in the literature, it has the advantages of having low cost, not attracting vectors, controlling temperature and humidity without the need of management, thus becoming a promising technology for the management of organic solid waste.

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# GENERATION OF BIO-BASED PRODUCTS FROM OMSW BY USING A SOLID-LIQUID SEPARATION TECHNIQUE AND AN ANAEROBIC TREATMENT

Jan Kannengiesser [1,*], Celina Kuhn [1], Timo Mrukwia [1], Daniel Stanojkovski [2], Johannes Jager [1] and Liselotte Schebek [1]

[1] Technische Universität Darmstadt, Institute IWAR, Faculty of Civil and Environmental Engineering, Franziska-Braun-Strasse 7, 64287 Darmstadt, Germany
[2] Jager Biotech GmbH, Roßbergweg 7b, 64380 Roßdorf, Germany

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**ABSTRACT**

The present paper provides an overview of the investigations (involving different liquids) regarding a new technology that is able to generate valuable bio-based products by using the liquid phase from organic municipal waste as raw material. The liquids used in this study were tested and treated in different ways to find out which substrates are most suitable for the process. For the purpose of generating bio-based products from organic waste, a solid-liquid separation process was performed first. Thereafter, in order to increase the amount of non-polar fatty acids (FAs) in the liquid substrate, an anaerobic digestion process was used. However, after the digestion, most of the FAs found in the liquid substrate were polar FAs. To increase the amount of non-polar FAs further, another treatment step, "ethanol maturation", was carried out as the third step. Subsequently, the refining process was started with the extraction of the FAs from the liquid substrate by using a non-polar extraction solvent, such as Oleic acid methyl ester (OME). At this point, "extractive digestion" takes place. The FAs can be extracted over a longer period of time and during the digestion. As a result, the amount of longer-chain fatty acids in the OME increases. After re-extracting the FAs from the solvent, a transesterification process was used to produce fatty acid ethyl esters, which can be sold as cleaners or solvents to the metal industry for surface treatment. The production of other bio-based products, such as lubricants, fuels and polymers, is also possible. The following five substrates were mainly used for the investigations: (1) a liquid phase from organic municipal waste, produced by a percolation process; (2) fresh percolate from German kitchen waste; (3) percolates of different stages from a digestion plant in Germany; (4) old leachate from German landfills; and (5) young landfill leachate or leachate from pressed municipal solid waste, which was used to imitate African waste. These substrates were treated using the method described above. The results show that young landfill leachate has the highest potential. The highest availability of FAs occurs after biological pre-treatment of percolates. For this reason, very fresh percolates hardly contain any fatty acid. They must first be generated by the biological degradation of the organic ingredients.

## 1. INTRODUCTION

In 2016, the amount of waste produced by households in Germany was 46,6 million tonnes. Around ten million tonnes of these wastes were classified as biological waste (including garden and park waste) (Destatis, 2018a). Biological wastes are also produced in different industrial sectors. In 2016, about 4,4 million tonnes of biological wastes were collected separately in households (Destatis, 2018a). The amount of the collected biological waste is estimated to increase since 1 January 2015 (Krause et al., 2014). As of this date, the Kreislaufwirtschaftsgesetz (KrWG) contains the German implementation of the EU Waste Framework Directive into national law. The law enforces the change in the collection system of biological waste from voluntary to obligatory, §11 KrWG (BmJV, 2012). These wastes can still be recycled or used to generate energy. At the moment, the dominant waste treatment method is

* Corresponding author:
Jan Kannengiesser
email: j.kannengiesser@iwar.tu-darmstadt.de


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composting. Presently, separately collected bio-waste is typically recycled in a composting or anaerobic digestion (AD) system, producing compost (or digestate) as fertilizer and soil conditioner, and in the case of AD biogas as renewable energy (Andreottola et al., 2012; Edwards, 2015). However, in the near future, other forms of high-value bio-waste recycling methods, such as the production of bio-products and chemicals in bio-refineries, are expected to also be available in the market (De Jong et al., 2012). The new idea of using biodegradable waste is meant to combine a composting plant with a facultative anaerobic treatment step. Organic acids carry the main part of the energy content of biomass at the early stage of anaerobic digestion process and are easy to transfer into liquid products by further biorefinery steps. After the extraction of organic acids, the residual waste can be transported to the composting plant to generate high quality compost. Based on the separate collection of the different waste fractions, a new biotechnology focusing on the special requirements for these kinds of waste can be developed.

Biogenic waste is not only produced in the waste management sector but also in the agricultural sector. More than 56,4 million tonnes of agricultural waste are generated every year in Germany (Destatis, 2018b). These wastes are also treated by composting or mainly digestion. Based on the new law for renewable energy in Germany, the treatment of agricultural waste is getting more and more expensive, because co-financing of treatment facilities was changed by the law. Also, in the agricultural sector, innovations are needed in order to make the treatment processes more efficient (economical).

In developing countries, other suitable substrates are fresh landfill leachate and the water obtained by pressing the collected waste (Mahmud et al., 2012). It is common knowledge that developing countries are still deprived of sanitary landfills and still do not employ the multi-barrier system. The waste is not pre-treated, and there are no covers used to protect the environment from greenhouse gases (GHG) emissions and hazardous substances. Even today, nearly all municipal solid waste (MSW) is just landfilled in developing countries, like Morocco. Waste analyses have shown that the landfilled waste contains a high amount of water, 50-70 wt% of landfilled MSW (Smahi et al., 2013). This high amount of water leads to the generation of an enormous amount of landfill leachate. In this research, young landfill leachate and the water obtained by pressing the collected waste will be tested as substrate for the new technology.

The application of the new integrated technology in developing countries could help to minimise the problem of landfill leachate production and create a new platform for industrial bio-based chemicals.

## 2. MATERIALS AND METHODS

In this chapter, the processes involving the different liquid wastes are described. In the first subsection, the investigations in Germany are presented, while the subsequent subsection shows the treatment process using typical waste from Africa.

### 2.1 Investigations with biological waste from Germany

This chapter describes the generation of non-polar fatty acids from organic waste and the investigations into the production of bio-based products from this type of fatty acid. Four of the five substrates were directly made out of biological waste from Germany. Two of them were based on the biological wastes from Darmstadt, which were collected from private households and treated at the composting facility in Darmstadt. One of them was obtained from biological waste from the digestion plant in Lemgo. By these examples, the different treatment stages were sampled. The last one originated from German landfills and was taken directly on site.

*Medium 1: Biological waste from a composting facility*

The first experiments were performed from 2012 to 2014. These experiments served as reference experiments for the subsequent tests. Based on these data, the processes were extended and other liquids were treated.

As a first step, a solid-liquid separation process was integrated into the existing treatment concept of the composting facility in Darmstadt. The separation of the liquid and the solid phases took place in a modified rotting box. This rotting box was equipped with a percolation system, which allowed repeated percolation of the liquid phase of bio-waste from the solid phase in order to transfer the majority of the organic compounds from the solid phase to the liquid phase. Next to the percolation system, the rotting box was also equipped with a special air circulation system that allowed not only the injection of fresh and circulating air through the bio-material but also the injection of exhaust air collected from other rotting boxes. The schematic diagram of the rotting box is shown in Figure 1.

These modifications led to the creation of facultative anaerobic conditions in the box and the acidification of the bio-substrate. After the percolation process, non-polar fatty acids (FAs) were found in the liquid phase (Kannengießer, 2015).

To increase the amount of non-polar FAs in the liquid substrate, an anaerobic digestion process was used.

The treatment involved the digestion of the liquid phase in the modified rotting box by starting the aeration with warm exhaust air from other rotting boxes. The exhaust air had a temperature of 55 to more than 60°C. By storing the liquid phase in intermediate bulk containers (IBC) in the rotting box, an anaerobic digestion process started, and the organic fraction in the liquid substrate was degraded into carboxylic acids.

Even after digestion, the FAs found in the liquid substrate were mostly polar FAs. To increase the amount of non-polar FAs once more, a treatment step called "ethanol maturation", which involves the addition of ethanol, was carried out next. Some anaerobic bacteria, such as Clostridium Kluyveri, can convert polar FAs to non-polar FAs by using ethanol and polar FAs for its anaerobic energy metabolism. The reaction of the process is shown in Equation 1.

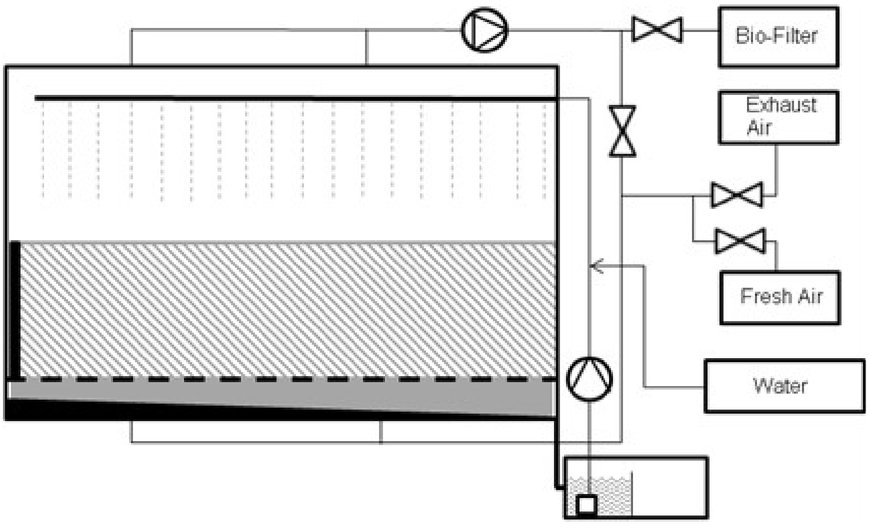


**FIGURE 1:** Schematic diagram of the modified rotting box (Kannengiesser, 2015).

Reaction of acetic acid and ethanol to produce hexanoic acid and water (Levy et al., 1981):

$$2CH_3CH_2OH + CH_3COOH \rightarrow CH_3(CH_2)4COOH + 2H_2O \qquad (1)$$

After the ethanol maturation process, a liquid substrate of bio-waste with a high content of non-polar FAs was transported to a refinery.

Based on the low water solubility of the non-polar FAs, which increases with the length of the carbon chain of the FA, there is a specific acid concentration in the liquid substrate of bio-waste above which the production of non-polar FAs stops (Kannengießer, 2015).

The microorganisms are no longer producing medium chain fatty acids (MCFA) from short chain ones because of the saturation. In order to find a solution to this problem as well as increase the generation of non-polar FAs, an extraction solvent was added during the processes of digestion and ethanol maturation. By adding the extraction solvent, the non-polar FAs were directly extracted, thereby avoiding saturation, and the microorganisms could go on with the production of non-polar FAs from polar ones. This process is called "extractive digestion". The treatment process of the bio-waste in order to generate bio-based products is described in Figure 2.

*Medium 2: Kitchen waste*

This medium was made of the liquid phase of kitchen waste. It originated from restaurants, schools and canteens in Darmstadt, Germany. For the liquid-solid separation process, some waste bins were filled with this biological material and simply separated by tipping over of the bins into solid and liquid phases. This separation process is shown in Figure 3.

As a result, wastes with various colours and phases were generated from the unseparated waste components. Figure 3 shows the distinctions clearly. The treatment processes were conducted with a liquid volume of 4 to 9 litres.

*Medium 3: Biological waste from a digestion plant*

The liquid samples were taken from different treatment steps of a digestion plant in Germany. In particular, the aim was to find out which step had the highest fatty acid potential. Therefore, liquid samples were taken from the pre-rotting stage and after the fermentation process.

*Medium 4: German Landfill Leachate*

Three different landfill leachates from Germany were sampled. According to the Council Directive on the landfill of waste in Germany, usually, mineral and inert materials are deposited. This means that only a small amount of organic material is deposited on the landfills. The samples were taken before the leachate treatment step.

### 2.2 Investigations with Substrate from developing countries

The investigation with substrate from developing countries also took place in Darmstadt. In small-scale tests, landfill reactors with a total volume of 120 l were simulated. Although the waste was also taken from the solid

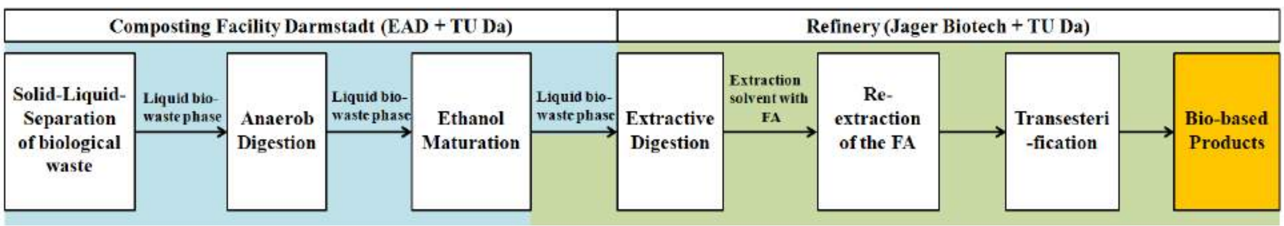


**FIGURE 2:** The "bio-waste to bio-based products" stream.

FIGURE 3: Treatment of the kitchen waste to get the liquid phase.

waste of this city, it was adapted to have the composition of waste from Morocco. Approximately 90% of the waste could be installed in one reactor. After the airtight sealing of the reactors, they were always stored at a temperature of around 38°C in a test container.

Afterwards, the fermentation process started under anaerobic conditions. A sketch of a reactor cross-section is shown in Figure 4. A system was used to simulate in every reactor the constant compressive stress in a landfill site. The leachate was removed regularly by means of a valve (Medium 5). In order to simulate rainfall events, liquids were added to the system via a lockable opening. The gas was always collected outside the container in an air bag.

## 2.3 Post-treatment of the different mediums

The post-treatment of all the liquids were based on the first experiments (described in 2.1). In accordance with this, further maturing and extraction took place after the separation of the liquid phase. About 4 to 10 litres of the liquids were stored in an extra canister at a temperature below 38°C. The ethanol maturation was carried out by the addition of ethanol gotten from the percolation process, as described above in the previous experiments. The chain elongations proceeded according to the reaction in Equation 1. One week after the extractive digestion started, longer-chain fatty acids could be extracted by adding the extraction medium, Oleic acid methyl ester (OME). Also, in this case, two different post treatment methods were applied, the punctual extraction and the new extractive digestion.

## 2.4 Refining process

The refining process commenced with the extraction of FAs from the liquid substrate by using a non-polar extraction solvent. Two different extraction solvents were tested. First, a vegetable oil fuel, made out of used kitchen oil, was used to extract MCFA. The problem of this extraction solvent is that it changes composition. The change in composition of the extraction solvent is dependent on the used kitchen vegetable oil. This makes it nearly impossible to analyse the extracted MCFA in the solvent, because the method of analysis has to be optimised to the composition of the extraction solvent. Considering this, in 2015, another

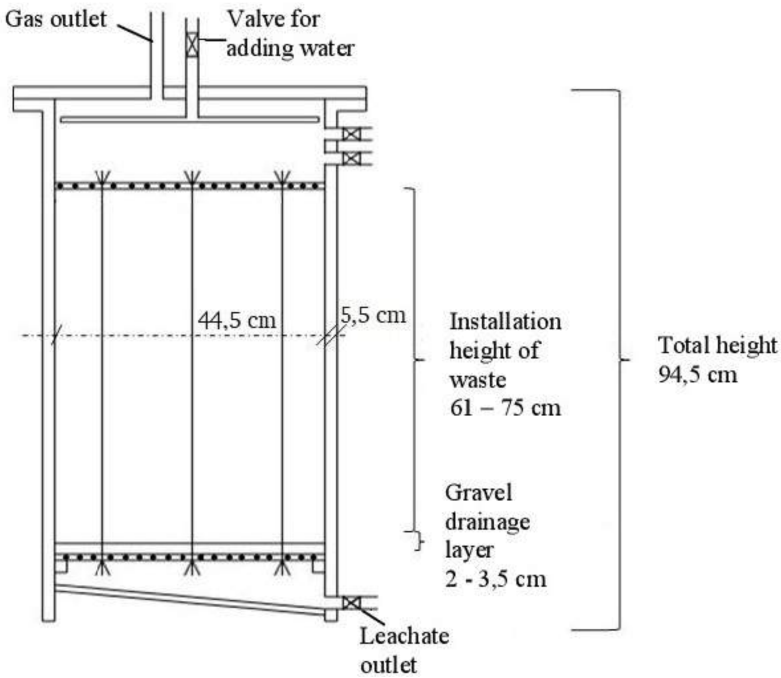


FIGURE 4: Schematic diagram of the landfill reactor.

extraction solvent (Oleic acid methyl ester OME) with similar properties but a stable composition was used.

By using a non-polar solvent, majority of the non-polar FAs were dissolved in the extraction solvent, and the polar FAs remained in the liquid phase of the bio-waste substrate.

After the extraction of the non-polar FAs into the solvent, they were re-extracted by a special extraction technology. After re-extracting the FAs from the solvent, a transesterification process was used to produce fatty acid ethyl esters, which can be sold as cleaners or solvents to the metal industry for surface treatment. The production of other bio-based products, such as lubricants, fuels and polymers, are also possible.

The generation and extraction of FAs as well as the production of bio-based products have been proven to be possible by a project involving TU Darmstadt, Eigenbetrieb für kommunale Aufgaben und Dienstleistungen (EAD), Jager Biotech and Handelshaus RUNKEL (LOEWE-Verbundprojekt 350/12-40, 2014).

## 3. RESULTS AND DISCUSSION

In this chapter, the results of the investigations with substrate from Germany and the typical substrate from Africa are presented. First, the different waste characteristics of the used substrates are given. Next, based on these characteristics, the properties of the different liquid substrates are demonstrated and discussed. The third part of this chapter shows the potential of extraction of carboxylic acids from these two substrates. Furthermore, the enhancement of the treatment by the new method is pointed out. Finally, the potentials of the different substrates are explained.

### 3.1 Substrates' characteristics – German bio-waste and residual waste from Africa (Morocco)

As explained earlier, the first experiments were performed with liquid substrates generated from biological and residual wastes from Darmstadt, Germany. Afterwards, the produced results and investigations were compared with the simulated Moroccan waste. Based on previous studies (Soudi and Chrifi, 2007), a typical residual waste of Morocco was generated by using biological and residual wastes from Germany. A comparison of both wastes is shown in the following diagram.

As shown in Figure 5, the basic compositions of the two substrates are quite similar. Both have a high amount of organic compounds, about 75 wt% for the African waste and 83 wt% for the German waste. The amount of residual waste is between 14 and 10 wt%, and the amount of plastic is about 3.5 to 5.7 wt%. However, there is a significant difference between both substrates with respect to the fraction of organic matter. In the used German waste, 75 wt% of the organic matter can be defined as green waste, while the organic fraction of the African waste consists of nearly 100 wt% of kitchen waste.

Also, the comparison of the dry matter content showed similar results. The German waste consists of 39 wt% of dry matter, while the dry matter of the African waste is 34.5 wt%. Also, 68.5 wt% of the dry matter can be defined as organic in the German waste, while only 60 wt% of the dry matter of the African waste is organic.

### 3.2 Characteristics of the liquid waste fractions

In this subchapter, the major properties of the two liquid substrates, which were produced by the different kinds of waste previously described, are presented. Major parameters, such as pH-value, electrical conductivity, redox potential, the sum parameter of organic acids (shown as acetate equivalents), the chemical oxygen demand (COD), the total nitrogen and total phosphorus, are analysed. The organic acids, COD as well as total nitrogen and total phosporus were analysed by quick tests of Hach Lange (LCK 365 for organic acids, LCK 514 for COD, LCK 138 for total nitrogen and LCK 348 for total phosphorus).

As shown in Table 1, the two different substrates have similar pH-values, about 5, and a negative redox potential, which indicates anaerobic conditions. However, the concentration of organic acids and the COD show big differences. For the German waste, on average, the COD is about 62,611 mg $O_2$/l, and the amount of organic acids is about

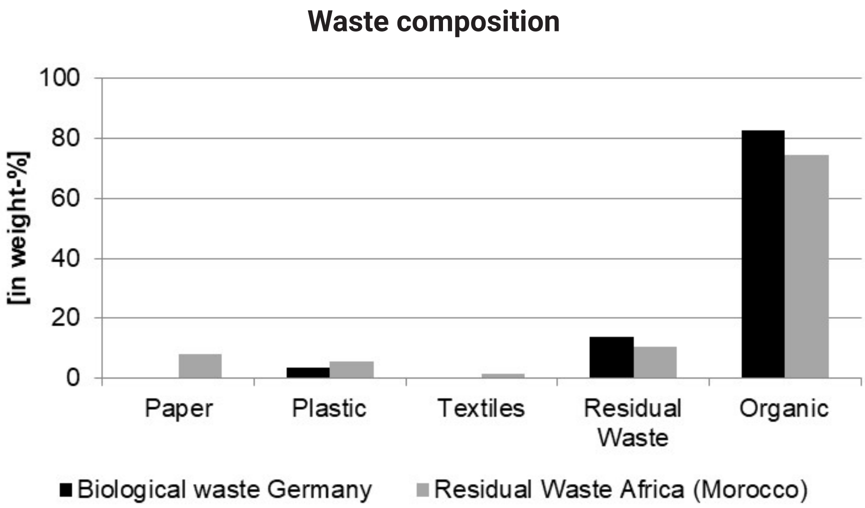


**FIGURE 5:** Comparison of the used wastes from Germany and Morocco.

20,418 mg $CH_3COOH$/l. On the other hand, the COD of the liquid phase of the African waste and the concentration of organic acids are about 50% and 25% higher than the respective German values. These simple analyses indicate that the liquid phase produced from African waste has more organic acids, which can be used for the extraction process. German and African bio-wastes used for this study as substrates were analysed to compare total nitrogen and phosphorus values of these materials.

The results of total nitrogen show a slight difference between African and German bio-wastes. The average nitrogen concentration in the liquid phase of the German as well as African wastes is approximately 3,000 mg/l and 3,600 mg/l, respectively. The range of total nitrogen in African wastes is from 2,997 ± (45) to 4,243 ± (94) mg/l with a very low relative standard deviation between 1.5 and 2.2%. Total nitrogen in German wastes is in the similar range, namely, 2,997 ± (96) mg/l.

In contrast, a significant difference was observed between total phosphorus in the African and German wastes. Total phosphorus of the African bio-wastes ranges from 7.4 ± (0.92) to 323.7 ± (106.8) mg/l, whereas total phosphorus of German bio-waste reaches about 25.77 ± (5.49) mg/l. The large fluctuation in total phosphorus in the liquid substrate from African wastes could result from their heterogeneous compositions of the bio-wastes fed to the reactors. Unlike German wastes, these African wastes were not homogenized and different kinds of organic wastes were used in the reactors.

The concentration of nitrogen and phosphorus of the liquid phase of these bio-wastefrom Africa and Germany is low compared to other similar substrates in Germany, like liquid fertilizers from digestion plants. The average concentration of nitrogen and phosphorus in liquid fertilizers are about 5,000 – 5,400 mg and 1,500 – 1,800 mg/l, respectively (BGK, 2013). However, low nitrogen and phosphorus content of our tested substrates will not inhibit the microbiological process. In order to optimize the microbiological reactions, the effect of bio-waste compositions on the microbiological reactions in the reactor could be further investigated.

**TABLE 1:** Comparison of the liquid waste phases from Germany and Africa.

| Measured Parameters | Liquid phase from | |
|---|---|---|
| | German waste | African waste |
| pH-value [-] | 5.23 ± (0.73) | 5.12 ± (0.8) |
| Electrical conductivity [mS/cm] | 18.82 ± (2.48) | 28.85 ± (4.72) |
| Redox potential [mV] | -188.42 ± (135.3) | -95,99 ± (61.23) |
| Organic acids [mg $CH_3COOH$/l] | 20,418 ± (4,788) | 25,075 ± (3,058) |
| COD [mg $O_2$/l] | 62,611 ± (15,207) | 95,944 ± (24,144) |
| Total Nitrogen [mg/l] | 2,996.67 ± (95,68) | 3,620 ± 667.67 |
| Total Phosphorus [mg/l] | 25.77 ± (5.49) | 165.53 ± (175,24) |

### 3.3 Results of gas chromatography – mass spectrometry (GC-MS) analyses of the liquid phases

In this study, samples taken from the substrates were diluted in a ratio of 1:10 with Milli-Q water, and the pH was adjusted to 1.5 or 3.0 by adding 1N-HCl solution. Samples were filtered using a micro filter (WIC 80845: 0.45 µm PTFE with glass fibre, WICOM, Heppenheim) before the GC/MS analysis.

The analysis was conducted with Trace GC Ultra coupled with ISQ-MS, electron ionisation (EI) as well as to TriPlus RSH liquid injection autosampler (Thermo Scientifics, Dreieich). The column used for this study was TG WAX-A (30 m; i.d. 0.32 mm; thickness 0.50 µm; stationary phase: polyethylene glycol; Thermo Scientifics, Dreieich). For each measurement, 0.5 µL of the sample was injected into a split/splitless injector heated at 260°C and analysed at a split ratio of 1:20. The GC oven programme was set as follows: at 80°C for 1 min; 20°C/min till 120°C; 6.1°C/min till 205°C; at 205°C for 10 min. In Figure 6, the comparison of carboxylic acids in the two liquid waste phases is shown.

As shown, three major compounds, acetate, butanoic acid and hexanoic acid, can be found in the liquid substrates. The African waste samples mainly contained acetate (40 wt%) and butanoic acid (36 wt%). In the liquid phase of the German waste, the main compounds were acetate (35 wt%) and hexanoic acid (24 wt%).

These acid compositions could be based on the compositions of the different wastes that were used for the generation of the liquid substrates. For experiments involving African waste, kitchen waste was used as organic compounds. For Germany, organic waste mainly contained green waste, consisting of longer organic substances.

### 3.4 Extraction potential and product estimation

The liquid phase obtained from the German bio-waste underwent chain elongation with the addition of ethanol. After the addition of ethanol, the fatty acids were enlarged, as described in Equation 1.

Extraction tests on the liquid phase from the German bio-waste showed the production of acetate (about 37 wt%), propionic acid (about 7 wt%), butanoic acid (about 30 wt%), pentanoic acid (about 15 wt%), hexanoic acid (about 34 wt%), heptanoic acid (about 10 wt%) and octanoic acid (about 13 wt%).

Nevertheless, the extraction procedure was done with mediums 1 and 2 of the liquid substrates. An estimation of extractable carboxylic acids and producible bio-based products using only one extraction step involving liquid waste from Germany is shown in Figure 7.

Using a single extraction step, only 20-25 mg of MCFA could be extracted from the biological waste of Darmstadt. The following figure shows the material flows within the composting facility in order to extract these MCFA.

Based on the data, in the treatment process of the used vegetable oil and the extracted MCFA from the biological waste, about 105 mg of bio-based product in form of a bio-based fuel with bio-diesel properties was generated at the composting facility in the study carried out in 2014.

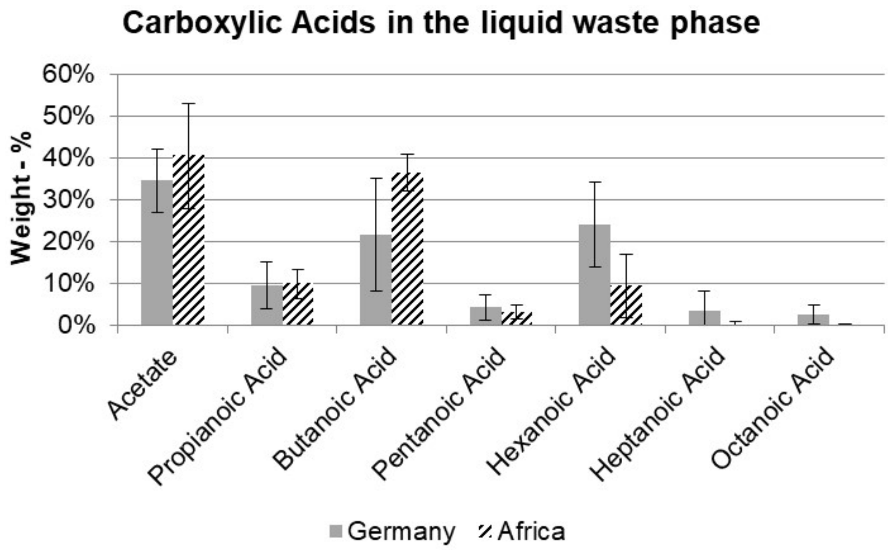


**FIGURE 6:** Concentration of carboxylic acids in the liquid waste phases.

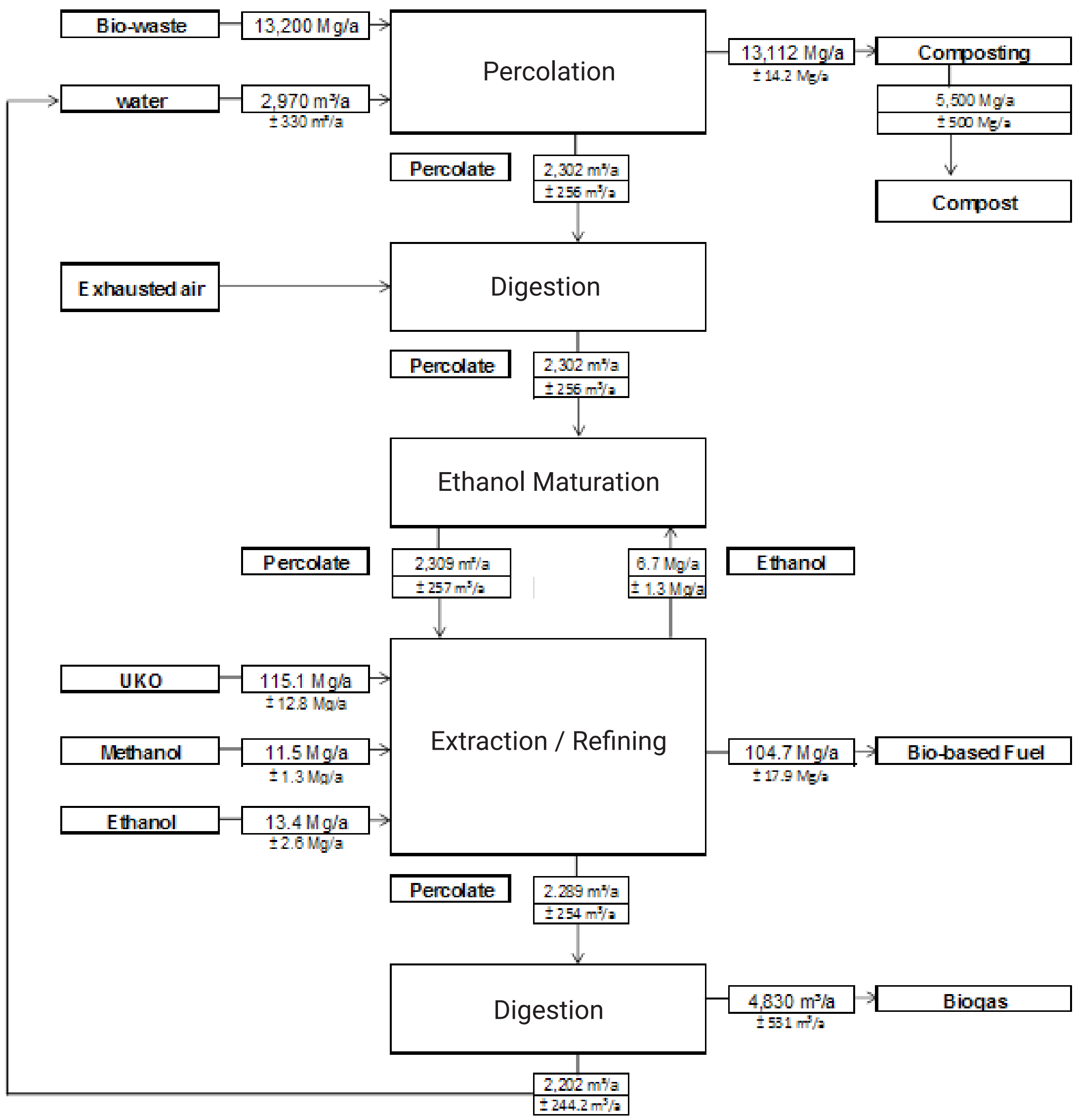


**FIGURE 7:** Estimated product quantities before technology optimisation .at the composting facility in Darmstadt [Kannengiesser, 2015].

### 3.5 Results of the technology optimisation

This chapter deals with the GC-MS analyses of the bio-waste liquid phase (the percolate), the leachate and the solvent phase (OME). Because of the solubility of the fatty acids in water, only acetate and octanoic acid were analysed in the bio-waste liquid phase.

To determine the fatty acids that were changed to the solvent phase, a new GC-MS method was developed. By using this new method, non-polar fatty acids from heptanoic to decanoic acid were analysed. The examined spectrum of the MCFA was enlarged to decanoic acid in order to find out if the chain elongation process stops after octanoic acid or not.

The titration of the solvent shows only the quantity of the acids, but not the quality. In order to examine the fatty acids that are in the solvent phase, the developed GC-MS method was used.

After the first GC-MS analyses, some conspicuities were noticed in the measurements, such as column bleed, carry over or asymmetric peaks. As a result of this, the detection of the fatty acids remained difficult. Assuming that the detection of fatty acids is better at high temperatures, the GC-MS column was changed from TG WAX MS A, which is a more polar column with a maximum temperature of 250°C, to a capillary type thermal column TR-5MS, which is non-polar, consists of 5% Phenyl methylpolysiloxane and can be heated to a maximum of 330/350°C.

The analysis method was optimised for better and specific detection of FAs. First, a temperature adjustment programme was carried out. The MS transfer-line temperature and MS ion source temperature were increased. By using the new column, which can withstand temperatures up to 350°C, a higher temperature was set.

After the separation of the fractions in the gas chromatograph at 260°C, the fragments were transported through the transfer line into the MS. The temperature should be as high as the starting one, so no cooling was considered at this stage. The mass spectrometer temperature was raised to 270°C.The results from the German liquid waste are shown in Figure 8.

The extractive digestion took about two weeks. In Figure 8, the acid concentrations at the beginning of the investigation and after two weeks are shown in total. The two samples were treated by the extractive digestion procedure, and as a result, both showed the same behaviour. The concentrations of the non-polar fatty acids that were dissolved in the extraction solvent increased within the two weeks. Heptanoic acid increased by about 49 wt%, octanoic acid by about 60 to 120 wt%, nonanoic acid by about 300 to 650 wt% and decanoic acid by about 100 to 300 wt%.

Unfortunately, based on the results at this point, certain estimations about the product quantities could not be made. More extraction tests still have to be done. At the moment, the presented data just shows some trends, which have to be proven by repeating the experiments.

The following direct comparison will show the differences between the old and the new treatment processes. In Figure 9, the extraction potential of fatty acids in the percolate using the punctual method is illustrated. It also shows the comparison between the availability of FAs in the percolate before the extraction and the FAs concentration in the OME.

In total, the extraction potential of the fatty acids in the percolate was about 21.800 mg. With the old method, as described above, about 3.720 mg of fatty acids can be extracted. Due to the polar property, the longer fatty acids had the highest extraction potential. Figure 10 shows the extraction of the FAs in the leachate. It also presents the results of the old extraction method.

The total amount of fatty acids in the leachate was about 27.700 mg, while about 4.720 mg can be extracted. In summary, around 17% of the FAs could be extracted with the punctual extraction. The comparison results of the two extraction methods showed that the new method, which was able to extract about 7.685 mg of fatty acid, has a 63% higher extraction rate than the old one. Figure 11 shows the extraction result using the new method. In this new extraction

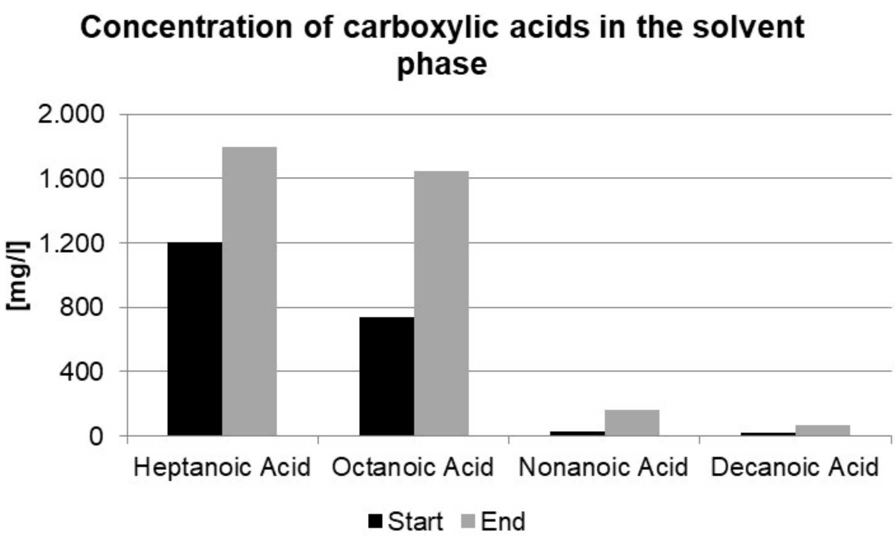


**FIGURE 8:** Concentration of carboxylic acids in the solvent phase from the African waste.

process, the extended measuring method was applied. The results show the extraction of additional amounts of the fatty acids from nonanoic and decanoid acid.

According to the extension of the residence time of the extractant, it is possible to enhance the amount of non-polar FAs. This is as a result of the chain elongation during the ethanol maturation. The extraction values are twice as high as those previously present in the liquid.

On average, the dissolved MCFA have 8 carbon atoms, and approx. 82 mmol MCFA were dissolved in one liter of

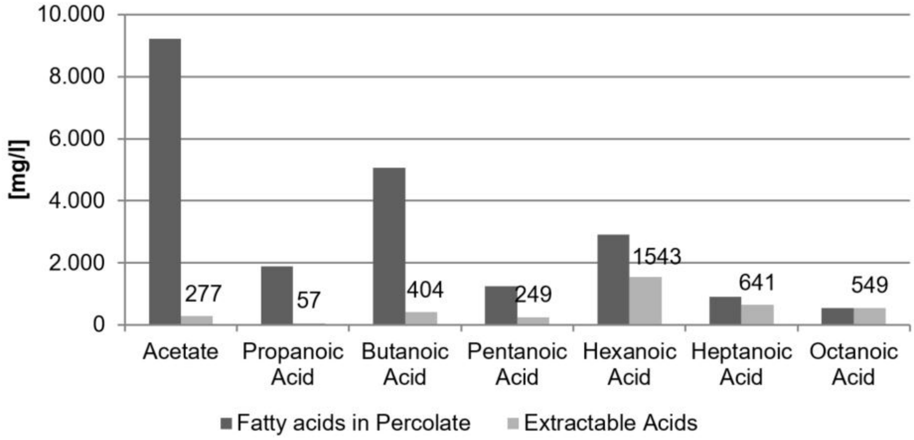


**FIGURE 9:** Fatty Acids and Extraction Potential from Percolate (old extraction method).

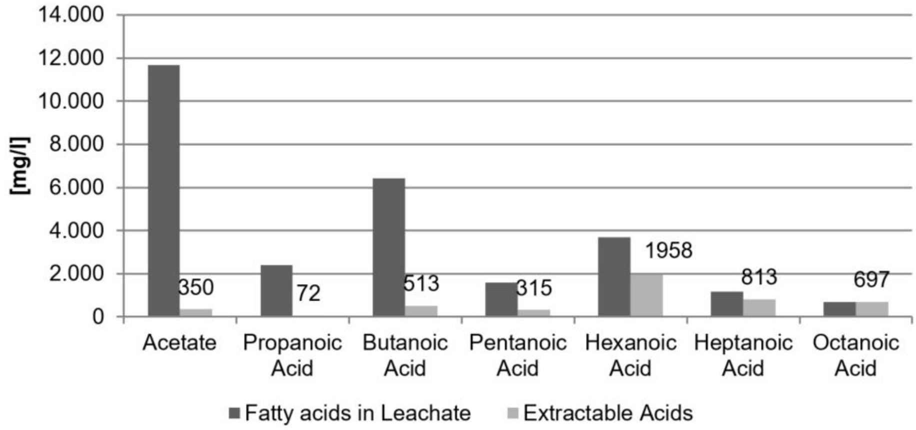


**FIGURE 10:** Fatty Acids and Extraction Potential from Leachate (old extraction method).

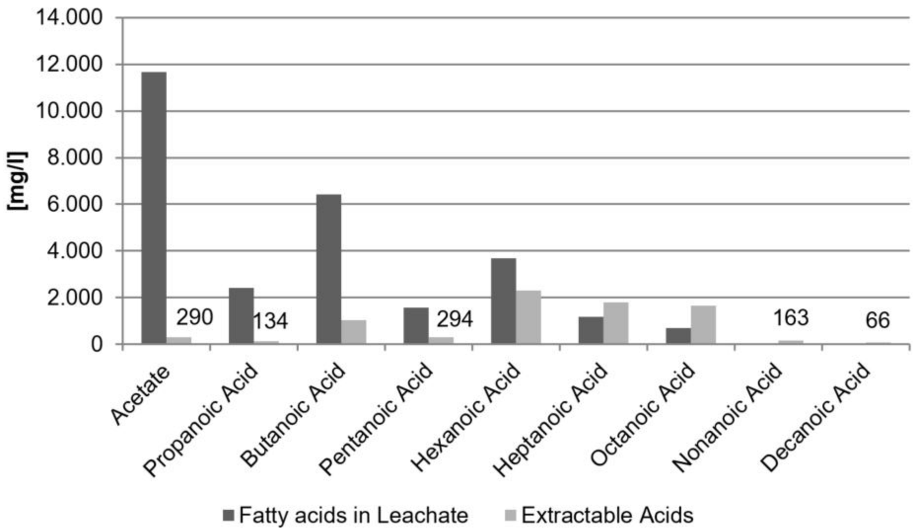


**FIGURE 11:** Fatty Acids and Extraction Potential from Leachate (new extraction method).

OME. The GC-MS analysis shows that MCFA with more than 8 carbon atoms were produced and extracted. This leads to the impression that the chain elongation process, described in Equation 1, does not stop with octanoic acid. The chain elongation process can also be used to produce longer chains of fatty acids, such as decanoic acid or even longer ones.

### 3.6 Comparison of the different liquid phases

Hereinafter, the first analysis results will be displayed. All liquid samples were analysed, and their fatty acid potentials were examined. The fresh pressed off water from organic waste will be shown. Apart from that, the liquids of the digestion and the leachate of different ages are introduced.

In Figure 12, the fatty acids concentration of the water from pressed organic waste is illustrated. In addition to the conditions at the start of the investigation, the concentration distributions at the end are also shown. It was observed that the fatty acids variation was limited to acetate and propionic acid at the beginning. After the maturing phase, longer-chain fatty acids were produced in higher quantities. This led to a reduction in acetate and an increase in butanoic acid due to the chain elongation. The rapid increase is partly due to the ethanol maturation of fatty acids. However, it should also be noted that this very fresh pressed off water has not been pre-treated and the biological decomposition took place over the storage period. During the storage of the substrates, the organic substances were degraded and the process of generation of the fatty acids began.

Figure 13 presents the different phases of the digestion plant. Based on the comparison of the different liquids' concentrations, a clear conclusion can be drawn. The percolate obtained after the pre-rotting process had the highest extraction potential. With around 11.000 mg/l acetate, it had a 24 times higher concentration than after the fermentation. After the pre-rotting in the facility, the condition of the liquid could be estimated as good. As a result, the material was not suitable for further processing after fermentation. The fatty acids were almost completely converted into methane and $CO_2$ during the anaerobic process.

Figure 14 shows the measured fatty acids concentrations in the leachate of the Moroccan waste. A distinction is made between the results after about 3 months of storage of the waste under partially anaerobic conditions and after a maturation period of about one year. The development of the fatty acid lengths can be seen here. A higher enrichment of around 60% butanoic acid was observed after one year, while acetates were reduced by almost 50%. A small increase was noted for octanoic acid. Due to the waste mixture and the incomplete anaerobic conditions, only a slow decomposition of the organic components occurred in the simulated landfill.

The leachate from German landfills has a very low organic potential. As described earlier, only a small amount of organic waste is deposited in Germany. With a pH value of about 7, it is in the neutral range. Also, the redox potential with positive values does not indicate a high organic activity. This is also confirmed by the general organic acid content of 278 mg/l. Compared to the values in Table 1, these are only 1% of the value measured in the percolate. Only small amounts of acetate and butanoic acid from 50 up to a maximum of 800 mg/l were generated. According to these results, even further tests for the maturation of the leachate show no potential.

## 4. CONCLUSIONS

The investigation of the different waste substrates indicates clear results. The process started with percolates from biological waste after anaerobic treatment. Subsequently, further processes were conducted to evaluate as well as implement the previously tested maturation and extraction method on other liquid waste substrates. In summary, it was noted that not all percolates from organic materials are well suited for this processing method. For example, the percolate from fresh waste must first be pre-treated in order to provide ideal conditions for the subsequent improvement in fatty acid generation.

However, it was established that the leachate from household waste was suitable for the experiments. The young landfill leachate extracted from simulated landfill sites also showed a high organic potential. The difference from organic household waste can be attributed primarily to the high proportion of kitchen waste. Short-chain carboxylic acids were present in higher quantities at the start of the experiments. Also, in this regard, ethanol maturation with subsequent extractive digestion was implemented after separation of the leachate.

## 5. OUTLOOK

In order to optimise this new bio-technology and integrate it into the existing biological waste treatment facilities, more investigations and analyses have to be done. The new treatment step, called extractive digestion, has a high potential to improve the quality and quantity of extractable non-polar fatty acids from bio-waste substrate. The first titrations and the GC-MS analyses show that the quantity of extracted non-polar fatty acids is much higher when extractive digestion is applied than without the use of this method. In order to perform some representative calculations, more analyses of the liquid phase and the solvent phase by GC-MS are needed.

A second important step is the optimisation of the bio-technology in order to extract the longest producible fatty acids (nonanoic and decanoic acid). The extraction of these fatty acids and the generation of bio-based products will make the productivity range of the facility higher and wider as well as make its $CO_2$ emissions lower.

In addition to these experiments, new tests should be carried out to examine the amount of solvent that is needed to extract the optimal amount of non-polar fatty acids from the liquid phase of bio-waste in the sense of sustainability (social, economic and ecologic conditions).

The experiments involved in extractive digestion were only done on the lab scale. In order to prove the practical use of this method in treatment facilities, tests in the relevant environment have to be carried out.

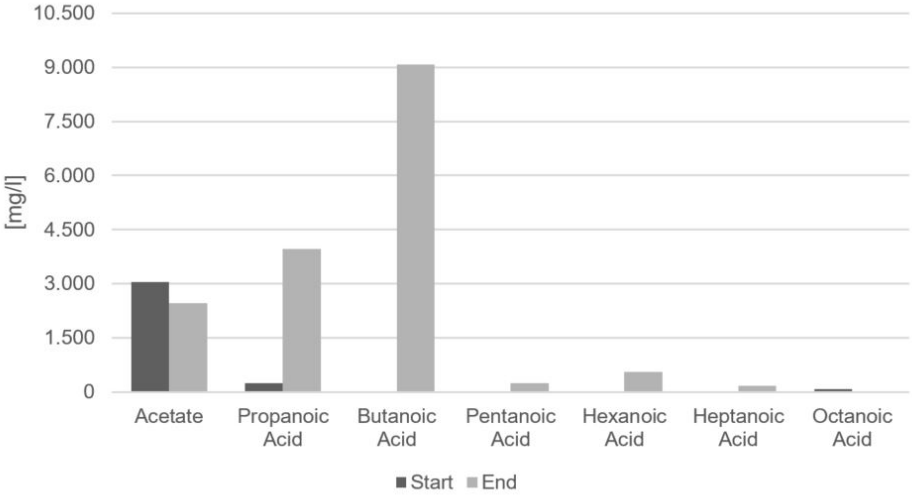


**FIGURE 12:** The comparison of fatty acids in the pressed off water samples at the start and end of the treatment process.

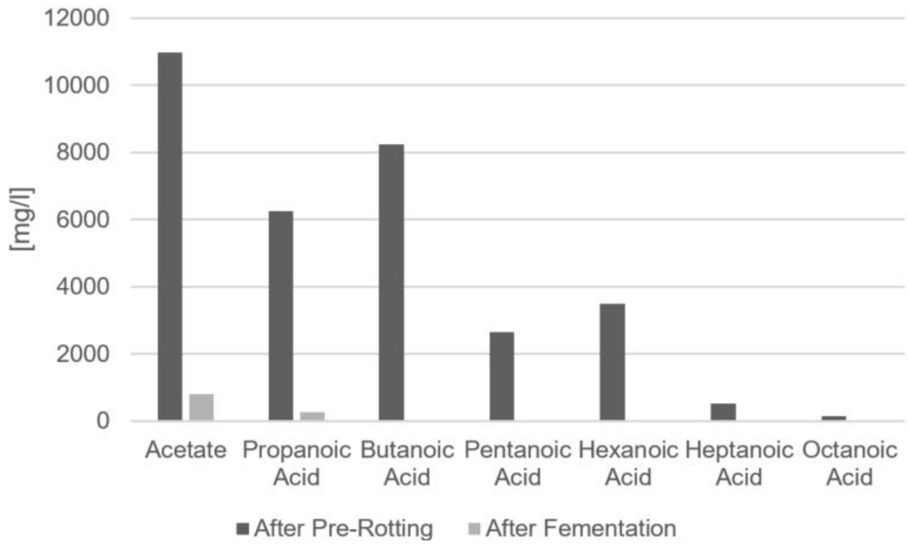


**FIGURE 13:** Fatty Acid concentration of different treated liquid samples from a digestion plant.

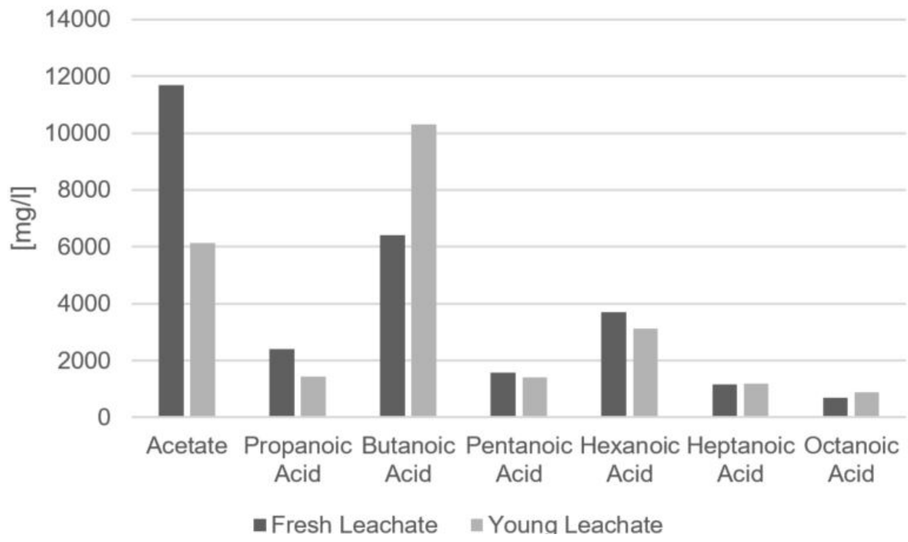


**FIGURE 14:** Fatty Acid concentration of the fresh and young leachate from the simulated Moroccan waste.

The experiments involving leachate from municipal waste dumps must also be further conducted. Old leachate from existing landfills must be investigated to determine its real potentials. The added value of MCFA should be determined using adapted examination conditions and maturing methods.

## AKNOWLEDGEMENTS

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# END OF SERVICE SCENARIO FOR UNIVERSITIES' INFORMATIC EQUIPMENT: RECOVERY AND REPAIR AS EDUCATIONAL AND RESEARCH TOOL FOR CIRCULAR ECONOMY AND URBAN MINING

Alessandra Bonoli [1], Nicoletta Dolci [1,2], Eleonora Foschi [1,*], Francesco Lalli [1], Daria Prandstraller [2] and Sara Zanni [1]

[1] *Department of Civil, Chemical, Environmental and Materials Engineering (DICAM), University of Bologna, via Terracini 28, 40131 Bologna, Italy*
[2] *Technical Unit for Waste Management (NU.TE.R), University of Bologna, via Filippo Re 10, 40126 Bologna, Italy*

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**ABSTRACT**
The urgent need to change unsustainable patterns of consumption, natural resources depletion rates, together with environmental impact and $CO_2$ emissions requires tangible initiatives that can accelerate the transition towards sustainable practices and provisions. Universities have the possibility to teach, operate and contribute to the improvement of global knowledge. They have the special responsibility of providing leadership on education and developing virtuous circles of "learning-by-doing" to demonstrate how to face the multiple challenges of sustainability. Considering electrical and electronic equipment (EEE) management at their end of service life, a Universities' laboratory can have a strategic role to promote circular economy and urban mining together with students' involvement in research and education pathways. In this paper, the case study is going to do an overview on RAEE generation at Italian University. Focusing on University of Bologna, an unusual solution has been provided by experimenting a new circular economy lab working, at first, on this type of products. In particular, the lab has investigated the possibility to valorize disused EEE thanks to restoration, preparing for reuse and refurbishment of obsolete informatics electrical and electronic equipment at the University of Bologna.

## 1. INTRODUCTION

Electronics has experienced exponential penetration in daily life. Everything, from product to services, is moving toward digitalisation, and people resorts to electronic devices with increasing frequency to quickly solve routine tasks. The acceleration on electronic devices consumption has been carried on by smart digitalisation that has also hastened the transition toward virtual communication system represented by information and communication technologies (ICT). For example, only in 2016, 1.5 billion units of smartphones were sold on the market compared to 680 million units sold in 2012. According to recent statistic, over 28 percent of the world population owned a smart communication device in 2016 (https://www.statista.com/statistics/263437/global-smartphone-sales-to-end-users-since-2007/). At the same time, digitalisation is influencing manufacturing and services industry taking advantage of open data system for research as well as treatments. Open and big data management needs high performance computing that has been contributing to implement technologically advanced multi-core processors, which require continuous maintenance and, often, replacement. The United Nations Environment Program (UNEP - 2009) assesses that by 2020 - the quantity of dismissed computers will increase 5 times over current levels. According to Cui and Forssberg (2003), the production of electrical and electronic equipment (EEE) is one of the fastest growing industrial sector. In 2016, this sector represented a world market share of 14.6%, with a remarkable growth accomplished in the short term and with good perspective in the long term (https://www.statista.com/statistics/263437/global-smartphone-sales-to-end-users-since-2007/). However, with the rapid advancement or progress in technology and the consequent fast obsolescence, consumer demand and strong incentives for consumption bring along a drastically reduced lifespan and increase faster replacement rates of most EEEs (Borthakur et al., 2017), with the consequence of increasing quantity of electronic waste, E-waste (Gu et al., 2016, Özkir et al., 2015). It follows that large stocks of materials are stored into cities, in buildings and infrastructure, where EEEs are present in large quantities. These

* Corresponding author:
Eleonora Foschi
email: eleonora.foschi3@unibo.it


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stocks may represent a large potential resource that would become available for the reuse at the end of the product lifetime (Brunner, 2011).

Electrical and electronic industries have the responsibility to cope with the challenge about increasing resource consumption by putting into practice innovative and sustainable processes and products based on design for disassembly, recyclability, etc. Considering these, urban mining becomes the new juncture for applying circular economy at urban level. This means that urban areas have to be rescheduled as sustainable districts applying circular models to close the loop of this huge stream of goods. In fact, the e-waste generation at a global scale in 2016 was around 44.7 million metric tons (Mt), i.e. 6.1 kg per person., and it is expected to grow to 52.2 Mt in 2021 (Baldé et al., 2017), with an annual growth rate of 3 to 5% (Agamuthu et al., 2015). The most urgent problem related to this trend is the management of toxic material included in electrical electronic equipment waste (WEEE) that may cause serious damage to the environment and have negative effects on human health. Hazardous materials typically involved in EEs are Lead (used in glass panels and gasket in computer monitors, solder in printed circuit boards and other components), Cadmium (used in chip resistors, infra-red detectors, semiconductor chips and battery), Mercury (used in thermostats, sensors, relays, switches) and Chromium VI (used for corrosion prevention of untreated and galvanized steel plates and as a decorative or hardener for steel housings) (Hagelüken, 2008). Consequently, landfilling, as end-of-life option for EEEs, not only strategically affects land occupation but also can cause air, water and soil pollution, while incineration may result in remarkable gaseous and particulate emissions. It is, therefore, crucial to manage the resulting WEEE properly in a sustainable development perspective (Gutiérrez et al., 2010). Currently, one third of European WEEE is being reported by compliance schemes as separately collected and appropriately managed (note that it might partially be accomplished via destinations outside the Member State of origin of EU). From 2016 onwards, the minimum collection rate shall be 45%, based on the total weight of WEEE collected in the Member State on a yearly basis, expressed as a percentage of the average weight of EEE placed on the market in the previous three years in the same Member State. The remaining WEEE is either 1) collected by unregistered enterprises and properly treated 2) collected by unregistered enterprises and improperly treated or even illegally exported abroad or 3) disposed of as part of residual waste (e.g. to landfills or incinerators). The problem gets worse if WEEE are illegally disposed (Li et al., 2012). The traditional illegal waste treatment is the disposal in open land, using primitive methods such as manual dismantling, open burning and acid leaching, rather than being properly extracted for reuse and recycling (Awasthi et al., 2016). According to a UNEP - United Nations Environment Program, a study on e-waste trafficking in 2013 reveals that most of the e-waste originating from the developed countries (European Union, the U.S., Japan, and Korea) has illegally destined to developing countries, especially India and China. The developed countries are also shipping out their used EEEs by incorrectly labelling them as electronic goods or as direct donations to institutions in developing countries (Yedla, 2016). However, the vision about electrical and electronical goods could change due to the presence of rich fraction of Copper, Gold, Nickel, Palladium and Silver in the EEEs (Bigum et al., 2012) and valuable bulky materials, such as Iron and Aluminum, along with plastic fractions, would definitely change. Overall, United Nations University estimates that the resource perspective for secondary raw materials from e-waste is worth 55 Billion € of raw materials (Baldé et al., 2017).

Restoration, refreshing, reuse and recycling actions can contribute to impact's reduction throughout the life cycle of the equipment, from the extraction of raw materials to the production and marketing. This is the approach proposed at the University of Bologna to manage the huge amount of disused EEE generated by the various research activities ongoing. Almost every research and teaching activity is driven by or depends on electronic devices, every general activity relies on computers and each laboratory, office, classroom has, at least, a computer. The increasing digitalization on research activities, especially on Artificial Intelligence (AI), mobile, social and Internet of Things (IoT) applications where the generation of huge quantities of digital data needs to be digitally captured, stored, and processed, leads to an higher demand for hardware, especially for storage and processing, such as high-performance servers, and obviously, on WEEE production (https://www.cbi.eu/market-information/electronics-electrical-engineering/trends/). Universities experienced a rapid growth in innovative EEEs and, at the same time, a quick increase in the number of obsolete equipment sent to disposal.

In 2012, Remedia, an Italian consortium promoting services for the integrated handling of end-of-life EEE, (batteries and accumulators) established at UNIRe a free program for collection and recycling of University WEEE. The estimation of WEEE flows in 2015, handled in the major Italian Universities and managed by Remedia, is reported in the Table 1. It is shown that a high amount of WEEE is produced in just a limited sample of Italian universities. Re-

**TABLE 1:** WEEE flows from Italian Universities to Remedia - 2015 - Source: Remedia

| Italian University | WEEE collected by Remedia (Kg) |
|---|---|
| Technical university of Milano | 33.055 |
| University of Milano | 29.100 |
| University of Bologna | 14.500 |
| University of Roma | 14.700 |
| University of Siena | 13.850 |
| University of Trieste | 13.640 |
| Technical university of Torino | 10.700 |
| University of Parma | 10.500 |
| University of Modena - Reggio Emilia | 7.150 |
| University of Catania | 6.900 |
| University of Napoli | 6.650 |
| University of Pavia | 3.900 |
| Others | 32.400 |

media managed about 14500 kg of University of Bologna WEEE.

WEEE are codified trough the European Waste Code (EWC), according to European Waste Catalogue and Hazardous Waste List (Environmental Protection Agency, 2002; European Commission, 2000), by defining the right code for the identification of product and process. EWC describes adequately the responsible management toward the proper end-of-life (EoL), i.e. waste being transported, handled or treated. The most common WEEEs generated at the Universities are hardware and computers, cathode ray tube (CRT) monitors, electric and electronic components, cables and refrigerators, which are respectively codified as 160214, 160213, 160216, 160211 EWCs. Table 2 shows Italian Universities' WEEE production in kg, for each EWC, collected by Remedia during the period 2012 – 2015.

The WEEEs outnumber all other waste fractions generated by the Universities or by a University, suggesting that Universities can be considered urban mines.

In fact, as it is universally recognized the fact that this kind of waste represents the main sector where an urban mining activity could be designed in order to maximize the life cycle of products and to exploit useful, precious and rare materials with a very high efficiency of separation.

## 2. CASE STUDY AT THE UNIVERSITY OF BOLOGNA: THE RAEEUSE LAB

The huge amount of WEEE disposed every day by the University of Bologna has suggested and motivated the opening of the rAEEuse lab in which trashware activities about restoration of IEEE are performed. The case study on a centralized management system to collect, prepare for reuse and recover most of the end-of-service life informatics equipment is carried out at the Department of civil, chemical, environmental and materials engineering (DICAM) in which education and research activities are carried on by powerful and advanced electrical and electronic equipment remains. The rAEEuse lab, planned in a bottom up approach, manages disused University IEEEs, pursuing research activities related to circular economy and digitalization. In accordance with EU priorities, the proper management goal of disused IEEEs is primarily to maximize the rate of reuse, trying to repair inoperative equipment or part of them. The project involves many economic, social and environmental benefits according to the university sustainability policy. The project design has been anticipated by a study of feasibility of the progressive adaptation of the laboratory at the total amount of waste managed by the University of Bologna.

The e-waste stream has been controlled and monitored since the 2012, year in which the UNIRe project was implemented. In Figure 1 WEEE streams, from 2012 to 2015, are reported.

Laptops and computers, together with cables and electrical and electronic components, are typically recyclable and cover more than a half of the entire stream of disused EEE (referred, when are waste, to the EWC 160214 and 160216).

The project is designed with the aim of extending the value chain of this type of IEEE maximizing their time span (Figure 2). Nowadays, the University management of IEEE is unfortunately set on the make-use-dispose model: a computer begins decrease its performance after three

**TABLE 2:** WEEE flows from Italian University to Remedia, for each EWC – 2012/2015 (kg/y) - Source: Remedia

| IWEE from University to Remedia | 2012/2013 | 2013/2014 | 2014/2015 |
|---|---|---|---|
| 160214 EWC (PCs, printers) | 142.375 Kg | 164.898 Kg | 135.351 Kg |
| 160213 EWC (CRT monitors) | 69.850 Kg | 104.162 Kg | 62.206 Kg |
| 160216 EWC (electric and electronic components and cables) | 1.336 Kg | 2.823 Kg | 3.724 Kg |
| Others | 5.082 Kg | 3.602 Kg | 16.646 Kg |
| **Total** | **218.373 Kg** | **275.485 Kg** | **217.927 Kg** |

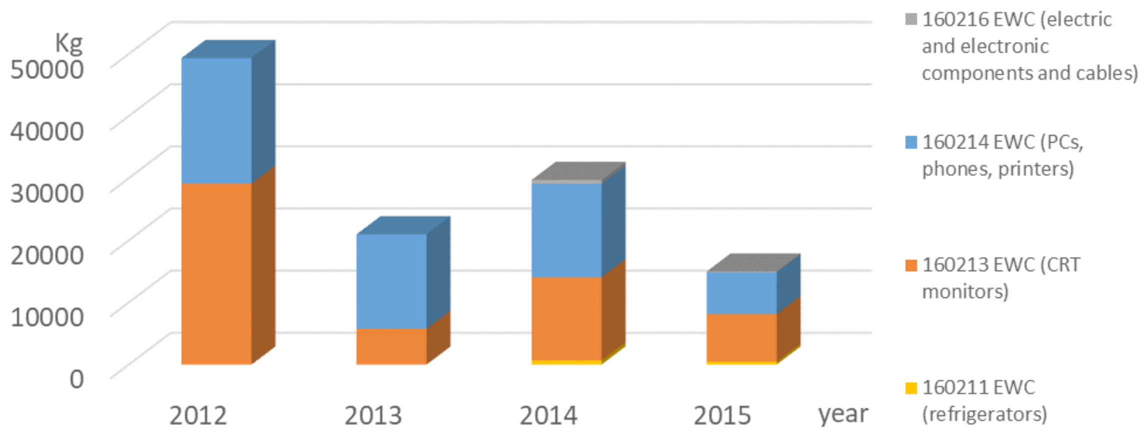


**FIGURE 1:** University of Bologna, WEEE stream (2012-2015).

years becoming incompatible with research activities, and so it is ready to disposal. The restoration activity allows the improving of performance giving to IEEE a second life.

A pilot test has been implemented to verify the feasibility setoff setting up the process, whose plan, shown in the Figure 3, illustrates the long-term strategy for university IEEE management.

More specifically, the rAEEuse lab manages disused equipment that in the past was directly sent to disposal. As shown in the Figure 4, all the input IEEE flow are analyzed through CPU-Z software that reveals the hardware components characteristics and performances. If the obsolescence does not permit the restoration, devices are dismantled manually, allowing to recover hard disk and other components. If IEEE are obsolete, materials are separated and unusable components collected. Once the reusable fraction is separated and/or repaired, the remaining flux is stored as proper waste and, due to the high quality and good collection and separation; it can be given to waste Management Company or consortium at very convenient economic conditions. Precious metals can be extracted from obsolete components thank to the collaboration with local SMEs that can take out any rare-earth and strategic metals (e.g. lanthanum, thulium and neodymium) through advanced metallurgical technologies (Işıldar, et al. 2018, Reuter et al., 2013; Tuncuk et al., 2018; Zhang et al., 2016; Graedel et al.,2011). Both regenerated PCs and recovered components can be reused within University or can be activated a virtuous process of donation to local community (no profit association, school, etc.).

The insight of the rAEEuse lab was born within a University course involving several engineering students. The feasibility was investigated by the University group "Terracini in Transizione", a living lab of students, researchers, academics and technicians, in collaboration with Nu.Te. R. (Nucleo Tecnico Rifiuti, i.e. the university waste technical unit). The method used for planning the activities in the lab is based on the transition thinking theory (Hopkins, 2009). The transition research refers to an interdisciplinary research field focused on structural change in societal systems (Wittmayer et al., 2017), with a multi-level perspective (MLP). In fact, the lab was carried on by students and researchers that, using a "bottom up" approach, have involved technicians and academics, pursuing a systematic

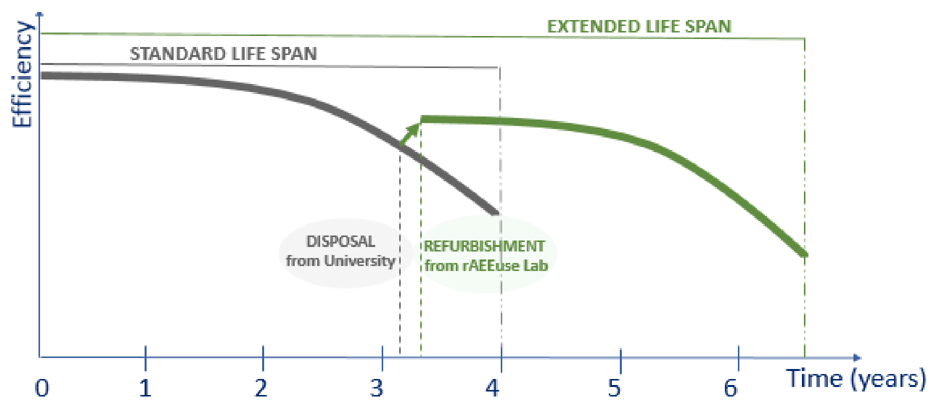


**FIGURE 2:** rAEEuse lab, the vision.

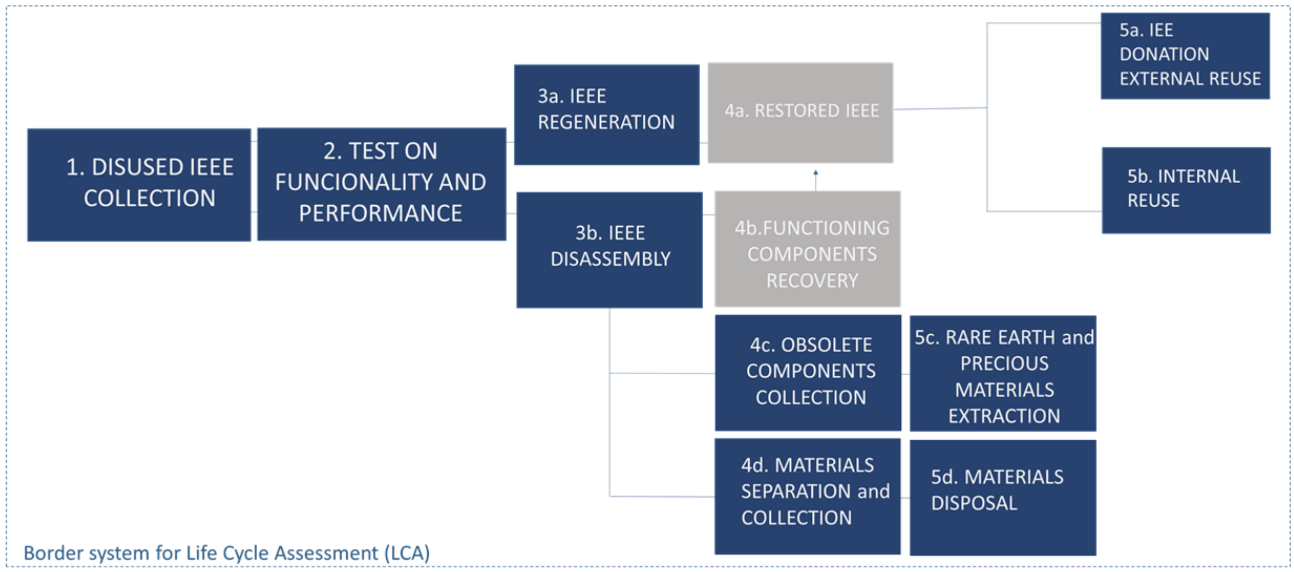


**FIGURE 3:** rAEEuse lab, the long-term strategy.

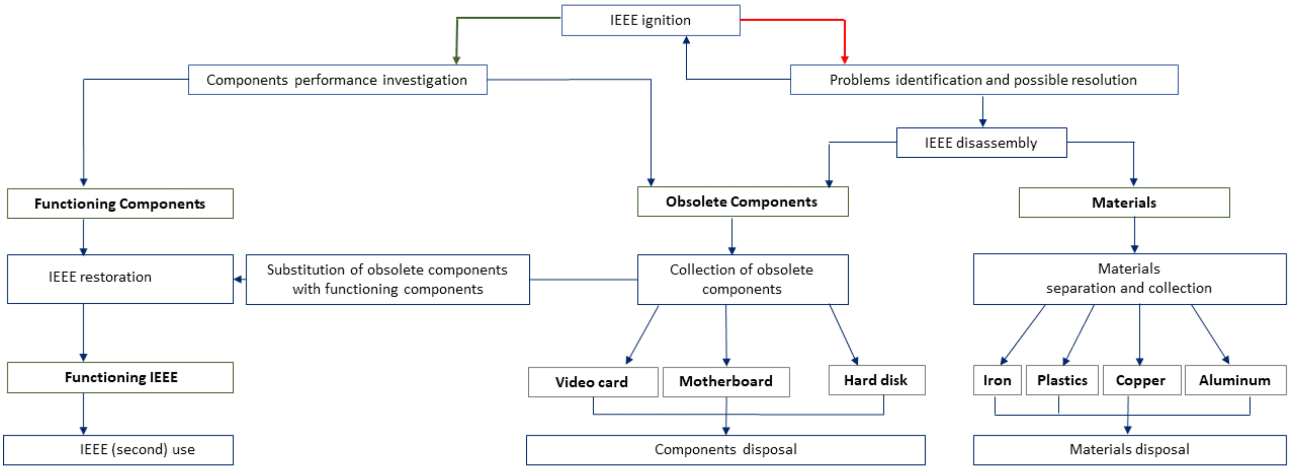


**FIGURE 4:** rAEEuse lab, the process.

innovation inside university, enabling the interactions at different levels and demonstrating then how different levels can be combined and can interact (Devolder et al., 2015).

## 3. RESULTS AND DISCUSSION

The experimentation has involved 19 computer facilities almost at the end of their service life by DICAM. Regarding these, 5 equipment have been restored while 14 have been disassembled. The restoration has been carried on by the replacement of Random Access Memory (RAM) with the most efficient ones. Instead, the disassembly has aimed to components and materials extraction. From a primary extraction operation, iron, plastic, aluminum have been manually recovered. An additional manual extraction has been done to get back copper from obsolete components and cables. All type of components and materials have been collected, separated and weighed to register data in a specific database. This database classifies all the available components (Motherboard, Processor, RAM, Video Card, power supply, Hard disk, Audio card, sink, Keyboard and Mouse) with a code, as shown in the Figure 5, thus identifying component types, properties and location in the lab. A dataset, containing detailed information on resources and processes, has included allowing the performing of a Life Cycle Assessment (LCA) and Life Cycle Cost (LCC) study. Database, supporting the outline of LCA and LCC studies, takes into account the assessment of duty and the responsibility in matter of privacy and the protection of data recorded inside the hard disks (Bonoli et al., 2013).

The average composition of an IEEE shown in Figure 6, reveals that, iron covers the bigger part of total composition of typical computers, followed by functioning and obsolete components. Taking into account the overall amount of components, the 55,7% wt is reusable, compared to the 44,3% wt that is obsolete.

While functioning components have been stored for future applications, the obsolete components have been collected to valorize printed circuit boards (PCBs). Their importance, from an economic and environmental point of view, relies in the metals present (high concentration and purity), and in the hazardous nature of some of its constituents (Choubey et al., 2015). In fact, PCB may contain about 250 g/t of gold, which has very high values compared to the gold ores with concentrations between 1 and 10 g/t (Tuncuk et al., 2012). However, the process to recover precious metals from EEE involves consists in pyrometallurgical, hydrometallurgical and bioprocesses, particularly different in terms of energy consumption.

The current challenges of rAEEuselab are related to the identification of sustainable separation and extraction processes to recover rare earth and precious metals from PCBs. In fact, while the pretreatment related to components extraction and materials collection, are manual processes, the metallurgical processes are much more difficult and impactful. LCA methodology permits to compare different operations in terms of environmental impacts and energy consumption and it can help decision-making processes about which End of Life (EoL) strategy is better to adopt for the critical resources extraction.

Considering the hydrometallurgical process performed by ENEA (Italian National Agency for New Technologies, Energy, Sustainable Economic Development) on end of life (EoL) household computers, results reveal a good percent-

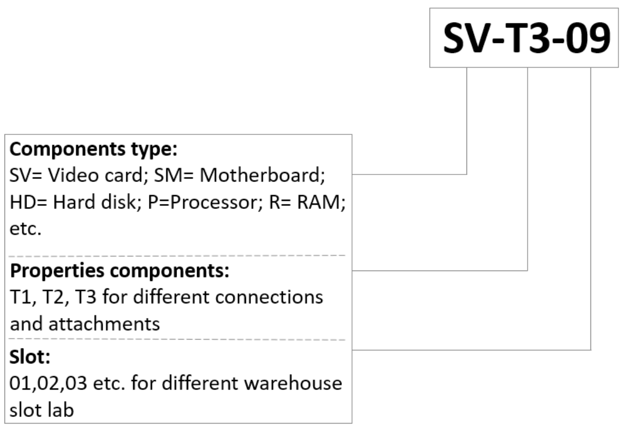


**FIGURE 5:** rAEEuse lab database, code example.

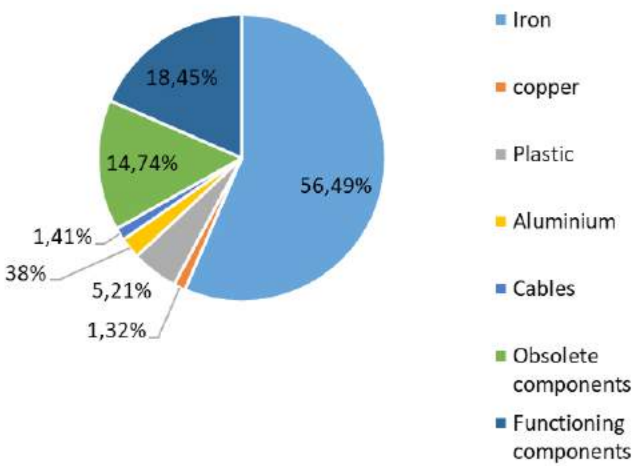


**FIGURE 6:** Average composition (by weight) of IEEE that has processed during the experimental stage in the rAEEuse lab.

age of metals recovery, as shown in the Table 3. A study on economic feasibility and profitability has been detailed, as shown in the Table 4.

In the past, disused EEE represented a problem due to the waste management and the high disposal costs, but today they can be considered as a source of resources, perfectly integrated with circular economy principles. In fact, before 2017, 19 IEEE were considered as waste to dispose, today, they have been managed as source to restoring 5 computers, valorizing precious metals and materials from 3 component restoration.

From the environmental point of view, the rAEEuse lab can be assumed consistent with the sustainable University policy. From the economic point of view, the fact that waste management costs have been reduced and revenues have been obtained by EE equipment and components valorization has added value to the lab. In addition to environmental and economic benefits, another important advantage is the disruptive system innovation going on at the University. The lab facilitates the university e-waste valorization and avoids the complexity, time and financial costs traditionally due to dispersive waste management, whilst ensuring that disused EEE is handled responsibly. In fact, while the procedure associated to waste disposal usually lasts many months owing to the hardworking practice related to University waste disposal according to the European waste directive, the average time to process a computer is about 15 minutes, thus permitting to cope with the overall amount of e-waste managed by the University of Bologna.

**TABLE 3:** Potential value of 1 Kg of PCB in October 2018.

| | Process recovery (g/Kg PCB) | Material value (EUR/g) |
|---|---|---|
| Ag | 0,35 | 0,4 |
| Au | 0,24 | 33,87 |
| Cu | 129 | 0,004 |
| Pb | 15 | 0,0017 |
| Sn | 43 | 0,019 |

**TABLE 4:** Potential revenue from obsolete EE components managed by rAEEuse lab during the experimental phase.

| Obsolete components (Kg) | | 2,09 |
|---|---|---|
| Materials | Material composition for obsolete components (g) | Estimated value for obsolete components (EUR) |
| Ag | 0,7315 | 0,293 |
| Au | 0,5016 | 16,989192 |
| Cu | 269,61 | 1,07844 |
| Pb | 31,35 | 0053295 |
| Sn | 89,87 | 1,70753 |
| Total revenue for obsolete components valorisation (EUR) | | 18,414 |

At the end, the lab has been putting into practices the so-called third mission of the University, generating public assets to the local community by donation directing the activities to a strong focus on socio-technological innovation in different socio-technical sub-systems or societal domains (Geels, 2002; Loorbach et al., 2010). However, this type of reality is already operational in the "trashware associations" throughout Italy and, in particular, at the University of Bologna at Cesena Campus, where a student association is funded by the municipality for refreshing disused computers. These practices could be extended to the city, starting by collaboration with companies and municipality that want to implement reuse and recycling of disused IEEEs. Moreover, the University of Bologna could potentially take advantage of its leadership inside the Italian universities for the sustainable development network (R.U.S.), to support other Universities to implement good practices for reuse and recycling of EEE but also to promote a clarification of ambiguous legislations.

## 4. CONCLUSIONS

The rAEEuse lab takes place in a favorable climate pushed by European regulation. The EU package on circular economy and its action plan (EU Commission, 2015; 2017), the policy recommendations like "European resource efficiency platform" (EREP, 2014), the European Cohesion Policy "Horizon 2014-2020" encourage the creation of dedicated paths for refurbishment, recovery and recycling of waste. This is also encouraged by the European WEEE regulatory (EU Directive 2012/19/UE), implemented in Italy in 2014 (D.L. 2014/49), where the new target of 45 kg of WEEE recovered is settled compared to 100 kg of EEE, placed on the market in the period 2016-2019. On the other side, this project, would take great advantage by a simpler and clearer local and national legislation, as well as many others in the field of circular economy.

In the University's context, the project represents an important contribution in the reduction of a remarkable quantity of EEE Waste (WEEE according with the prevention principles suggested by the Waste Hierarchy. The lab fits in the sustainable University policy and, is become an example of best practices of circular economy among all national Universities, having the potentiality to help to greatly reduce WEEE's environmental impact and to create a network of sustainability hotspots.

At the same time, the progressive broadening of this experience may contribute to reduce wider national and international problems, such as the large imports of metals and rare earths from China and South Africa and the illegal traffic of WEEE in developing countries. The appropriate recovery of disused EEE also helps to decrease the environmental damage caused by the improper disposal of them by the illegal international trade pathways. Therefore, limited action at local would help solving a big problem at a global scale, since each small action lays the foundation for the development of the transition.

It is also important to highlight that the Universities have the responsibility to provide leadership on education for sustainable development. They can apply a "learning-by-doing" approach in relation with the multiple challenges faced by the society and so they can influence actual and future decision makers. Education plays a crucial role especially in catalyzing proactive measures that can positively affect the present and the future. Due to its educational and institutional role, universities have an important role as research institution where innovative technologies can be developed, and sustainable operations can be put into practice (UNESCO, 2015). Universities can further facilitate the application of circular economy and sustainable development principles by a bottom up approach: not only through the activities of researchers and technicians, working to improve the efficiency of waste management and material substitutes to support resource productivity, but also engaging students as active participants in living lab initiatives like rAEEuse lab. In this way, they indirectly increase awareness about the importance of "closing-the-loop" and promote more sustainable lifestyles, as well as the transition to a low carbon future.

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# APPLICATION OF SUB-CRITICAL WATER FOR RECOVERY OF TIN AND GLASS SUBSTRATES FROM LCD PANEL E-WASTE

Hiroyuki Yoshida [1,2], Shamsul Izhar [1,2]*, Eiichiro Nishio [3], Yasuhiko Utsumi [3], Nobuaki Kakimori [3] and Salak Asghari Feridoun [2]

[1] Department of Chemical and Environmental Engineering, Faculty of Engineering, Universiti Putra Malaysia, 43400 UPM Serdang, Selangor, Malaysia
[2] Ecology Research Center, Research Organization for the 21st Century, Osaka Prefecture University, 1-1 Gakuen-Cho, Naka-ku, Sakai, Osaka 599-8570, Japan
[3] Environment Research and Development Center, Environmental Protection Group, Sharp Corporation, 1 Takumi-Cho, Sakai-ku, Sakai, Osaka 590-8522, Japan

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**ABSTRACT**

Tin and transparent glass substrate were efficiently recovered from color filter (CF) and thin-film transistor (TFT) glasses in LCD panel wastes using sub-critical water (sub-CW) at various treatment temperatures and reaction time. Treatment for 5 min using sub-CW added with NaOH resulted in a 95% recovery of tin from CF and TFT glasses. The tin oxide did not liquify in the liquid phase instead stayed in the organic multi-layers, which exfoliated together from the CF glass and TFT glass. This is a huge advantage because the organic multi-layers were readily separated by sub-CW and tin was recovered by filtration with ease. Transparent and clean glass was also recovered from LCD panel waste. The amount of tin oxide recovered depended on the sub-CW reaction temperature, reaction time and NaOH concentration. Treatment of smaller CF glass improved the recovery of tin. With this advance, we have showed that sub-CW method is technically feasible for tin oxide recovery in LCD waste.

## 1. INTRODUCTION

Liquid crystal display (LCD) panel consists of layers of polymer, thin-film transistor (TFT), liquid crystal, and color filter (CF) substrates. These layers contain complex circuits, electrodes, multiple organic layers and glass substrates as shown in Figure 1. The electrode used in LCDs comprises of indium tin oxide (ITO), a mixture of 90 wt% indium (III) oxide ($In_2O_3$) and 10 wt% tin (IV) oxide ($SnO_2$) (Alfatanzani et al., 2003). Thus, many studies have been carried out to recover valuable material from scrap LCD screens. Most studies have concentrated on hydrometallurgical methods (Hsieh et al., 2009) such as acid dissolution or acid leaching (Li et al., 2009; Li et al., 2011; Gabriel et al., 2017a; Gabriel et al., 2017b), solvent extraction (Fortes et al., 2003; Honma and Muratani, 2005) and chlorination (Park et al., 2009) methods. However, those studies require the need for wastewater treatment after recovering the metals. Furthermore, clean glass cannot be recovered because only a part of the metal is extracted while multilayers of polymer films, TFT, CF and liquid crystals are still stuck on the glass surface.

In the present work, sub-critical water (sub-CW) was employed. Sub-critical water has been developed for solid waste resources recovery and is gaining interest due to its potential as solvent and catalyst for organic reaction. This practice is based on the use of water as medium, at temperatures between its boiling point (100°C) and critical point (374°C) and at pressures higher than or equal to the pressure of saturated vapor pressure. At sub-critical conditions, the dielectric constant of water decreases, thereby lowering its polarity as depicted in Figure 2. Secondly, the magnitude of ionic product of water increases three orders around 250°C compared to room temperature. This will facilitate the dissociation of water making sub-CW advantageous for hydrolysis and decomposition of organic compounds including polymeric materials. Only a few papers have been published for metal recovery from multilayer films using sub-CW. Pure aluminum foil from waste composite laminates by decomposing plastic films by sub-CW hydrolysis reaction (Kulkarni et al., 2011). The plastics were also recovered as monomers.

Previously, a study revealed that 90% of indium was recovered from CF glass but only less than 7% indium from TFT glass (Yoshida et al., 2014). However, adding alkali improved the process where indium was totally removed and recovered from CF and TFT glass (Yoshida

* Corresponding author:
Shamsul Izhar
email: shamizhar@upm.edu.my


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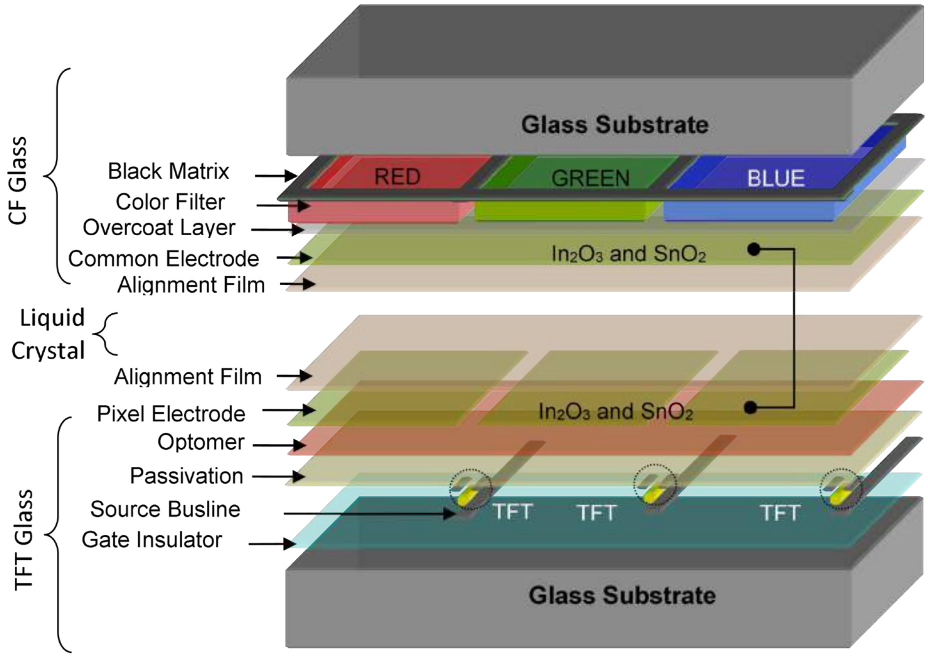


**FIGURE 1:** Detail cross section layout of a single pixel in a common liquid crystal display panel.

et al., 2015). However, the recoveries of other crucial elements, such as $SnO_2$ and high-quality LCD glass substrate was not considered. Thus, in the present work, we investigated tin and glass substrate recovery using sub-CW. In addition, the recovery of clean and transparent glass was carried out by changing the sample sizes before sub-CW treatment.

## 2. MATERIALS AND EXPERIMENTAL PROCEDURE

### 2.1 Material preparation and sub-critical water treatment

Preparation of raw material and sub-CW treatment were referred to Yoshida et al (2015). The waste LCD panel was a 40-inch TFT type (Sharp Corp.), composed of a polarizer, liquid crystals and arrangement of TFT and CF glasses. The polarizing films on the outer sides of the LCD were stripped off and separated into CF glass and TFT array by cutting the panel edge using a glass cutter. Liquid crystals on both glass surfaces were washed away by acetone. Each glass was crushed into 5-10 mm size, small enough to fit the reactor. The reactor as shown in Figure 3 was a stainless-steel tube with end-caps (Swagelok) were fitted to both ends of the tube (16 mm internal diameter, 150 mm length). Pure water or NaOH aqueous solution with volume approximately 20 mL together with 6 g of either CF glass or TFT glass was filled into the reactor. The air in the reactor was replaced with argon gas before the reactor was sealed and weighed.

For the sub-CW treatment, the reactor was immersed quickly in a preheated molten salt bath (Tomasu Kagaku, Celsius 600) at a desired temperature. After the desired time was reached, the tube was taken out and immediately cooled at room temperature. The pressure inside the reactor was the saturated vapor pressure of water at the subsequent reaction temperature. The reaction time was defined as the time from immersing the reactor into the salt bath until before immediate cooling. The sample was then transferred from the tube to a beaker. The solid residues (substrate glass) were separated from the aqueous phase by means of vacuum filtration through a 1 µm pore-sized membrane filter (Advantec, cellulose acetate). The filters were dried at room temperature and weigh.

### 2.2 Measurement and analysis

After sub-CW treatment, concentration of metal in each medium was determined. Tin was extracted using 7% hydrochloric acid solution from the remaining glass and membrane filter according to equation (1).

$$SnO_2 + 2HCl \rightarrow SnCl_4 + H_2O \quad (1)$$

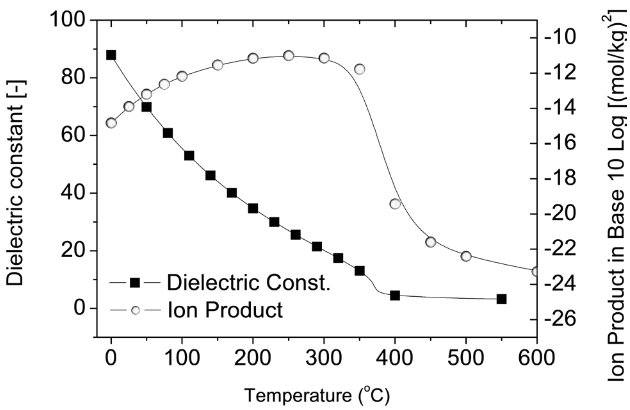


**FIGURE 2:** Dielectric constant and ionic products of water at sub-critical and supercritical state (Wolfgang and Kretzschmar, 2008).

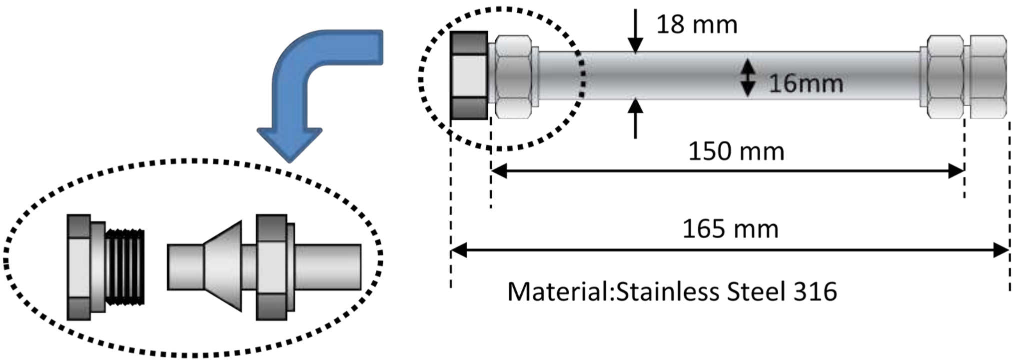


**FIGURE 3:** Layout of the reactor used for subcritical water treatment.

The tin concentrations, defined as mass of tin for every kg of LCD glass [mg/kg-LCD] in the solution extracted from the remaining glass substrate ($W_g$), filter ($W_f$) and liquid-phase ($W_l$) were measured using a plasma atomic emission spectroscopy ICP (Shimadzu, ICPE-9000). The mass balance of fresh tin (Winit) before the reaction and after the reaction ($W_g$, $W_f$ and $W_l$) was verified. The recovery of tin ($\eta_{Sn}$) was defined by the amount of tin that was filtered ($W_f$) to the amount of tin in the fresh LCD ($W_{init}$):

$$\eta_{Sn} = \frac{W_f}{W_{init}} \times 100 \quad (2)$$

## 3. RESULTS AND DISCUSSION

### 3.1 The concentrations of tin on CF and TFT glasses before sub-CW reaction

To evaluate the amount of tin recovery, the total amount of tin on each fresh CF and TFT glasses were determined by extracting tin using 7% hydrochloric acid aqueous solution. Tin quantity in fresh CF glass and TFT glass ($W_{init}$) were 36 and 28 mg/kg-LCD, respectively. The tin in TFT glass is 80% of the CF glass due to the sensitivity of the TFTs towards backlight illumination during application (Lee et al., 2008).

### 3.2 Sub-CW reaction in CF and TFT glasses (without NaOH)

TFT and CF glasses were treated by sub-CW without NaOH for 5 min reaction time at different reaction temperatures. The amounts of tin dissolved in the liquid-phase ($W_l$), in the organic multi-layers on the membrane filter ($W_f$) and in the residues on the glass surface ($W_g$) after the reaction was detected by the same method as the untreated fresh TFT and CF glasses as mentioned is section 3.1. Figure 4 shows the results from CF glass treated with sub-critical water for 5 min reaction time at various temperatures. Tin was not detected in the liquid-phase ($W_l$) when reaction temperature was lower than 360°C. $W_g$ reduced steadily with increasing reaction temperature beginning from 220°C to 360°C. At supercritical state (400°C), $W_g$ reduced to 4 mg/kg-LCD. Identically, $W_f$ showed the opposite trend to $W_g$ where $W_f$ increased with increasing treatment temperature. At supercritical state (400°C), $W_f$ was 31 mg/kg-LCD. Higher treatment temperature resulted in a more tin exfoliation from the LCD surface, although small amount of tin was detected in the liquid phase ($W_l$) when temperature was higher than 360°C. This is because the exfoliated multilayer organics, in which tin oxide was sandwiched, decomposed at very high temperature and tin oxide exposed to sub-CW.

Figure 5 demonstrates the results in TFT glass treated with sub-critical water for 5 min reaction time at various temperatures. Tin was not discovered at all in $W_l$. Treatment below 260°C and above 340°C showed no drop of $W_g$

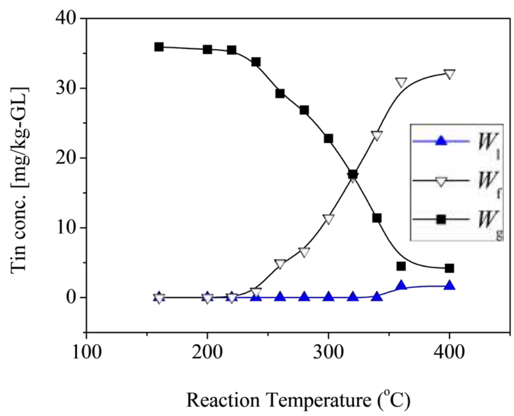


**FIGURE 4:** Tin distribution in filter ($W_f$), liquid phase ($W_l$) and glass ($W_g$) after sub-CW treatment of CF glass for 5 min reaction time at various temperatures.

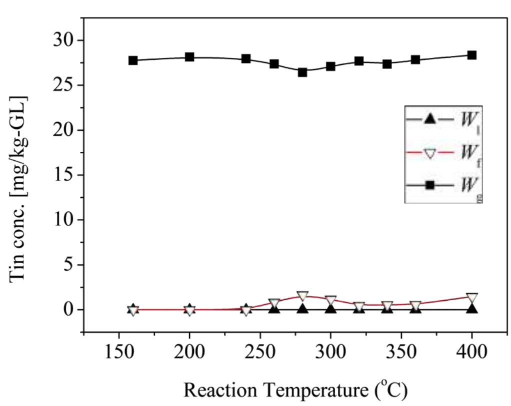


**FIGURE 5:** Tin distribution in filter ($W_f$), liquid phase ($W_l$) and glass ($W_g$) after sub-CW treatment of TFT glass without NaOH for 5 min reaction time at various temperatures.

suggesting no tin was removed from TFT glass. At around 280°C, $W_g$ was 26 mg/kg-LCD, demonstrating small amount of tin was removed. This could be attributed to hydrolysis, because at this temperature the ionic product is high as indicated in Figure 2, suggesting hydrolysis is active. At 400°C, $W_g$ was 28 mg/kg-LCD indicating greater part of the tin remained on TFT glass. These results suggest that multilayer organics which sandwiched tin oxide were difficult to be exfoliated by sub-CW and even by super critical water. TFT layers are usually covered with a thin alignment layer (~80-100 nm thickness), with a function to insulate the optomer layer and to protect the passivation layers (Lee et al., 2008). The optomer film shown in Figure 1 is an organic mixture consisting of mostly diethylene glycol methyl ethyl ether and acrylic resin. The passivation layer wrapping the drain and source terminals are made of silicon nitride ($SiN_x$). This implied that the optomer and passivation layers on the TFT glass were insufficiently decomposed by treatment with only sub-CW and supercritical water. However, it is worth noting that $W_l$ was near zero in every circumstance. Consequently, the results in Figures 4 and 5 showed that tin did not dissolve in the liquid-phase but was appended with the organic multilayers and were removed together from the CF and TFT glasses.

### 3.3 Tin recovery from CF and TFT glasses with Sub-CW added with NaOH

To overcome the insufficient decomposition, NaOH was added in sub-CW. Figure 6 illustrates the effect of NaOH concentration in Sub-CW on tin recovery $\eta_{Sn}$ from TFT and CF glasses. The reaction time was 5 min. For TFT glass, at 220°C, almost no tin was recovered when treatment was performed using less than 0.03 M NaOH. However, when treated with concentrations above 0.1 M, the amount of tin recovered was more than 80% for TFT glass. This suggests that concentration of NaOH in sub-CW is important for decomposing the organic multilayers in CF and TFT glasses.

In CF glass at 160°C, a similar result with TFT glass at 220°C was observed. But maximum recovery was more than 90% when NaOH concentration was higher than 0.2 M. Then a slightly higher reaction temperature (180°C) for CF glass was examined. The recovery of tin increased substantially, and 90% tin recovery were obtained in 0.1 M NaOH concentration.

Figure 7 demonstrates the effect of reaction temperature on the recovery of tin $\eta_{Sn}$ by sub-CW treatment for 5 min with water-only and with 0.1M NaOH added. For CF glass (Figure 7A), $\eta_{Sn}$ with water-only treatment was found to be inactive below 240°C. The $\eta_{Sn}$ increased from none at 240°C to 80% at 360°C. $\eta_{Sn}$ remained 80% until 400°C. These results exhibited that treatment temperature directly affects tin exfoliation from CF glass. However, when 0.1 M NaOH was present, the tin recovery was 45% even at 100°C. This suggests that hydrolysis process was probably active and had already occurred at water boiling temperature. The tin recovery increased to 95% at 160°C and remained about 90% at 360°C. With the presence of NaOH, $SiN_x$ from the passivation layer has dissolved into the liquid-phase as a result of the exfoliation of the organic multi-layers. Addition of NaOH probably caused corrosion of the substrate glass in NaOH environment causing the formation of sodium silicate. This showed the significant effect of the presence of NaOH to the tin recovery from CF glass.

In Figure 7B, for TFT glass treated by sub-CW without NaOH, $\eta_{Sn}$ gradually increased from none at 240°C to 6% peak at 280°C. However, when 0.1 M NaOH existed in sub-CW, $\eta_{Sn}$ increased drastically with reaction temperature. When the temperature was higher than 160°C, $\eta_{Sn}$ increased to 95% at 220°C. The $\eta$ was around 93% between 220 and 340°C. These results implied that the decomposition of the passivation and optomer layers on the TFT glass was enhanced significantly by the presence of NaOH in sub-CW. However, at 360°C the $\eta_{Sn}$ slightly dropped to 70%. A previous study by Yoshida et al., has shown that high content of Si ion in $W_l$ caused of the formation of sodium silicate at

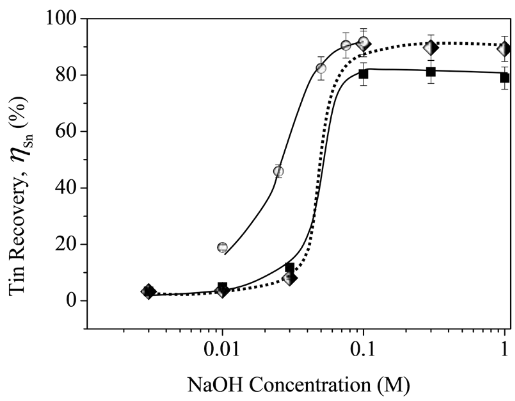


**FIGURE 6:** Effect of NaOH concentration in Sub-CW on the recovery of tin (5 min of reaction time). (□) CF glass treated at 160°C, (Δ) CF glass treated at 180°C and (■) TFT glass treated at 220°C.

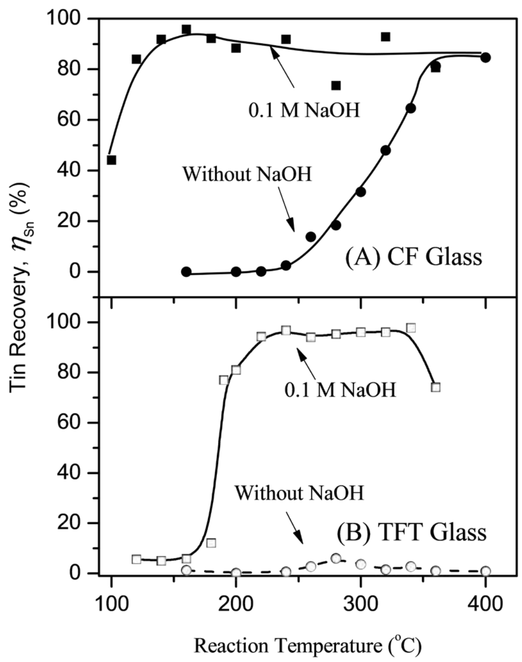


**FIGURE 7:** Effect of sub-CW reaction temperature on the recovery of tin ($\eta$) with 0.1N NaOH added and without NaOH (reaction time was 5 min).

very high temperature. During ICP analysis of $W_l$, due to the adjacent wavelength of tin and silica, the high intensity of silica interfered with tin causing the appearance of tin in $W_l$. Thus, this resulted in the small reduction of $\eta_{Sn}$.

Figure 8 illustrates the time course of the recovery of tin $\eta_{Sn}$ from CF glass during sub-CW treatment with (a) 0.1 M NaOH at 160°C, (b) 0.05 M NaOH at 180°C, and 0.1 M NaOH at 360°C. The three curves were close to each other until 30 min. They showed the reactions were rapid because after only 5 min, $\eta_{Sn}$ immediately became 80-90%. The reaction time of 5-15 min showed maximum recovery at 91-95%. After 30 min, curve (a) was close to curve (b) and $\eta_{Sn}$ in both (a) and (b) slightly decreased with time. However, $\eta_{Sn}$ in (c) decreased significantly with time. When reaction time was extended, the exfoliated multilayer organics that sandwiched $SnO_2$ were decomposed in small amount for (a) 160°C and (b) 180°C, but in large portion for (c) 360°C. The drop of $\eta_{Sn}$ at 360°C is probably due to the difficulty to discriminate the intensity of silica and tin when analyzing the ICP results.

Figure 8 illustrates also the tin recovery from TFT glass by treatment with 0.1 M NaOH at 220°C. The $\eta_{Sn}$ was 75% when treatment was performed for 2 min, but was maximum at 80% when treated for 5 min. The $\eta_{Sn}$ did not go above 80% when treated above 5 min. Thus, when the concentration of NaOH was 0.1 M and reaction time was 5-15 min, maximum recovery of tin from CF and TFT glasses was attained. Most leaching using acid solution was able recover 89% to 99% of tin in the form of ITO as reviewed in the literature (Ueberschaar et al, 2017). The result in the present study using subcritical water showed a 80 to 95% recovery from CF and TFT glasses, which is comparable to the hydrometallurgical methods. Furthermore, only a short time of 5 min is required thus making subcritical water method a promising technology and excellent alternative to the hydrometallurgical method.

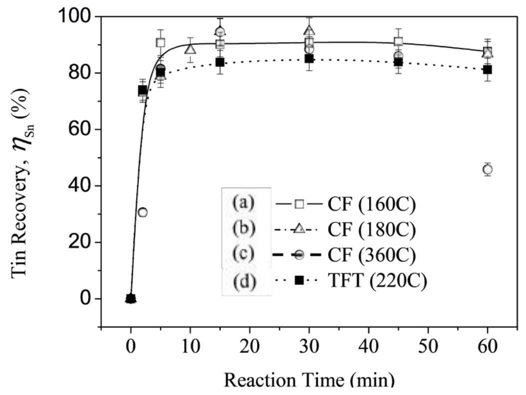


**FIGURE 8:** Time course of tin recovery from CF glass by sub-CW treatment at various reaction time: a) 0.1 M NaOH at 160°C, (b) 0.05 M NaOH at 180°C, (c) 0.1 M NaOH at 360°C, and from TFT glass (d) 0.1 M NaOH at 220°C.

### 3.4 The effect of sizes of CF glass on sub-CW reaction

Figure 9 illustrates the effect of glass size on the remaining color of CF glass after sub-CW treatment with 0.1 M NaOH for 5 min at 180°C. The CF glass was cut using a glass cutter into about 20 mm and crushed to about 5 mm length. After the sub-CW treatment, the larger size glass still has color remained from color filter pigment. While the smaller sized glass became completely clear of any pigment color. This showed that smaller sized glass which consisted of large amount of surface area effected the removal of CF pigments. As for the large sized glass, the color remained at the middle of the glass surface, indicating that as the surfaces of the glasses adhered each other, sub-CW contacted slightly with the surface but probably penetrated through the side of the glasses adhered.

Table 1 indicates the recovery of tin from CF glass cut into 5-10 mm, 20 mm and crushed to 5 mm length. Treat-

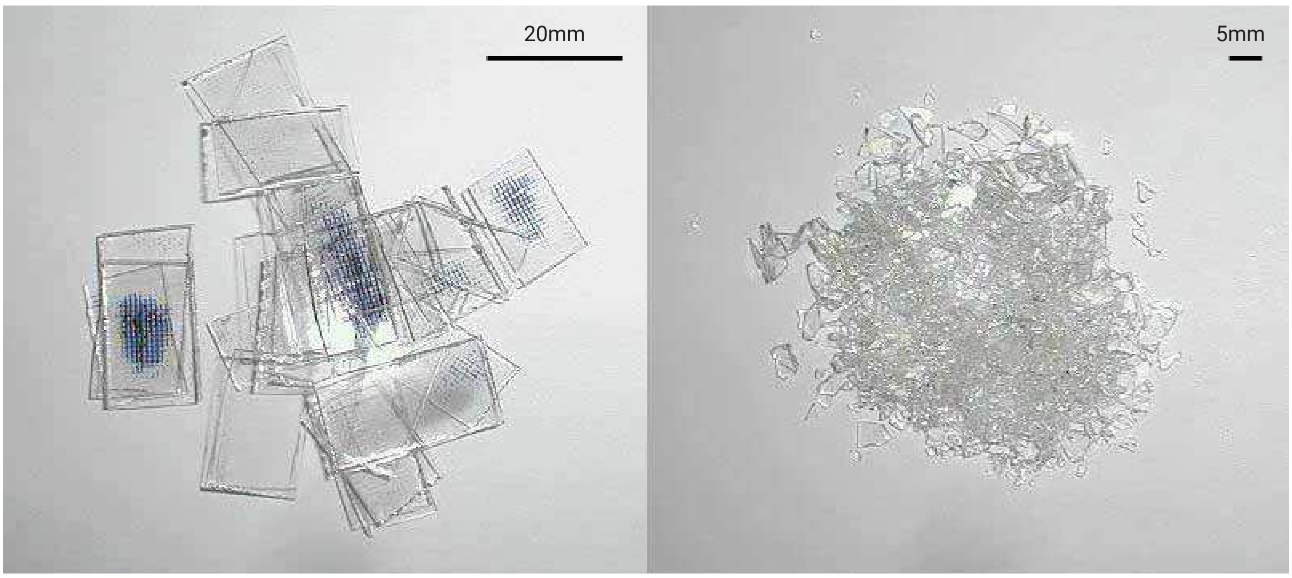


**FIGURE 9:** . The effect of glass size on the remaining color of glass after sub-CW treatment of CF glass with 0.1 M NaOH solution. Reaction time and temperature were 5 min and 180°C, respectively.

**TABLE 1:** Effect of treated CF glass size on tin recoveries when they were treated with sub-CW added 0.1 M NaOH at 180°C for 5 min.

| Treated CF glass size | Tin Recovery [%] |
|---|---|
| 5 mm | 92 |
| 5-10 mm | 92 |
| 20 mm | 90 |

ment was carried out using 0.1 M NaOH at 180°C for 5 min. The recovery of tin $\eta_{Sn}$ in the 5 mm and 5-10 mm CF glasses were 92%, just slightly 2% higher than the 20-mm sized CF glass. Since the smaller 5 mm sized CF glass gave only 2% better recovery of tin than the larger 20 mm-sized CF glass, the effect of tin recovery is not significant. Thus, glass breaking process would be sufficient enough than crushing process since breaking process might require less energy consumption .

In this study, high purity glass substrates were recovered from CF glass, in addition to tin. The glasses are superior value substrate products because it combines Corning's fusion process and optical melting technology for its production. The glass substrates recovered from CF glass by sub-CW has smooth surface and transparent, therefore require no repolishing upon reuse. Thus, by recovery of tin and glass substrates simultaneously, it is certain that the sub-CW technology is highly economical. Furthermore, tin is readily separated just by filtration. This could probably lead to the lowest operational cost compared to the other methods that require chemical post-treatments to separate and treat the chemical residues after the recovery.

## 4. CONCLUSIONS

The recovery of tin and high-quality glass substrate from LCD panel waste was technically feasible by NaOH added sub-CW treatment. Detail conclusions are given as follows:

1. Treatment for 5 min using sub-critical water without NaOH added showed a maximum of 80% and 6% tin recovery from CF glass and TFT glass, respectively. However, when NaOH was added (0.1 N), 95% of both CF glass at 160°C and TFT glasses at 220°C was recovered. More, tin did not dissolve in the liquid-phase but was appended with the organic multilayers. The multilayers were removed together from the CF and TFT glasses. This phenomenon is very beneficial because tin in the form of ITO was readily separated from CF and TFT glasses, and easily recuperated on the filter.
2. The reaction temperature 160-180°C in CF glass and 220°C in TFT glass, NaOH concentration 0.05-0.1N, and reaction time 5-15 min were enough to get maximum recovery of tin ($\eta$ = 91-95%).

More tin was recovered, and the glass substrate became completely clean and transparent when sub-CW treatment was performed on smaller glass size (5 mm).

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# CHARACTERIZATION OF PLASTIC MATERIALS PRESENT IN MUNICIPAL SOLID WASTE: PRELIMINARY STUDY FOR THEIR MECHANICAL RECYCLING

Mónica Calero [1], María Ángeles Martín-Lara [1], Verónica Godoy [1,*], Lucía Quesada [1], David Martínez [2], Francisco Peula [2] and José Manuel Soto [2]

[1] *Department of Chemical Engineering, University of Granada, Avenida Fuentenueva s/n, 18071, Spain*
[2] *INGESIA Ingeniería y Medio Ambiente S.L., C/Parque de las Ciencias, 1, 18014, Granada, Spain*

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**ABSTRACT**

In the EU, 25.8 million tons of plastic wastes are generated each year and more than 30% end up in landfills. In Spain, this percentage rises up to 50%. Mechanical recycling is currently one of the best alternatives to reduce problems associated with poor management of plastic waste. In this paper, an analytical laboratory study of several samples of municipal plastic waste from Granada (Spain) was presented. The samples were supplied by the Waste Treatment Plant (Ecocentral). The study was based on the measurement of the moisture and dirt content of the selected plastic waste. Those parameters were determined by washing and drying the waste and analyzing the washing wastewater; in order to determine/justify the need of a washing step and a post-treatment of the washing water. The results showed that the differences in moisture and dirt content were significant between the different types of polymers, which could influence in the economic profitability of mechanical recycling. Polystyrene (PS) is the material that loosed the most weight while polypropylene (PP) loosed least weight. Moreover, the washing wastewater shows parameters that comply with the discharge regulations of Granada (Spain), except for the case of polyethylene film, whose wastewater would require pre-treatment prior to discharge. Overall, the results were satisfactory, as they show that most of the ordinary plastic waste can be recycled without high cost.

## 1. INTRODUCTION

Since the Second World War, plastic has been established as one of the essential materials in many areas of everyday life, gaining strength in sectors such as automotive industry, clothing and decoration. From the 1960s to the present day, the demand for plastic products has grown continously. The annual production of plastic has increased twenty-fold in the last fifty years. Among the most commonly used polymers today, different typologies can be cited (Table 1).

One of the major environmental problems is the large amount of plastic waste generated. Both, the production of this material and the incorrect waste management cause several environmental problems. One of those problems is the amount of oil needed to manufacture virgin polymers (up to 6% of world oil production). Other problems are greenhouse gas emissions during manufacture (more than 1% of the world total), low recycling rates of waste and dumping of this waste at sea (World Economic Forum et al., 2016). It is estimated that 80% of the waste present in seas are plastics which come from land (Rojo-Nieto and Montoto, 2017). The problem is that it takes between 100 and 1000 years to degrade plastics, so they suppose a real threat for sea flora and fauna.

In 2015, 322 million tons of plastic were produced worldwide. Europe is the second largest producer behind China. Its production reached 58 million tons that same year. Of this amount, 25.8 million tons were introduced annually into the municipal waste stream, which supposed a 12.4% of the total municipal solid waste (MSW). A 30.8% of this plastic waste was deposited in landfills, 39.5% was utilized for energy recovery and 29.5% was recycled (PlasticsEurope, 2016). In Spain the situation is even worse than in the EU. In fact, the amount of plastic waste that ends up in landfills is over 50% in Spain (PlasticsEurope, 2015).

The statistics show that the production of plastics and the plastic waste recycling do not grow in the same way. Increasing the percentages of mechanical recycling would be

* Corresponding author:
Verónica Godoy
email: vgcalero13@gmail.com

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**TABLE 1:** Most commonly used polymers.

| Society of the Plastics Industry (SPI) Code | Polymer | Applications |
|---|---|---|
| 1 PET | Polyethylene terephtalate | Food packaging, carbonated soft drink bottles, water bottles, oil bottles, etc. |
| 2 PEAD | High density polyethylene | Bags, detergent bottles, dairy bottles, etc. |
| 3 PVC | Polyvinyl chloride | Pipes, cards, sanitary fittings, etc. |
| 4 PEBD | Low density polyethylene | Bags, film, packaging, etc. |
| 5 PP | Polypropylene | Food packaging, lids, reusable cups, etc. |
| 6 PS | Polystyrene | Single-use plates and cutlery, yogurts, butter packaging, etc. |
| 7 Otros | Other plastics | Multiple applications |

a good alternative for reducing the amount of oil used and greenhouse gas emissions, as well as reducing landfill waste.

The aim of this study is the characterization of the various plastic materials existing in mixed municipal solid waste and to assess the need of the washing/drying steps and their potential impact as preliminary preparation steps on the global recycling process line.

## 2. MATERIALS AND METHODS

### 2.1 Materials

The raw material used in this study came from the municipal solid waste collected and treated at the Waste Treatment Plant (Ecocentral) in Granada (Spain), and corresponded to the fraction that had not been collected selectively. Municipal solid waste in Granada is made up of different fractions (Figure 1), among which the organic matter (34.4%) stands out, while plastic represents 12.6% of the total. This information comes from periodical characterizations carried out in the plant.

Figure 2 shows the different fractions of the plastic waste.Among them, polyethylene film is the most important, representing approximately 43% of the total. Plastic waste from all fractions except for rigid high-density-polyethylene, HDPE (mainly bottles) and plastics belonging to the category "Others" were analysed in the laboratory. The category "Others" includes many multilayer plastics, fibres and other polymers which are not the focus of this study.

At the Waste Treatment Plant of Granada, plastic waste is mechanically pre-treated to separate one type from

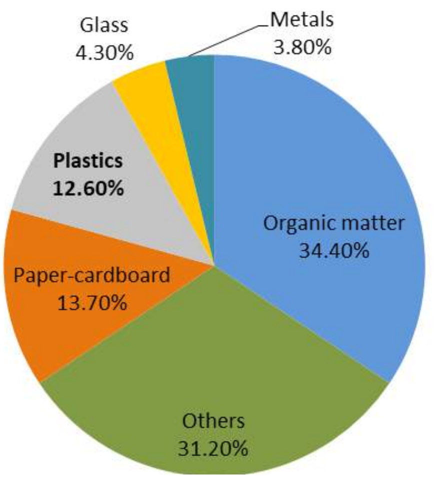



**FIGURE 1:** Composition of municipal solid waste in Granada (Spain).

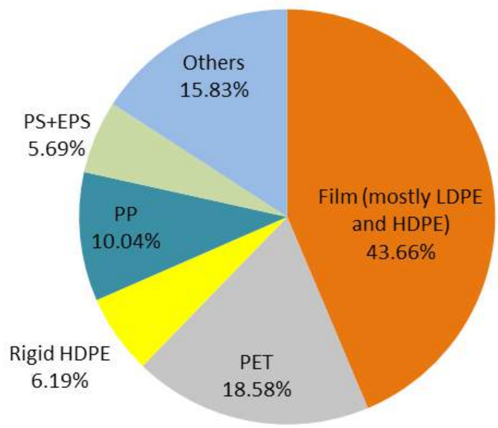



**FIGURE 2:** Composition of plastic fraction present in municipal solid waste of Granada (Spain).

another to facilitate the subsequent recycling and baling. The samples analysed in the laboratory came from the bunker corresponding to each type of plastic separated inside the plant (Figure 3). The plastic types analysed are:

- Polyethylene (PE) film (mostly bags and packaging film);
- Polyethylene terephthalate (PET) packaging;
- Polypropylene (PP) packaging;
- Polyestirene (PS) and Expanded Polystyrene (EPS) packaging.

## 2.2 Methods

The first step was the characterization of the plastic waste in the Waste Treatment Plant of Granada (Spain). The plastic waste was then transferred to the laboratory, where a series of analyses were carried out in the following order (Figure 4): 1) measurement of moisture content, 2) grinding, 3) washing and 4) measurement of dirt content. In parallel, the wastewater from the washing of each type of plastic material was analyzed, measuring the dissolved solids content, the total solids content and the chemical oxygen demand (COD).

*2.2.1 Characterization of raw material in the treatment plant and in the laboratory*

The purpose of this characterization was to determine the type of polymer that make up the waste material in order to separate and classify it. Several techniques were used for this purpose. At the municipal waste treatment plant, the plastic waste obtained from the bunkers was analyzed directly by Near Infrared Spectroscopy (NIR) using a portable Panatec Thermo Scientific microPhazir AG, with a wavelength range of 1600-2400 nm. The analysis was performed on the materials while they were in the waste stream, i.e. dirty and wet. This portable NIR spectrometer gives a reference spectrum with a correlation coefficient that indicates the similarity between the two spectra, appart from the measured spectrum.

Only the measurements with coefficients higher than 0.90 were considered. This value indicates that the material of the polymer can be considered to be the same as the reference material. However, this equipment does not provide the numerical values of the absorption peaks. In addition, moisture and dirt present in the material can lead to erroneous absorption peaks, which do not correspond to the polymer.

Thus, in the laboratory, this technique was complemented with Fourier Transform Infrared spectroscopy (FTIR), once the materials were washed and dried, to avoid disturbances caused by moisture and dirt. In addition, this equipment provides the values of the main absorption peaks, which allows comparing them with those in the literature (Bozaci et al., 2012; Rodríguez-Bruceta et al., 2014; Smith, 1999; Vahur et al., 2016; Zieba-Palus, 2017) and confirming the type of polymer that compose the waste.

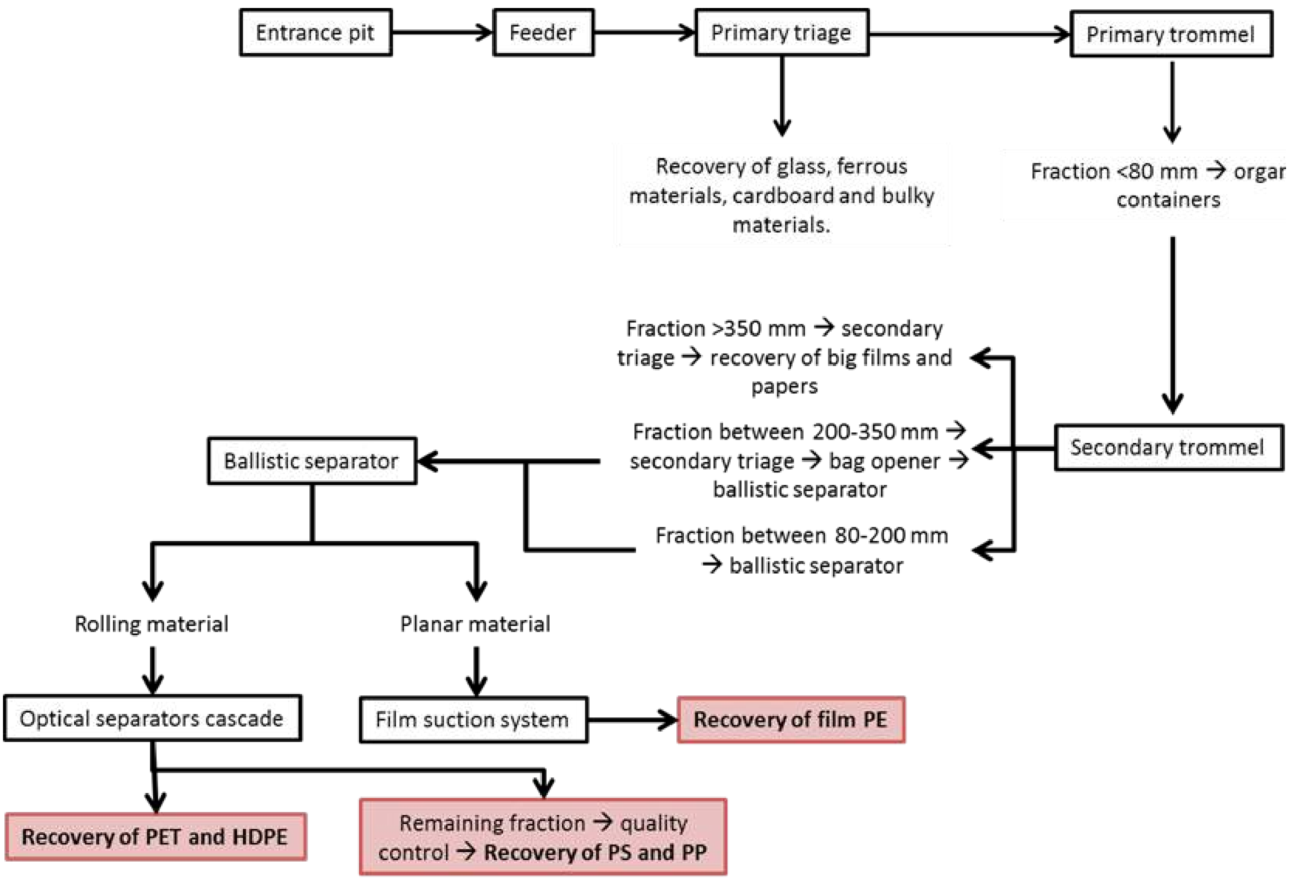



**FIGURE 3:** Scheme of mechanical pre-treatment carried out inside the Waste Treatment Plant of Granada (Spain).

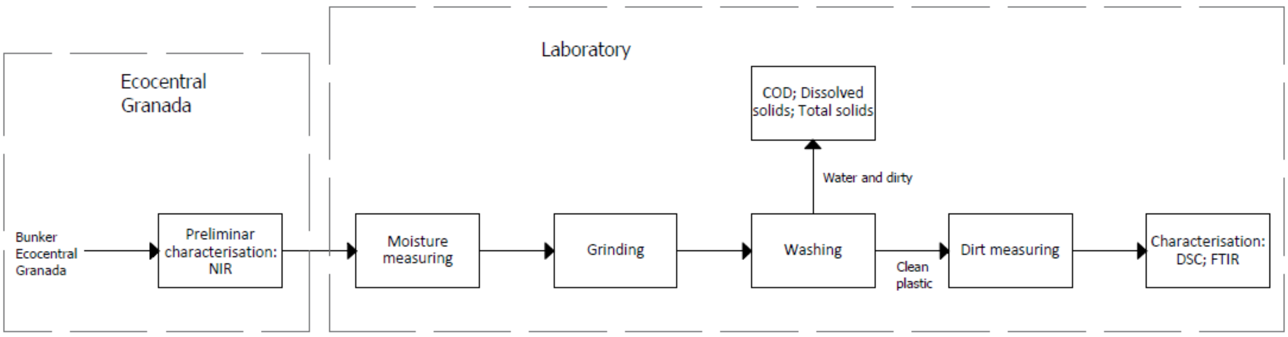



**FIGURE 4:** Experimental sequence of the study.

The analysis was performed with a Perkin-Elmer FT-IR Spectrometer model Spectrum 65, which has a wavelength range of 4000-400 $cm^{-1}$ and a resolution of 2 $cm^{-1}$.

In the laboratory, tests were also carried out using Differential Scanning Calorimetry (DSC) on the samples in order to obtain the melting points of the different polymers that compose the plastic waste, a technique that is also widely used to complement the previous ones. The analysis was performed on Perkin-Elmer Thermobalance model STA 6000, in accordance with the standard ISO 11357-3:2018.

### *2.2.2 Determination of moisture*

Moisture content was determined according to standard UNE-EN ISO 18134-3:2016 using a drying oven at a temperature of 105 ± 2 °C during 24 hours.

### *2.2.3 Washing process*

Three samples of each of the plastic materials described in section 2.1 were taken and washed. In order to determine the amount of water used in the washing process, several companies expert in the design of washing machines for the mechanical recycling of plastic were consulted. The minimum quantity acceptable for a correct washing is 1 L per 100 g of plastic. This quantity was complied for all the polymers except for EPS. A solid:liquid ratio of 1:40 was used for this material, because it is very bulky and it has a very low density, so it takes large amounts of water to submerge it completely and wash it properly. The duration of each wash was 30 minutes. Similar to our methodology, different water quantities were reported for diverse polymers. For example, after performing a cradle-to-gate life-cycle inventory (LCI) (EU data) for polymer recycling from post-consumer sources, Hopewell et al., (2009) found that the amount of consumed water was 32 kL/tonne for high-density polyethylene (HDPE), 43 kL/tonne for polypropylene (PP), 66 kL/tonne for (PET) and 140 kL/tonne for polystyrene (PS).

**TABLE 2:** Washing conditions.

| Polymer | Temperature | Ratio (plastic:water) |
|---|---|---|
| PE film | Room; 60°C | 1:10 |
| PET packaging | Room; 60°C | 1:10 |
| PP packaging | Room; 60°C | 1:10 |
| PS packaging | Room | 1:10 |
| EPS packaging | Room | 1:40 |

Washing was carried out using water from the urban network, which has a total dissolved solids (TSD) of 74.4 mg/l.

With regard to temperature, the plastic was washed at room temperature (23-25°C) and using hot water (60°C) following the recommendations given by other authors in the literature (Al-Sabagh et al.,, 2016; Awaja and Pavel, 2005; Kratofil et al., 2014; Luijsterburg, 2014; Rodríguez-Bruceta et al., 2014) (Table 2).

### *2.2.4 Determination of dissolved solids, total solids and COD in wastewater*

After the washing, the wastewater was analyzed and the following parameters were measured: dissolved solids, total solids and chemical oxygen demand (COD). These parameters were measured because they are the ones that determine the quality of the wastewater before it is discharged into the network. The methodology used to determine each of these parameters is described below:

- Dissolved solids: They were determined gravimetrically by vacuum filtration, according to UNE 77031:2015, using a Filter-Lab 1240 filter, with a pore size of 14-18 μm. The filtered water was then allowed to dry in a 105°C oven and the remaining solid residue was weighed. The result is expressed in g/L.
- Total solids: They were determined by the difference in weight between dirty plastic and clean plastic. The result is expressed in g/L.
- COD: This parameter has been determined in accordance with ISO 6060:1989, using the dichromate method. The chemical oxidizer was added to the wastewater and boiled. It remained in this state for a time, after which it was reduced through a reducing agent. Finally, it was evaluated in order to measure the amount of chemical oxidant consumed, expressed in mg/L of equivalent oxygen.

### *2.2.5 Determination of dirt loss*

Dirt loss was determined considering the weight loss of the plastic after the washing process. This parameter corresponds to the total solids parameter, as both measure the same solid fraction, but expressed in different units.

This is why the same values are presented in Table 3 for dirt loss as in Tables 4 and 5 for total solids.

The loss of dirt is expressed in percentage.

## 3. RESULTS AND DISCUSSION

### 3.1 Identification of polymers existing in raw material

The infrared spectra obtained by the NIR technique in the plant revealed that the majority of plastic materials analysed were composed of PE, PET, PP, PS, EPS and PA. This result coincides with the characterization presented in Figure 2, so it can be stated that the majority of plastic wastes contained in MSW are composed of PE, PET, PP, PS, EPS and PA (Figure 5).

However, due to the heterogeneity and characteristics of the sample in the field analysis with the NIR equipment, in some cases FTIR technique in the laboratory was performed as well (Figure 6). The spectra obtained were compared with spectra from the literature of pure polymers (Bozaci et al., 2012; Rodríguez-Bruceta et al., 2014; Smith, 1999; Vahur et al., 2016; Zieba-Palus, 2017). It was found that the absorption peaks coincided with those described for PE, PET, PP, PS and PA by these authors. In the case of PE, in the characterization with FTIR, it was possible to differentiate between high density polyethylene (HDPE) and low density polyethylene (LDPE), since there is at least one different absorption peak between both, according to other authors (Kochetov et al., 2017; Rodríguez-Bruceta et al., 2014; Smith, 1999).

Another method used to verify the type of polymer, as described in section 2.2.1, was the DSC. This method is considered as the most decisive in polymer characterization, since it provides the melting temperature. All the polymers analyzed gave consistent results except PA. In the characterization phase, this material was detected only in multilayer products, containing PET or PE in addition to PA, and the layers could not be separated. This resulted in numerous overlapping and difficulties to identify fusion peaks in the DSC. For that reason, the DSC of these materials has not been included in the results.

Figure 6 shows the results obtained in the laboratory. In the case of PET, the melting temperatures reported by other authors (Awaja and Pavel, 2005; EAG Laboratories, 2018) were between 250-265°C, although slightly lower temperatures can be obtained,which is consistent with the results presented in this paper. For PP, the melting temperature is usually higher than 160°C (Hindle, 2018; Mofokeng et al., 2011), coinciding with the DSC shown in Figure 6. In the case of PS and EPS, the temperature obtained in the DSC is the glass transition temperature since they do not have melting temperature because they are amorphous polymers (Oliveira et al., 2013; Parres-García, 2005). This value is usually around 100°C, which is in accordance to the value obtained in the present investigation.

It should be noted that, within the PE, the DSC technique could be used to distinguish between HDPE and LDPE whose melting temperatures are different. HDPE may have melting temperatures between 120-130°C (Chianelli et al., 2013), but in most cases the values are closer to 130°C or even higher, such as those obtained by some authors (Araújo et al., 2008; Shnawa et al., 2015) and in the DSCs presented in this article. On the other hand, the melting temperature of low-density polyethylene is usually between 115-125°C (Batra, 2014), but it can fluctuate slightly above or below these values since other authors have obtained melting temperatures of 113°C or 127°C (Ashraf, 2014; Poley et al., 2004). These values are also in accordance with those shown in Figure 6 for the LDPE.

All the results obtained by these three techniques allowed to separate and quantify precisely the composition of the fraction of plastic material contained in the MSW, as well as to confirm the results of Figure 2.

### 3.2 Moisture and dirt content of plastic materials

Table 3 shows the data related to moisture and dirt obtained for all types of plastic materials separated from mixed municipal solid waste of the province of Granada (Spain). The samples with the highest moisture content were PS, EPS and PE film. Other authors such as Carranza et al.,, 2010 obtained moisture values of 10-20% for PE film waste from greenhouses.

With regard to the dirt present in plastics, it was found that most of it was made up of organic matter (soil, plant debris, etc.), paint and chemical residues, as well as labels and glue residues. Organic matter was more abundant in PE waste, paint and chemicals were present in PP waste, while labels and glue residues were more abundant in PET, PS and EPS. The labels were removed after the material had dried, as they were easily detached. After removal they were weighed. They represent between 10-14% of the plastic material and they are therefore an important parameter to take into account for mechanical recycling. Organic matter and paint residues were removed during washing.

The waste that contains the most dirt is PE film, followed by PS. PP had very low dirt loss values. This causes the viability of mechanical recycling and the quality of the final product can vary from one material to another. The

**TABLE 3:** Moisture and dirt content at different temperatures for each polymer.

| | PE | PET | PP | PS | EPS |
|---|---|---|---|---|---|
| Moisture (%) | 11.78 | 8.90 | 1.58 | 20.98 | 16.10 |
| Dirt (%) at room temperature | 13.79 | 8.50 | 2.65 | 10.20 | 7.91 |
| Dirt (%) in hot water | 13.17 | 6.45 | 1.53 | -- | -- |

**TABLE 4:** Characteristics of washing water at room temperature.

| | PE | PET | PP | PS | EPS |
|---|---|---|---|---|---|
| Total dissolved solids (g/l) | 5.64 | 3.52 | 0.59 | 5.71 | 1.00 |
| Total solids (g/l) | 13.79 | 8.50 | 2.65 | 10.20 | 7.91 |
| COD (mgO$_2$/l) | 1920 | 851.57 | 208.67 | 526 | 340.25 |

**TABLE 5:** Characteristics of hot washing water.

| | PE | PET | PP | PS | EPS |
|---|---|---|---|---|---|
| Total dissolved solids (g/l) | 5.68 | 2.04 | 0.40 | - | - |
| Total solids (g/l) | 13.17 | 6.45 | 1.53 | - | - |
| COD ($mgO_2$/l) | 1733 | 510.29 | 210 | - | - |

loss in weight of dirt is not necessarily determined by the temperature of the washing.

Washing with hot water did not imply a greater loss of dirt in terms of weight. In fact, the opposite effect was observed (Table 3). It was found no significant difference in the loss of dirt from the material when washing PE, PET and PP at room temperature or in hot water. Therefore, PS and EPS were washed only at room temperature, in order to save water and electricity. However, the use of hot water made it possible to better remove paint and glue residues and improved the loss of fat, giving the washed material a brighter appearance.

Differences in overall weight loss were significant between different types of waste: While PP had an overall weight loss of approximately 4%, the weight loss measured for PS was 30% of its weight after drying and cleaning by water. These results are of special interest from the point of view of the recycling process of these materials.

### 3.3 Determination of dissolved solids, total solids and COD on washing water

Tables 4 and 5 show the characteristics of the washing waters at room temperature and hot temperature. Dissolved solids, total solids and COD were determined for the different kinds of water. Washing in hot water did not imply a greater presence of dissolved solids in the water.

The wastewater will need to be pre-treated depending on the legislation concerning discharge to the sewerage networks of each city or country.

In the case of Granada, the Spanish province where this study was carried out, both the values of dissolved and total solids and the COD value comply with the limits established by the Municipal Ordinance regulating discharges to the sewage network of the Granada City Council, except in the case of polyethylene film. The COD values obtained for this material exceed the limits (1400 $mgO_2$/L), so its wastewater will need to be pre-treated prior to discharge. This pre-treatment would include several steps of decantation and aerobic degradation designed in order to achieve the discharging limits. The need of these steps will increase the initial cost of the recycling plant. In those cases, where a separate discharging network for industrial waste stream exists, purifying equipment will not be necessary.

## 4. CONCLUSIONS

The study carried out in this work is the first part of an investigation for the mechanical recycling of different types of plastic materials in order to promote this type of

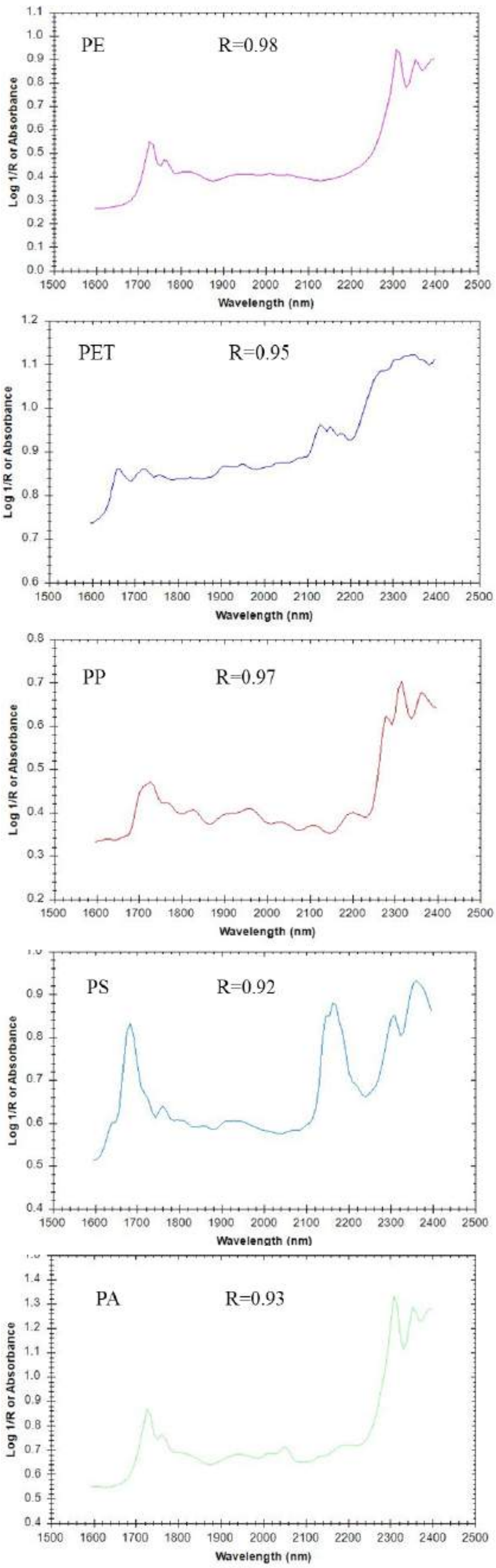
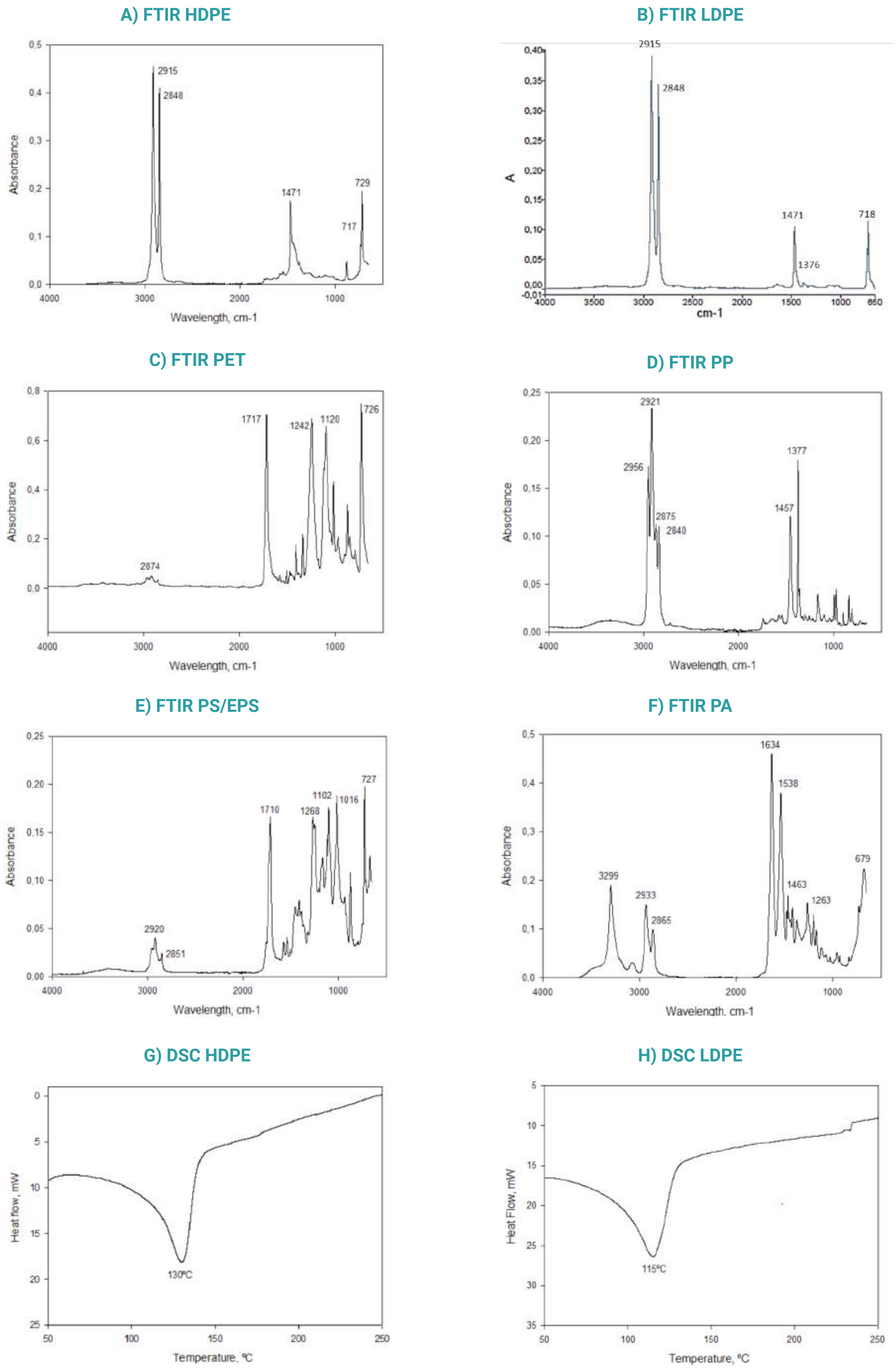



**FIGURE 5:** NIR spectra of the different plastic waste identified in MSW of Granada (Spain).


A) FTIR HDPE
Absorbance
Wavelength, cm-1
2915
2848
1471
729
717
B) FTIR LDPE
A
cm-1
2915
2848
1471
1376
718
C) FTIR PET
Absorbance
Wavelength, cm-1
1717
1242
1120
726
2874
D) FTIR PP
Absorbance
Wavelength, cm-1
2921
2956
2875
2840
1377
1457
E) FTIR PS/EPS
Absorbance
Wavelength, cm-1
1710
1268
1102
1016
727
2920
2851
F) FTIR PA
Absorbance
Wavelength, cm-1
1634
1538
679
3299
2933
2865
1463
1263
G) DSC HDPE
Heat flow, mW
Temperature, ºC
130ºC
H) DSC LDPE
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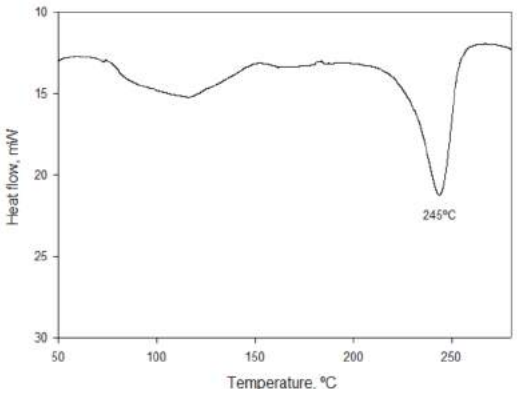
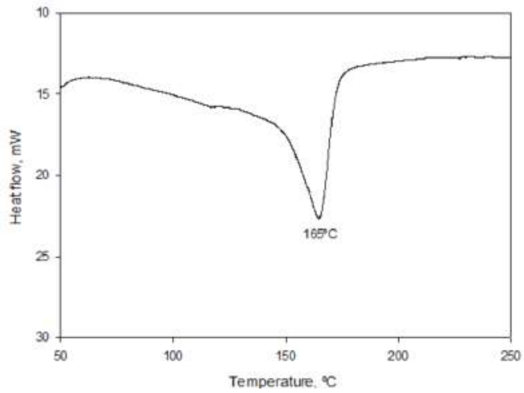
115ºC

I) DSC PET




J) DSC PP




K) DCS PS/EPS

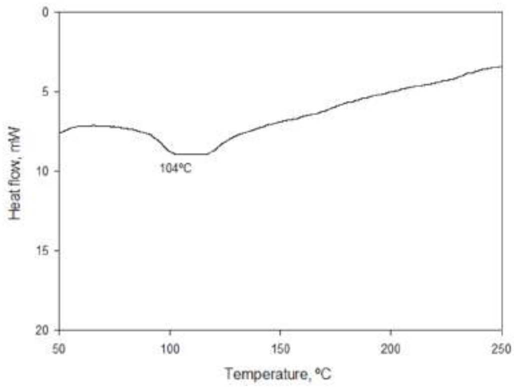



**FIGURE 6:** A-E) FTIR spectra obtained in laboratory of plastic waste present in MSW of Granada (Spain); F-I) DSC obtained in laboratory of plastic waste present in MSW of Granada (Spain).

recycling among plastic waste from mixed municipal solid waste. Mechanical recycling is a good alternative today to reduce the amount of waste that ends up in landfills in order to reduce the amount of oil needed to make virgin polymers and to contribute to reducing greenhouse gas emissions.

The characterization of the raw material carried out with various Infrared Spectroscopy and DSC techniques showed that the plastic fraction of the MSW from Granada (Spain) was composed mainly of PE, PET, PP, PS and PA.

With regard to the data found during the washing process, moisture and dirt in the waste were key factors in mechanical recycling, as it can account for up to 30% of its weight. This affects the performance of the process depending on the waste in each case. PS is the material that contains the most moisture and PE the most dirt. PP is a material with less moisture and dirt in general and for that reason it loses little weight during the process. It is important to take these parameters into account because depending on the final treatment that the recycled product will have (injection, blowing, extrusion, etc.), the previous stages of washing and drying may be decisive. It was also concluded that temperature was a non-significant factor in the amount of dirt loss in terms of weight, but it did allow better removal of paint, fat and glue. From the point of view of feasibility, materials such as polyethylene film can be washed at room temperature to reduce energy costs, since their main source of dirt is organic matter (plant debris and soil). Finally, it should be noted that, in most cases, the waste water from the plastic washing process would not require further treatment for discharge in Granada (Spain). However, it would be necessary in the case of the PE.

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# ASSESSING THE USE OF DEFAULT CHOICE MODIFICATION TO REDUCE CONSUMPTION OF PLASTIC STRAWS

Travis P. Wagner [1,*] and Patti Toews [2]

[1] University of Southern Maine, Environmental Science & Policy, Gorham, Maine, USA
[2] San Luis Obispo County Integrated Solid Waste Management Authority, San Luis Obispo, California, USA

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**ABSTRACT**

There is widespread global interest in eliminating and/or reducing the use of single-use plastics. An increasingly popular target is single-use, plastic drinking straws. Although plastic straws are not a significant component of the wastestream by weight or volume, they are one of the most commonly found items in coastal litter cleanups around the world. In addition, plastic straws are an avoidable product as their use is not essential. This paper examined the impact of an ordinance based on modifying the default choice of straws, which prohibited their distribution unless a customer requested one. Based on a survey of 133 affected businesses, the reported average decrease in straw consumption was 32% (SD=27.5%). For restaurants not using a self-service straw dispenser, the average decrease was 41% (SD=25.2%) The majority of businesses reported no impact to their business, some indicated a small decrease in costs, and others reported some negative feedback from customers. Based on the study's results, the straw-upon-request-only ordinance has been successful in reducing the consumption of plastic straws while minimizing impacts to businesses.

## 1. INTRODUCTION

Plastic waste, especially plastic marine debris, has risen onto the global agenda. Instrumental in this rise are the visceral focusing events involving marine debris that attracted worldwide attention of the public and policymakers. With over 32.6 million views, a viral video of a sea turtle with a straw lodged in its nostrils (Ramey and Tita, 2018) and viral videos of aerial and subsurface views of the Great Pacific Garbage Patch spurred mass and social media worldwide to focus on the problem of plastic marine debris. While these focusing events helped to bring the issue of single-use consumer plastics to the global agenda, plastic straws were the primary target as grassroots efforts worldwide sought to push businesses and governments to respond.

This paper examines local governmental efforts in the US to reduce plastic straw consumption. Aside from bans, the most common approach in the US is to modify the default choice of straws by requiring that straws by provided only upon request by a customer. This paper analyzes the ordinance adopted by San Luis Obispo, California, which modified the default choice of plastic straws.

### 1.1 Plastic Straws in the Environment

Although plastic straws are a very small component of municipal solid waste (MSW) by weight and/or volume, they are emblematic of the disconnect in society's understanding of the environmental consequences of unfettered consumption of disposable single-use plastics. Except for some limited circumstances, such as in health care and individuals with physical limitations, single-use plastic straws are a highly avoidable product. That is, their sole purpose is to convey liquid from a container to the mouth. Aside from the exceptions above, using a straw is about consumer preference as opposed to necessity. They are avoidable in that beverages can be consumed by most people without the aid of a straw. Their use is also culturally arbitrary as some beverages, like soft drinks and water from a glass, are routinely served with straws while using a straw for hot coffee, beer, wine, or bottled water would be considered unusual. Like many single-use plastic consumer items, straws have a short utility measured in minutes because they are no longer needed after the beverage is consumed.

Plastic straws are a common component of litter. The Ocean Conservancy's annual International Coastal Cleanup is a one-day event conducted in coastal areas of 116 countries. During the event, debris is collected by volunteers and is categorized, counted, and weighed. As shown in Figure 1, since 1988, plastic straws have consistently been among

* Corresponding author:
Travis P. Wagner
email: travis.wagner@maine.edu


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the top items collected based on item counts during annual cleanup events. For example, as shown in Figure 2, straws and stirrers were the 7th most prevalent item collected globally during the 2017 International Coastal Cleanup.

According to Keep America Beautiful (KAB, 2015), paper-related food packaging, which includes wrappers for plastic straws, is the 7th most littered item on land. Although plastic straws are not a commonly categorized item in litter survey counts in the US, as presented in Table 1, there is some data on land-based straw litter.

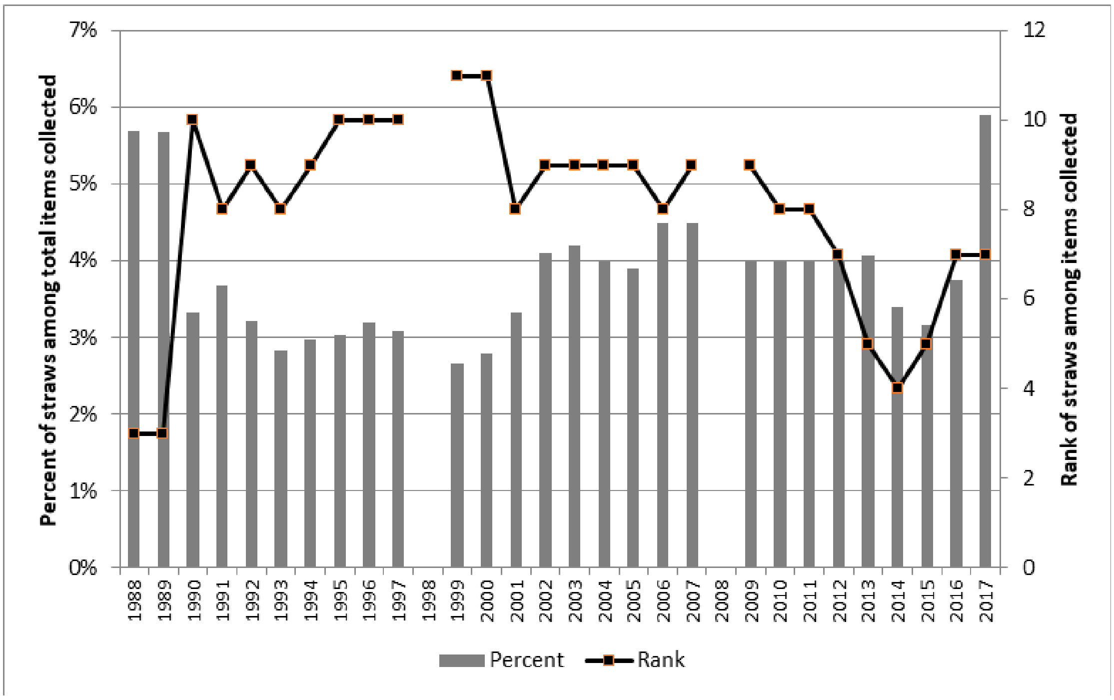


**FIGURE 1:** Coastal litter-plastic straws: percent among total items collected and rank among items collected, 1988-2017.
The left axis is the percent of plastic straws in relation to the total amount of items collected during the annual clean-up events. The right axis is the rank (1 is the highest rank) of the plastic straws in relation to all other items collected.
Notes: In the 1989 and 1990 counts, plastic straws were categorized as plastic eating utensils, which included cups, spoons, forks, and straws. Data from 1988 was for the US only, data from 1989 and 1990 was for North America only, and all data after 1991 is international. Between the 1991 and 2000 counts inclusive, plastic straws were a separate category. Starting in 2001, plastic straws were combined with plastic beverage stirrers. There is no data for 1998 or 2008.
*Source: Annual International Coastal Cleanup Reports. https://oceanconservancy.org/trash-free-seas/international-coastal-cleanup/annual-data-release.*

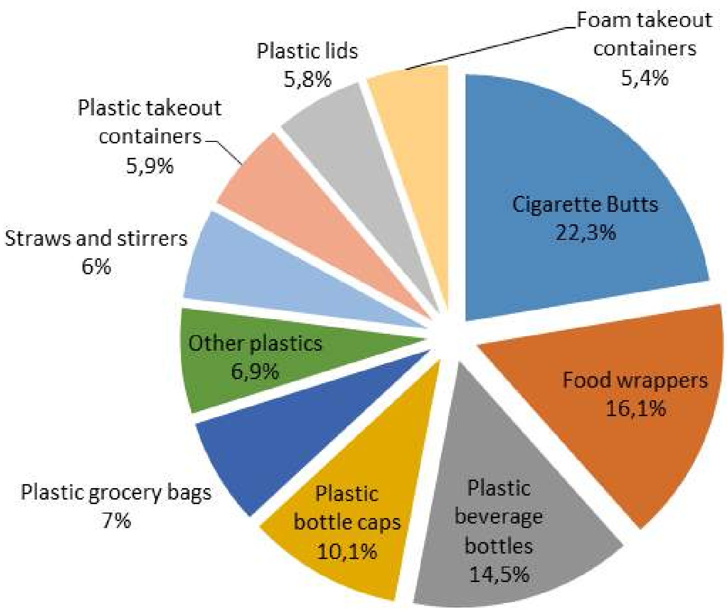


**FIGURE 2:** Breakdown of the top 10 items collected during the 2017 International Coastal Cleanup Event (Ocean Conservancy, 2018).

**TABLE 1:** Summary of litter collection results in the US that included plastic straws.

| Prevalence of Straws as Litter | Geographic Location | Source |
|---|---|---|
| Plastic straws, cups, and lids = 10,7% | Maine | Environmental Resources Planning, 2010 |
| Plastic straws, cups, and lids = 8,5% | New Hampshire | Environmental Resources Planning, 2010 |
| Plastic straws, cups, and lids = 7,1% | Vermont | Environmental Resources Planning, 2010 |
| Plastic straws = 2,5% | Anacostia River Watershed, greater Washington, DC area | Environmental Resources Planning, 2015 |
| Plastic straws = 2,2% | Litter from the curbside collection of recycling bins, Portland, Maine | Wagner and Broaddus, 2016 |

## 1.2 Consumption of Plastic Straws

Single-use plastic drinking straws are made primarily of polypropylene (Resin Identification Code, RIC, #5). In addi-tion, plasticizers, colorants, antioxidants, ultraviolet light filters, and inert fillers are added (Made How, 2018). When designed for individual use at food and beverage estab-lishments, plastic straws are commonly wrapped in paper sleeves and when they are attached to pre-packaged drinks in aseptic boxes or drink pouches, they are wrapped with plastic sleeves (Twede et al., 2014).

Plastic straws are used in a large number of establishments with food and beverages as a primary or secondary business. These include causal and fast-food restaurants, cafes, hotels and motels, theatres, food trucks, kiosks, airlines, cruise ships, bars, night clubs, delicatessens, coffee bars, and sports venues. They also are used at cafeterias in schools and colleges, governments, prisons, hospitals, public facilities, and businesses. And finally, they are used at special events including festivals, fairs, and concerts.

There is no solid data on the consumption of plastic straws because it is not routinely collected as they are generally purchased in bulk quantities, and for businesses, it is generally viewed as proprietary data. The most commonly cited figure, 500 million straws per day consumed in the US, which equates to about 1.6 per person each day for a total of 182.5 billion per year, is erroneous. This number has been cited by numerous environmental organizations, governments, and the media; however, the derivation of this figure has been shown to be highly faulty (Lombardo, 2018).

Straws, when consumed, are most often used when eating out or when purchasing prepackaged drink boxes or pouches. Regarding eating out, 70% of North Americans eat out at least once per week and of this amount, 27% eat out at least 3-6 times per week (Nielsen, 2016). Although the data is old, according to Miller (2001), Americans consumed 17 drink boxes per person per year in 2001. According to Britschgi (2018), based on an industry marketing analysis, Americans consume about 63.875 billion straws per year, which equates to 175 million per day or about 3.73 per person per week. Based on limited data available, we estimate that a more plausible range of plastic straw consumption in the US is 4 billion to 20 billion per year, which equates to 10.95 to 54.8 million per day or about 0.23 to 1.17 per person per week (see also Blackwell, 2010; Lombardo, 2018).

## 1.3 Policy Instruments to Reduce Plastic Straw Consumption

As plastic waste rose sharply on the global agenda, there was a corresponding rise in global grassroots efforts to curb the consumption of plastics. These grassroots efforts focused on governments and local, national, and multi-national businesses to address the problem of plastic waste; plastic straws were especially targeted because they are an easily avoidable product. These efforts have been successful. In North America, for example, there have been many local and national campaigns for consumers and businesses to voluntarily reduce straws including National Skip the Straw Day, Straw Wars, Straws Suck, the Last Plastic Straw, One Less Straw, and No Straw Please. A&W Canada was the first North American restaurant chain to announce a phase-out of plastic straws starting in 2019. One of the largest fast-food restaurant chains in the world, McDonald's, committed to phase-out the use of plastic straws in 1,300 of its restaurants in the United Kingdom and Ireland and Starbucks has announced the same for its 28,000 coffee bars around the world by 2020. KFC halted the use of plastic straws in its Singapore restaurants. Ikea announced a ban on all single-use plastic serving ware including straws by 2020. Carnival Cruise Lines has adopted a straw-upon request only policy and Royal Caribbean is phasing out plastic straws. And, private facilities in the US, such as zoos and aquariums, have also banned or are phasing out the distribution of plastic straws (Rogers, 2017). In May 2018, the European Union proposed the Single Use Plastics Directive to reduce the 10 single-use plastic products, including straws, most often found in marine litter. Jamaica and Grenada have banned plastic straws starting 2019; Belize will celebrate Earth Day 2019 by eliminating single-use plastic straws, bags, and utensils; and India announced a ban on all single-use plastics by 2020.

While the global social movement has focused on corporate social responsibility to reduce or eliminate straws, and a few national governments have acted. In the US, the national government has not acted and only one sub-national (state) government has, California. In the absence of national and state-level actions, it is up to local governments to act should they choose to. Given the intense public and media attention, local governmental efforts to reduce single-use plastic straws have increased as discussed below.

As summarized in Table 2, and explained below, there are five primary public policy instruments that are available to reduce the consumption of single-use plastic straws including bans, taxes/fees, education, default choice modification, and voluntary actions.

### *1.3.1 Ban*

Bans seek to prohibit the distribution or use of plastic straws at specified businesses or properties (e.g., govern-

**TABLE 2:** Major public policy instruments to reduce the consumption of single-use, plastic straws.

| Policy Instrument | Summary | Positive Attributes | Negative Attributes |
|---|---|---|---|
| Ban | Prohibit covered establishments prohibited from providing plastic straws. | Eliminates consumption, easy to enforce. | Eliminates consumer choice. Non-plastic alternatives cost more, which are borne by the establishment unless take-out fee charged. |
| Default Choice Architecture Modification | Provide straws to customers only ifwhen requested. | Reduces consumption, retains consumer choice, small cost decrease to retailer establishment. | Difficult to enforce. If self-service for straws prohibited, could require increased establishment involvement to provide straws. |
| Tax/Fee | Visible, separate tax or fee levied on straws at point of purchase. | Reduces consumption. Relatively easy to enforce. Retains consumer choice | Increased cost (but avoidable) to consumers and increased administrative cost for regulator and establishment. |
| Education | Educating retailers establishments and consumers on need to reduce consumption of straws. | Low or no cost to consumers; does not impose restrictions on consumers. | Not likely to have appreciable impact on consumption or recycling.. May impose some cost to retailer establishment. |
| Voluntary Actions | Adopting resolutions to encourage establishments to voluntarily reduce use of straws. | No cost to consumers; does not impose restrictions on consumers or establishments. | Impact on consumption uncertain and variable depending on breadth and duration of adoption. |

ment facilities, public parks, etc.) thus they are the strongest possible action to reduce the use of straws. Bans, however, tend to be unpopular because they reduce consumer choice (Coulter, 2009). In theory, bans are easy to enforce, but without enforcement, compliance can be spotty. Seattle adopted a ban on single-use plastic bags, but based on a random sample of compliance inspections, small and independent grocery and convenience stores had a low compliance rate (Hoffman, 2016).

A ban on plastic straws is feasible because there are available alternatives including, of course, avoidance--not using a straw. Common substitutes for single-use plastic straws include corn-based polylactic acid (PLA), paper, and pasta, and reusable straws include silicone, stainless steel, glass, and bamboo.

### *1.3.2 Default Choice Modification*

The most common default action for the distribution of straws is to provide them to customers automatically with the purchase of a beverage at food service operations regardless whether or not they are desired. (This is also the case with the purchase of many pre-packaged beverages in aseptic pouches and cartons, which have straws attached.) As a result, consumers have become conditioned through this repetitive action such that it has become an automatic expectation. In this case, the default choice architecture is to receive a free straw with a beverage regardless of whether it is desired. The policy approach, then, is to modify the choice architecture to alter consumer behavior (without banning the behavior), by encouraging a preferential selection (Thaler and Sunstein 2008). For straws, the approach is to change the default choice by requiring that straws be provided only upon request.

### *1.3.3 Tax/Fee*

Levying taxes or fees at the point of sale through a separate, visible, point-of-sale charge is a mechanism to internalize the cost to the consumer (Bury, 2010). Customers have been conditioned to expect free straws, thus they appear to be without cost as consumers do not see the price of the straw or the social/environmental cost (e.g., litter clean-up costs, reduced tourism, impacts to marine organisms, etc.) imposed by the straw resulting in excessive consumption (Taylor and Villas-Boas, 2016). When a consumer is presented with an additional cost to participate in an avoidable action, consumption tends to decrease. Even a low fee (e.g., $0.05) acts as a visible economic nudge, which is not meant to substantially increase the cost of an item, but to signify to the consumer that they face an economic choice (Rivers, Shenstone-Harris, and Young, 2017). Fees for single-use plastic and paper bags are popular in the US and other countries (Wagner, 2017).

### *1.3.4 Education*

Education is traditionally seen as the first-step in seeking to achieve a reduction in the consumption of a straw. If it is sufficiently successful, ordinances could be unnecessary. Ordinances that mandate some form of education generally rely on traditional passive education such as posting signs or notices discouraging the use of straws. The theory is that the behavior can be changed when given accurate knowledge of the impacts of the behavior. Regarding the adoption of education-based ordinances, this generally would require that mandatory signage be placed at self-serve straw dispensers or signage be placed at counters or table tops to encourage customers to forgo the use of a straw. Such education-based ordinances for consumer products typically would require the required posting locations and message content. Education, however, has had only limited success with regards to reducing single-use consumer products (Wagner, 2016).

### *1.3.5 Voluntary Actions*

Voluntary actions are primarily cooperative efforts undertaken without government intervention to achieve a certain desired goal. Voluntary-focused actions undertaken by the private sector rely more on social and corporate responsibility although they can be undertaken in an effort to avoid stronger potential government intervention. Individuals and firms generally will engage in voluntary actions if there are also economic benefits such as increased sales or the avoidance of decreased sales. The benefits of voluntary agreements is that they provide flexibility for the government and target population, can achieve the desired results at a lower cost, and can be relatively easily modified or ended. However, because they are voluntary, such an approach may be limited in its breadth (number of establishments adopting a voluntary straw ban), and the length of time that voluntary bans remain.

### 1.4 Local Plastic Straw Ordinances in the US

In September 2018, California became the first state in the US to enact a plastic straw law. The law adopted the "straw only upon request" approach. It applies only to full-service restaurants; it does not apply to fast-food restaurants, coffee shops, delicatessens, or restaurants serving takeout to customers. (California cities and counties may adopt more stringent ordinances involving straws such as which establishments are covered and the adoption of a ban.) Regarding local ordinances, which are far more prevalent in the US, as of September 2018, there were 31 ordinances that had been adopted by local governments (13 in California, 7 in Florida, 3 in New Jersey, 2 each in Massachusetts and Washington, and 1 each in Minnesota, New York, Ohio, and South Carolina). A shown in Table 3, of these 31 municipal ordinances, 16 are full bans, 6 are partial bans, and 9 default choice modifications. There have been numerous resolutions passed by local governments encouraging businesses to reduce the use of plastic straws, but these are not ordinances as they are suggestive and do not have the force of law.

## 2. METHODS

In this paper, we examined a default choice modification ordinance covering plastic straws enacted by the city of San Luis Obispo, California. This examination sought to answer the following questions regarding default choice modification ordinances: Can it reduce the consumption of plastic straws? What are the impacts to businesses? What is the level of customer acceptance?

San Luis Obispo, located on California's central coast, has a population of just over 47,500 and a population density of over 3,619 people per square mile. The majority of the city's population is Caucasian (84.5%) with Hispanic/Latino (14%), and other races.

On November 7, 2017, the city's straw ordinance was passed unanimously by the city council and become effective on March 1, 2018. An impetus for the ordinance was that plastic straws in the city are not recycled and thus are landfilled or become litter. Plastic litter is a significant problem in San Luis Obispo: 1,363 plastic straws/stirrers were collected during the 2017 SLO Coastal Cleanup Day making plastic straws/stirrers the 10th most collected item and had been among the top items collected over previous annual coastal cleanup days. In addition, the city council had made the elimination of plastic waste a formal objective.

To reduce the consumption of plastic straws, the city of San Luis Obispo enacted an ordinance that modified the default choice of straws. That is, the ordinance specified that restaurants could no longer be automatically handed to dine-in customers; straws could be provided only upon request. The ordinance does not apply to customers who purchase food to take away; they may be handed straws without requesting them. In addition, food trucks are not subject to the ordinance and self-serve straw dispensers were not prohibited.

To assess the impact of the plastic straw ordinance, a 9-question survey was developed (see Table 4). Following pilot testing, in June 2018, a hard copy of the survey was hand-delivered to all 161 restaurants covered by the ordinance. Each owner/manager was requested to complete the survey themselves. If they were unable to at the time, they were re-visited to collect the survey.

## 3. RESULTS

We collected 133 survey responses for a response rate of 82.6%. Of the respondents, 52% were casual/fine dining restaurants, 38% were fast food restaurants, 8% were coffee bars, and 2% were miscellaneous food establishments. Because food trucks and takeaway restaurants were not covered by the ordinance, they were not surveyed. For the purchasing source, 78% of the respondents purchase their straws while for 22%, a corporate or branch office purchases the straws.

As shown in Figure 3, 67% of respondents reported that they distributed fewer than 500 straws per week prior to the ordinance and the most frequent category chosen was fewer than 100 straws per week.

Regarding the question on the percentage decrease of straws since the ordinance, as shown in Figure 4, although 30 (22.5%) respondents stated that the amount was unknown, 103 respondents stated that the average decrease in consumption was 32% (SD=27.47%) and the median decrease was 30%. Of the 14% who reported that the decrease was zero, 63% of these respondents use self-service dispensers for straws. That is, customers do not have to ask for a straw, they can serve themselves,

**TABLE 3:** Summary of current US local ordinances to reduce plastic straw consumption (N=31).

| Policy Approach | Summary |
|---|---|
| Full Ban (16) | Covered establishments prohibited from providing plastic straws to customers.<br>Most of the bans do not cover the sale or distribution of pre-packaged drinks (aseptic boxes and pouches) with straws pre-attached, straws used at schools, use at medical/dental facilities, and for customers with physical limitations. |
| Partial Ban (6) | Bans the use or sale of straws only on or adjacent to public beaches, pubic parks, or city property.<br>Does not ban the use or sale of straws on private property. |
| Default Choice Modification (9) | Plastic straws are allowed at covered establishments, but may only be provided when requested by a customer (the new default choice); they may not be provided without request.<br>Ordinances vary as to the definition of a covered establishment, some exclude take-out restaurants and/or food trucks.<br>Three of these local ordinances authorize or encourage retailers to charge a "take out" fee to cover any additional cost incurred by non-plastic substitutes. |

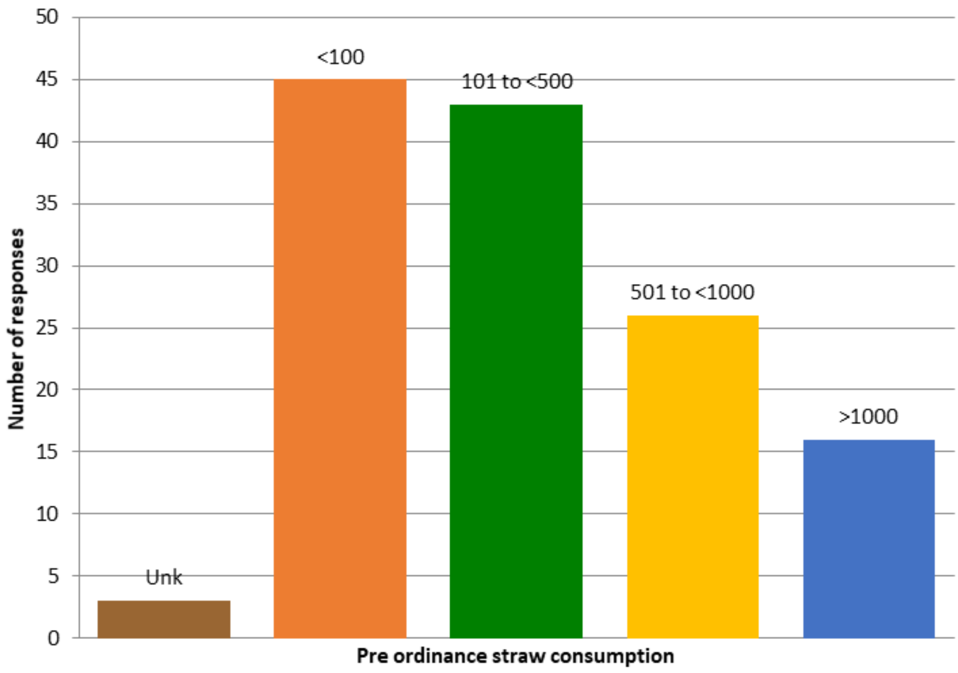


**FIGURE 3:** Reported weekly consumption of straws prior to the ordinance (N=133).

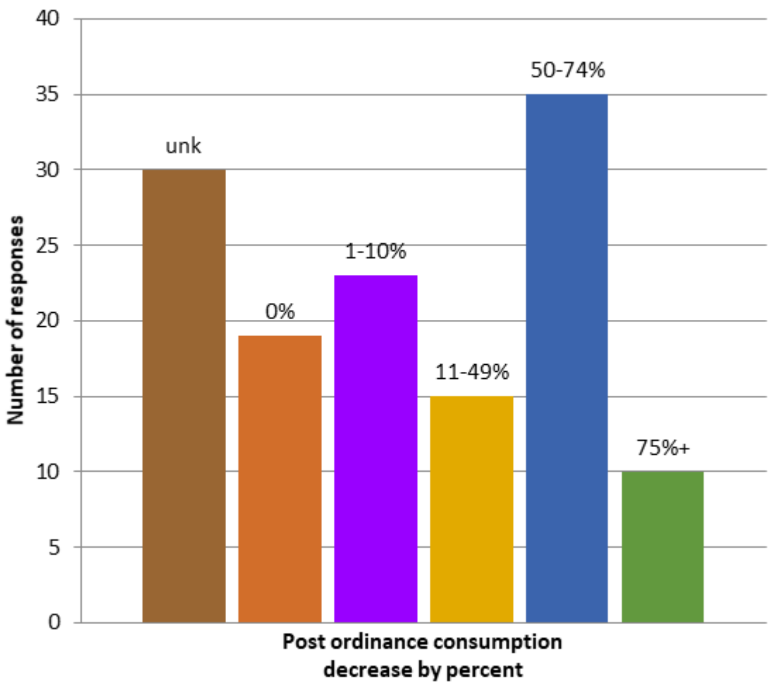


**FIGURE 4:** Reported reduction in consumption of straws post ordinance (N=133).

which is allowed under the ordinance. Nonetheless, of the respondents who use self-service dispensers for straws, the mean reported decrease in straw consumption was 19.8% (SD=25.8%) and the median decrease was 10%. For the respondents who do not use self-service dispensers, the average reported decrease in straw consumption was 41% (SD=25.2%) and the median decrease was 50%. Since the ordinance, 9.8% of respondents no longer offer a self-service option for straws as they removed their dispensers as a result of the ordinance. As presented in Table 5, of the business types reporting a specific percent decrease, casual/fine dining restaurant reported the highest decrease at 39.4%.

As shown in Figure 5, most of the businesses had not switched nor had offered non-plastic straw options while one business offers reusable bamboo straws for purchase.

The two most important open-ended questions sought responses on the impact of the straw ordinance on the business and the feedback to businesses received from custom-

**TABLE 4:** Survey questions to assess the impact of the San Luis Obispo plastic straw ordinance.

**1. Business Type:**

☐ Restaurant, fast food
☐ Restaurant, casual/fine dining
☐ Juice Bar
☐ Coffee Bar
☐ Supermarket
☐ Bar/Lounge
☐ Other:

**2. Are your straws provided by a main branch/corporate or do you purchase them for this business?**

☐ Branch/corporate
☐ Purchased by me
☐ Other:

**3. Are straws directly accessible to customers via self-service?**

☐ Yes
☐ No
☐ No, but only since the ordinance
☐ Other: ____________________________________________

**4. Prior to the straw ordinance, how many straws did your business use/provide per week?**

☐ <100 a week
☐ 101-500 a weekly
☐ 501-1,000 a week
☐ >1,000 a week

**5. Since the ordinance, what is the percentage decrease in straws used by your business?**

☐ ______%

**6. Have you switched to non-plastic straw options?**

☐ No
☐ Completely switched to non-plastic straw option
☐ We offer a non-plastic straw option for customers
☐ Other: ____________________________________________

**7. From your perspective, what has been the most significant impact to your business from the straw ordinance?**

**8. What has been the response from your customers?**

**9. What changes would you suggest to further reduce the consumption of single-use, plastic straws?**

ers. The responses were categorized into common themes based on the responses as presented in Tables 6 and 7.

## 4. DISCUSSION

### 4.1 Survey Findings

Based on the survey results, the ordinance has been successful in reducing the consumption of plastic straws a reported 32% average decrease per business. As noted, there likely would be a higher percentage reduction in straw consumption if self-service straw dispensers were not allowed. One important observation is that the survey was conducted three months after the effective date of the ordinance. Regarding customer feedback, 17.6% of the respondents stated that there was customer confusion over the ordinance as respondents noted many customers expressed ignorance of the ordinance. Some respondents reported increased acceptance following a brief explanation to the customer. This suggests that consumption can likely decrease as knowledge of the ordinance increases. This prediction must be tempered by the fact that this area has a significant tourism industry; visitors are less likely to be familiar with the straw local ordinance.

As time progresses, it will be more difficult to isolate the future impact of the ordinance on straw consumption. This is a result of the positive spillover effect. That is, as a "no straw" normalization is supported by grassroots campaigns, voluntary actions by businesses, and media coverage, this may have a positive effect on customer behavior as opposed to the ordinance alone.

Regarding the impacts to businesses, the number one response (42.5%) was that there was no significant impact while other responses were positive. The second largest impact (21.3%) was that respondents were pleased to report some cost savings in straw purchases and 6.4% expressed positively that there was also some cost savings in producing less waste and/or a positive impact to the environment. Only 23.4% of respondents expressed negative impacts to their business: 14.9% stated that the ordinance added an extra step in service and 8.5% of the respondents expressed concern over customers' perceptions of poor service because they did not receive a straw and were forced to request one. Regarding the poor service perspective, most of the respondents voicing this concern complained that this is due to a lack of customer awareness and suggested that a greater education effort for the ordinance could have reduced this perspective. This suggests that a government considering such an ordinance needs to have a well designed outreach plan to help reduce negative perceptions and impacts on businesses.

Interestingly, a few respondents noted that they have switched to straws wrapped in paper, which are more expensive and wasteful. With wrapped straws, servers can carry straws with them which allow them to reduce an extra step in service when they are requested by customers. Some businesses that switched to non-plastic alternatives noted a cost increase. Others commented on performance issues as consumers complained that paper straws tend to unravel or swell up especially in alcoholic drinks.

This specific ordinance only covers customers who dine-in; take-out restaurants, food trucks, and takeout orders are excluded from the ordinance. While removing

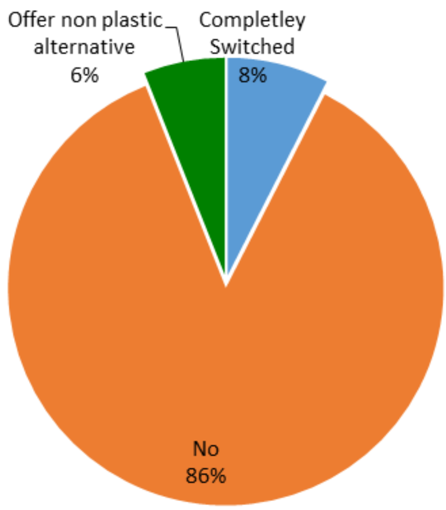


**FIGURE 5:** Response to the question, "Have you switched to non-plastic straw options?" (N=133).

**TABLE 6:** Categorized responses to the survey question: "From your perspective, what has been the most significant impact to your business from the straw ordinance?" (N=94).

| Response Category | Percent Responses |
|---|---|
| No significant impact to the business | 42.5% |
| Saves some money on straw purchases | 21.3% |
| Extra step in customer service | 14.9% |
| Negative, customers assume poor service when straws are not automatically provided | 8.5% |
| Less waste/Good for the environment | 6.4% |
| Miscellaneous impacts | 6.4% |

**TABLE 7:** Categorized responses to the survey question: "What has been the response from your customers?" (N=91).

| Response Category | Percent Responses |
|---|---|
| Mixed feedback (some for and against) | 29.7% |
| No feedback | 24.2% |
| Confusion over ordinance | 17.6% |
| Mostly positive feedback | 14.3% |
| Mostly negative feedback | 12.1% |
| Miscellaneous feedback | 2.1% |

these exemptions would likely have no effect on individual establishments, it is reasonable to presume that such a modification would decrease the overall consumption of plastic straws.

### 4.2 Policy Considerations

In exploring policy options to reduce plastic straw consumption, bans, fees, and default choice modification all have the potential to reduce straw consumption. Bans are likely to have the most negative response from the public, but would be the most effective at reducing straw consumption. Fees/taxes would also be highly effective, but are less popular and thus, more difficult politically. Although this approach has not yet been used for straws in the US, fees and taxes have been very effective at reducing single-use plastic bags (Wagner, 2017). As the survey has shown, in the case of San Luis Obispo, the default choice modification has been successful in reducing straw consumption with minimal impacts to businesses.

Regardless of a ban, fee, or default choice modification, exemptions should be considered for the health care industry, senior citizen facilities, and individuals with physical limitations or disabilities (e.g., paralysis, poor muscle control or contractures, etc.) necessitating the use of a plastic or reusable straw.

As discussed above, with default choice modification, consideration should be given to prohibit self-serving straw dispensers and not excluding take-out customers or restaurants (or phase-out the exemptions over time), which would further decrease the number of straws consumed. However, this would shift some of the burden on to the business by adding an extra step and creating the perception of poor service if one has to request something that previously was automatically provided. An extra step in service represents time opportunity costs for the business. To mitigate this impact, there should be a concerted effort to develop a strong outreach campaign tailored to the community on the importance of the ordinance and the environmental benefits. As this study found, the reduced costs were important to businesses suggesting that this should also be incorporated into an education campaign—positive economics.

While some communities are considering a ban with a goal to shift consumption to non-plastic straws, consideration should be given to the impact of moral licensing, which is when we engage in a socially desirable behavior, we tend to ignore the impacts of non-socially desirable behavior. Researchers have found that offering recycling actually increased the consumption of items offered for free as the moral licensing effect made consumption more acceptable (Catlin and Wang, 2013; Sun and Trudel, 2017). If paper and/or compostable straws are provided, with moral licensing, customers tend to feel okay about using and disposing, including littering, of paper and/or compostable straws. In sustainable materials management, the goal is source reduction rather than more recycling or composting.

## 5. CONCLUSIONS

As noted, plastic straws are not a significant source, by weight or by volume, of the global plastic waste problem, but they are highly visible and avoidable. Given that for most people, straws are basically superfluous, they are perhaps the easiest single-use plastic product that can be targeted for reduction. The global focus on straws represents an initial first step, and because they are avoidable, the relatively easiest step in a strategy to reduce and/or eliminate single-use plastics.

Government regulation often evolves very slowly, which explains why, during a time of high global interest in reducing single-use plastics, especially straws, there currently only a small number of straw ordinances in the US. Nonetheless, private businesses are moving much faster as there have been significant voluntary actions worldwide to reduce straw consumption. It is feasible that a global, no plastic straw normalization could arise that precludes the need for government intervention.

This paper reviewed one specific policy instrument, the default choice modification, which mandated that straws be provided only upon request. This instrument flips the default choice while retaining customer choice. In this study, it was found to be effective in reducing straw consumption with minimal impact to businesses.

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ities; and Chris Read, Sustainability Manager, City of San Luis Obispo.

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# CONSTRUCTION AND DEMOLITION WASTE MANAGEMENT IN CROATIA WITH RECYCLING OVERVIEW

Gordan Beděković *, Biljana Kovačević Zelić and Ivan Sobota

*Department of Mining Engineering and Geotechnics, Faculty of Mining, Geology and Petroleum Engineering, University of Zagreb, Pierottijeva 6, 10000 Zagreb, Croatia*

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**ABSTRACT**

Construction and demolition waste (C&D waste) is one of the most relevant waste types primarily due to large quantities and a high potential for re-use and recycle. This paper discusses the issues related to C&D waste management in the Republic of Croatia. It presents the overview of legislative changes and its impact on the C&D waste management in Croatia. C&D waste quantities per county in the period from 2001 to 2015 are given as well as the expected C&D waste quantities in the future. It is concluded that the legal framework is well established, but it is not implemented consistently. Therefore, it is necessary to plan certain activities and additional resources in order to make the C&D waste management more efficient. The paper also presents the options of C&D waste recycling and describes the recycling technology in a C&D waste recycling plant in the Republic of Croatia. Taking into account the relatively low price of recycled aggregate, long distance transport is not profitable. It is concluded that the use of mobile treatment facilities would be a good practical solution taking into account the underdeveloped infrastructure for C&D waste management.

## 1. INTRODUCTION

One of the major problems related to environmental protection in the Republic of Croatia is inadequate waste management. The quantity of waste is increasing, and the existing infrastructure is not sufficient (Official Gazette, 2017a). In addition to that, a major problem is a partial implementation of the regulations related to the waste management. The waste management sector is exceptionally important for the Republic of Croatia due to numerous reasons. The most important among them is a potential harmful effect on the environment (water, air and soil), on the quality of life and human health, plants and animals, affecting the attraction of Croatia as a tourist destination, as well as the international perception of Croatia as a country of preserved environment and healthy food production. In 2014 Croatia generated 3.7 million tonnes of waste, 3% of which is hazardous waste and 97% non-hazardous waste (Official Gazette, 2017a). Taking into account the source of waste, the largest part of 31% was generated in households (municipal solid waste and similar), 17% through commercial sources (various trades/crafts, insurance, brokerage, administration, cleaning, etc), 17% in the construction sector, 12% in manufacturing, 11% through collection, treatment, disposal and recovery operations and 12% through other commercial and industrial activities (Official Gazette, 2017a). If we analyse waste treatment in 2014, 56% of waste was deposited in landfills and 44% was processed, 40% of which for material recovery, 2% for energy recovery and up to 2% was treated by backfilling (Official Gazette, 2017a). Compared to the year 2012 the results demonstrate the decrease of 10% in disposal and the increase of 8% in recovery (Official Gazette, 2017a).

The prerequisite for solving the problem related to the waste management and raising the efficiency of waste management system is the creation and improvement of an appropriate legal framework. After the accession to the EU, Croatia is continuously harmonizing its legislation with EU legislative acts. According to the Act on Sustainable Waste Management (Official Gazette, 2013), C&D waste is the waste resulting from new construction, reconstruction, demolition and maintenance of existing buildings, as well as any excavation waste which cannot be used without prior recovery for the purposes of construction for which it was excavated. Taking into consideration the way it is generated, it includes demolition waste, road construction and maintenance waste, soil, stone and vegetation.

It is estimated that the C&D waste has a high potential for recycling and a high economic value and that 80% of the C&D waste can be re-used in China (Zheng et al., 2017). The Waste Framework Directive 2008/98/EC aims

* Corresponding author:
Gordan Bedekovic
email: gordan.bedekovic@rgn.hr


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to have 70% of C&D waste recycled by 2020 in EU. This will be achieved by improved waste identification, source separation and collection, improved waste logistics, improved waste processing, quality management, appropriate policy and framework conditions. The C&D waste is recognized as one of priority waste streams at the European level. Officially available data on generated C&D waste and the capacity for its treatment in the Republic of Croatia are not entirely comprehensive and reliable. According to the official data of the Croatian Agency for the Environment and Nature in the period 2011-2013, the registered quantities of the generated C&D waste were below 200 kg per capita. The growth rate of the registered quantities was recorded in relation to the registered quantities in the aforementioned period when they were below 200 kg per capita. Taking into consideration the actual growth of the construction sector in the Republic of Croatia until 2008 and its decline after 2008 due to the recession, the growth of the registered quantities (Table 1) can be primarily attributed to the improved data collection and processing system, as well as possible methodology differences in classification. Difficulties in determining quantities of the generated C&D waste are not present only in the Republic of Croatia, but also in other EU member states. One of the conclusions of the topic-related project "Management of construction and demolition waste" was that the methodology for determining quantities and composition of the C&D waste was not entirely harmonized at EU level, and the available data were not completely reliable (European Commission, 2011). It was concluded that all member states that in 2004 registered the quantities substantially lower than the average of EU member states (940 kg per capita), had underestimated their quantities of generated C&D waste. It indicated the importance of recognizing the C&D waste as one of priority categories of waste and the need to create prerequisite for the efficient C&D waste management in the following period.

C&D waste is related primarily to the construction sector, but also to mining operations. The mining sector is the main provider of primary raw materials and products necessary for construction operations, and at the same time it has to manage large quantities of its own, very similar waste. Mining engineering in Croatia is mainly oriented to oil and gas exploitation, and non-metallic mineral resources exploitation. Approximately 200 companies are engaged in non-metallic mineral resources exploitation, 50% of which in the exploitation of aggregate resources, 20% in sand and gravel exploitation, 30% in the exploitation of clay, raw materials for cement and lime, dimension stone, quartz and sea salt production (Salopek and Bedeković, 2001). The consumption of stone material in the City of Zagreb amounts to 2.5 million cubic meters per year, which is approximately 25% of average annual production of stone material in the Republic of Croatia, 20% of which is provided from sources in the City of Zagreb and 60% from Zagreb county (Salopek et al., 2003). Taking into account the increasing pressure on mining operations due to environmental protection, the recycling of C&D waste presents a huge and insufficiently used potential.

**TABLE 1:** Quantities of collected C&D waste in the Republic of Croatia (Croatian Agency for the Environment and Nature, 2017).

| Year | C&D Waste (t) |
|---|---|
| 2006 | 275 323 |
| 2007 | 266 457 |
| 2008 | 194 456 |
| 2009 | 131 863 |
| 2010 | 362 567 |
| 2011 | 579 240 |
| 2012 | 717 382 |
| 2013 | 872 782 |
| 2014 | 761 312 |
| 2015 | 882 256 |

## 2. LEGISLATION DEVELOPMENT AND ITS IMPACT ON THE C&D WASTE MANAGEMENT IN THE REPUBLIC OF CROATIA

The C&D waste management in the Republic of Croatia is regulated by various strategic and legal documents and plans (Figure 1). Inadequate waste management was recognized as the major problem in environmental protection in the Republic of Croatia both by National Environmental Protection Strategy (Official Gazette, 2002a.) and by National Environmental Action Plan (Official Gazette, 2002b.). They predicted waste management crisis due to the increasing quantities of waste and insufficient infrastructure unless significant changes were introduced soon. Inadequate waste management system negatively affects the environment, particularly groundwater, which is the main source of drinking water and a basic national resource.

Waste Act from 2004 (Official Gazette, 2004) introduced waste management permits (collection, recovery, disposal) that companies had to obtain, and its amendments from 2008 and 2009 introduced the concepts of the C&D waste recycling yards and concession contracts for waste management operations based on public tenders.

Waste Management Strategy of the Republic of Croatia from 2005 (Official Gazette, 2005b) defined the framework for sustainable waste management in relation to the evaluation of the current status, basic objectives and measures for hazardous and non-hazardous waste management, and guidelines for waste recovery and disposal. In the part referring to the C&D waste, the Strategy aimed at 80% C&D waste recovery. It proposed the conversion of sanitary landfills to inert waste landfills, treatment and disposal of the C&D waste next to waste management centres with mobile or stationary treatment facilities. The total annual quantity of C&D waste was estimated at 2.6 million t. It includes 1 million t/annually of construction waste and 1.6 million t/annually of waste generated during road construction and development projects, tailings generated by the extraction of mineral resources, demolition waste and the waste resulting from military destructions. The construction waste was composed of 75% of excavation material, construction and demolition debris accounts for 15-25%, while the asphalt, tar and concrete account for 5 – 10%. It is mostly inert waste (95%): fragments of ceramics, mortar, plaster, broken concrete, iron, steel, metals, wood, plastic material, paper, etc.) (Official Gazette, 2005b).

Waste Management Plan in the Republic of Croatia for the period from 2007 to 2015 (Official Gazette, 2007) anticipated the possibility of pre-treatment of the C&D waste in transfer stations and treatment at existing landfills until waste management centres were established. Based on a concession contract, mobile treatment facilities would be used and concession holders would be responsible. Recycling was encouraged by proposing a fee for the acceptance of C&D waste for recovery ranging from 5 to 15 €/t, substantially lower than the waste disposal charge. Market principles will apply to the price of recycled aggregate.

Ordinance on construction waste management (Official Gazette, 2008) defines participants in the C&D waste management and their relations and obligations. According to the Ordinance, the recovery of C&D waste is carried out in mobile treatments facilities at its source, and in stationary treatment facilities in the C&D waste recycling yards.

Act on Sustainable Waste Management (Official Gazette, 2013) defined waste management sites and waste disposal charges in order to stimulate recovery/recycling and reduce disposal in landfills.

One of the last documents is the Waste Management Plan in the Republic of Croatia for the Period from 2017 to 2022 (Official Gazette, 2017a). It set out objectives for recycling i.e. material recovery by the year 2020 (70%). Another objective is to collect 75% of the amount of generated C&D waste.

In addition to that, the waste management system is regulated by various ordinances like: Ordinance on Classification of Waste (Official Gazette, 1996), Ordinance on Waste Management (Official Gazette, 2017b), Ordinance on Conditions for Waste Management (Official Gazette, 1997), Regulation on categories, types and classification of waste with a waste catalogue and list of hazardous waste (Official Gazette, 2005a), Regulation on conditions on the handling of hazardous waste (Official Gazette, 1998) and Regulation on transboundary waste movement (Official Gazette, 2006).

After its accession to the EU, Croatia assumed the obligations resulting from various EU directives related to the area of waste management. Waste disposal according to hierarchy (avoiding – reducing – recycling – energy recovery) with the development of appropriate network for waste collection and processing and "the polluter-pays principle" is prescribed by the Directive 2006/12/EC on waste (European Commission, 2006a). The classification of landfills and waste, the handling of waste intended for landfill and the types of waste to be accepted in the various classes of landfill, specific permits and the management of landfill sites are prescribed by the Council Directive 1999/31/ec on the landfill of waste (European Commission, 1999), while the issues of monitoring and the environmental impact assessment are regulated by the Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment (European Commission, 1985). Council Directive 91/689/eec on hazardous waste (European Commission, 1991) defines the properties of hazardous waste, prohibits mixing hazardous waste with non-hazardous waste, sets the procedures for the handling of waste already mixed with other waste, substances or materials, regulates drawing up plans for the handling of hazardous waste, permit issuing, record keeping etc. in the member states, and it is amended by the Council Directive 94/31/EC and Commission Decision 2000/532/EC. Council Directive 87/217/EEC on the prevention and reduction

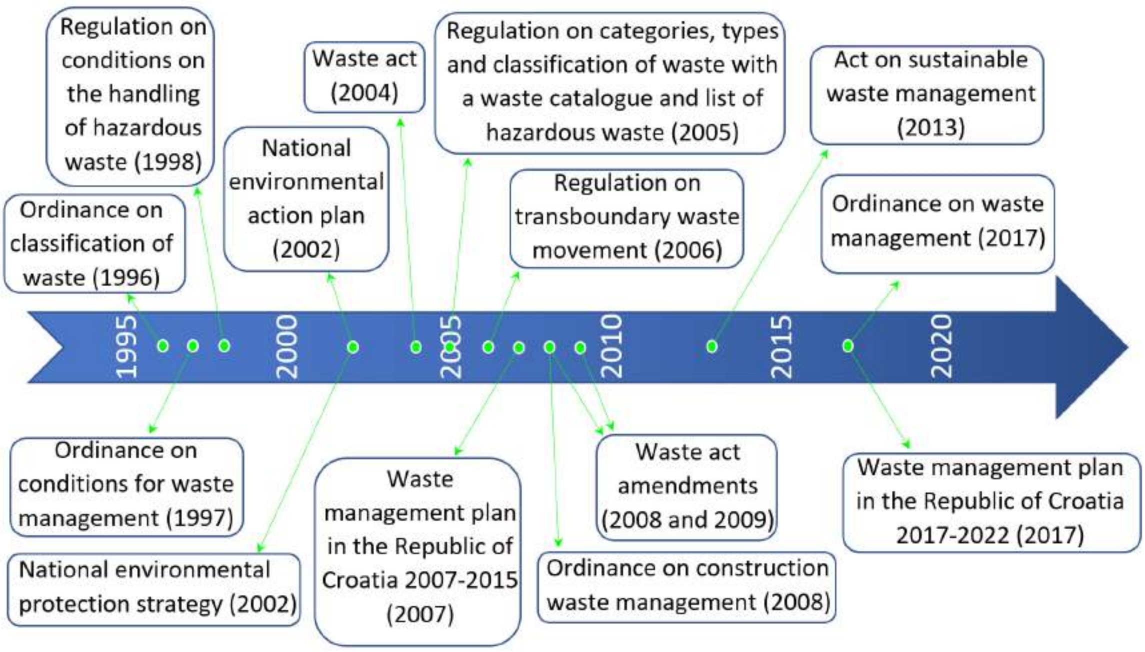


**FIGURE 1:** The timeline of the Croatia's legislation development.

of environmental pollution by asbestos (European Commission, 1987) joins supervision of emissions into the air and water, and defines procedures and conditions related to recycling and disposal of asbestos, as well as the monitoring methods for emissions into the air and water. It is amended by the Council Directive 91/692/EEC and Council Regulation (EC) No 807/2003. We also have to mention Directive 2006/21/EC on the management of waste from extractive industries (European Commission, 2006b) that gives guidelines, measures and procedures with the aim of minimizing negative impact on human health and environment in the waste management related to excavation, mineral processing and storage of mineral resources and the operations in quarries. This Directive sets the obligation of drawing up a waste management plan and its minimum requirements, and classifies waste facilities as Category A (presents a serious risk of major accidents in a case of a failure or incorrect operation), and Category B (all the other waste facilities).

## 3. C&D WASTE RECYCLING IN THE REPUBLIC OF CROATIA

The C&D waste recycling saves primary (mineral) resources and reduces the quantity of waste intended for landfill. It saves energy, therefore reducing emissions in energy production, and the result is a lower environmental impact (Ulubeyli et al., 2017). The process of recycling omits the phase of excavation of mineral resources and the other phases of a technological process are very similar. The EU recycles approximately 60% of C&D waste, but with huge differences among individual member states (from 10% to 90%). Five member states have met the objective of 70% re-use and recycling of the C&D waste set by the Directive 2008/98/EC (European Commission, 2008): the Netherlands 98.1%, Denmark 94.9%, Estonia 91.9%, Germany 86.3% and Ireland 79.5% (Ulubeyli et al, 2017). USA, South Korea and Japan have a high level of recycling, ranging from 70% to 95% (Huang et al., 2018). Compared to the aforementioned results, the Republic of Croatia has to invest additional effort. It is estimated that Croatia generates 2.5 million tonnes/annually of C&D waste, while only 7% of that amount is recycled (Bjegović, 2008). The largest part of a total amount is generated in the City of Zagreb (Table 2). Therefore the only stationary treatment facility is situated in the city of Zagreb, while in the other parts of Croatia recycling is carried out in mobile processing plants. The C&D waste in the City of Zagreb consists mainly of soil and stone (93%), metal (6%), mixed C&D waste (1%), concrete, brick, tiles and plaster material (up to 0,1%) (Bjegović, 2008). The recycling technology in the stationary treatment facility is determined according to the composition of generated C&D waste (Figure 2).

**TABLE 2:** Estimated annual quantities of C&D waste in the Republic of Croatia (Bjegović, 2008).

| Country | Estimated average annual quantities of C&D waste 2001-2005 (t) | Estimated average annual quantities of C&D waste 2006-2015 (t) |
|---|---|---|
| City of Zagreb | 140 175 | 581 297 |
| Split-Dalmatia | 138 865 | 283 767 |
| Primorje-Gorski Kotar | 113 329 | 246 131 |
| Istria | 76 544 | 166 242 |
| Osijek-Baranja | 109 164 | 112 422 |
| Zagreb | 78 992 | 111 026 |
| Zadar | 48 530 | 99 170 |
| Dubrovnik-Neretva | 36 798 | 75 195 |
| Vukovar-Srijem | 67 633 | 69 652 |
| Šibenik-Knin | 33 809 | 69 088 |
| Sisak-Moslavina | 47 285 | 66 461 |
| Varaždin | 47 127 | 66 240 |
| Slavonski Brod-Posavina | 58 384 | 60 127 |
| Krapina-Zagorje | 36 329 | 51 062 |
| Karlovac | 36 164 | 50 830 |
| Bjelovar-Bilogora | 43 956 | 45 269 |
| Koprivnica-Križevci | 31 746 | 44 621 |
| Lika-Senj | 19 911 | 43 245 |
| Međimurje | 30 206 | 42 455 |
| Virovitica-Podravina | 30 845 | 31 766 |
| Požega-Slavonija | 28 349 | 29 195 |
| TOTAL | 1 254 141 | 2 344 901 |

The construction and demolition waste recycling plant in Zagreb (Figure 2) consists of two parts: primary and secondary. The C&D waste is transported by trucks to the plants storage plateau and it is loaded to the storage bunker (1). Below the bunker, there is a mechanical feeder which doses waste to the grade (4) with interspace profiles of 400 mm. Oversize (grain size + 400 mm) should be comminuted by hydraulic hammer, while undersize (grain size -400 mm) goes to the rougher separation into (first) magnetic separator (8), where Fe is separated as a final product. Rest of the waste goes to a jaw crusher for the primary crushing (2). After the primary crushing Fe is separated in the scavenging 1 (second magnetic separator) (9) and the rest of the waste goes to an impact crusher for the secondary crushing (3). After the secondary crushing stage, Fe is separated in the scavenging 2 (third magnetic separator) (10) while the rest of the waste goes to the screening. The screening section consists of two stages. In the first stage, the waste goes to the vibration screen (5) with openings of 100x100 mm. This vibration screen (5) works in a closed circle with the jaw crusher, so screen oversize (+100 mm) go once again to a jaw crusher for crushing (2), while undersize (-100 mm) goes to the secondary screening stage which consists of two vibrating screens (6, 7). By adjusting a material flow router (11) it is possible to use two different vibrating screens in a secondary screening stage. If the material flow router (11) is set to use the first vibrating screen (6) with hole size of 63 mm, then grain size 100/63 mm (screen oversize) and grain size -63 mm (screen undersize) go to the dump as final products. If the material flow router (11) is set to use the second vibrating screen (7) then grain size 100/31.5 mm (screen oversize) and grain size -31.5 mm go to the dump as final products. An integral part of the recy-

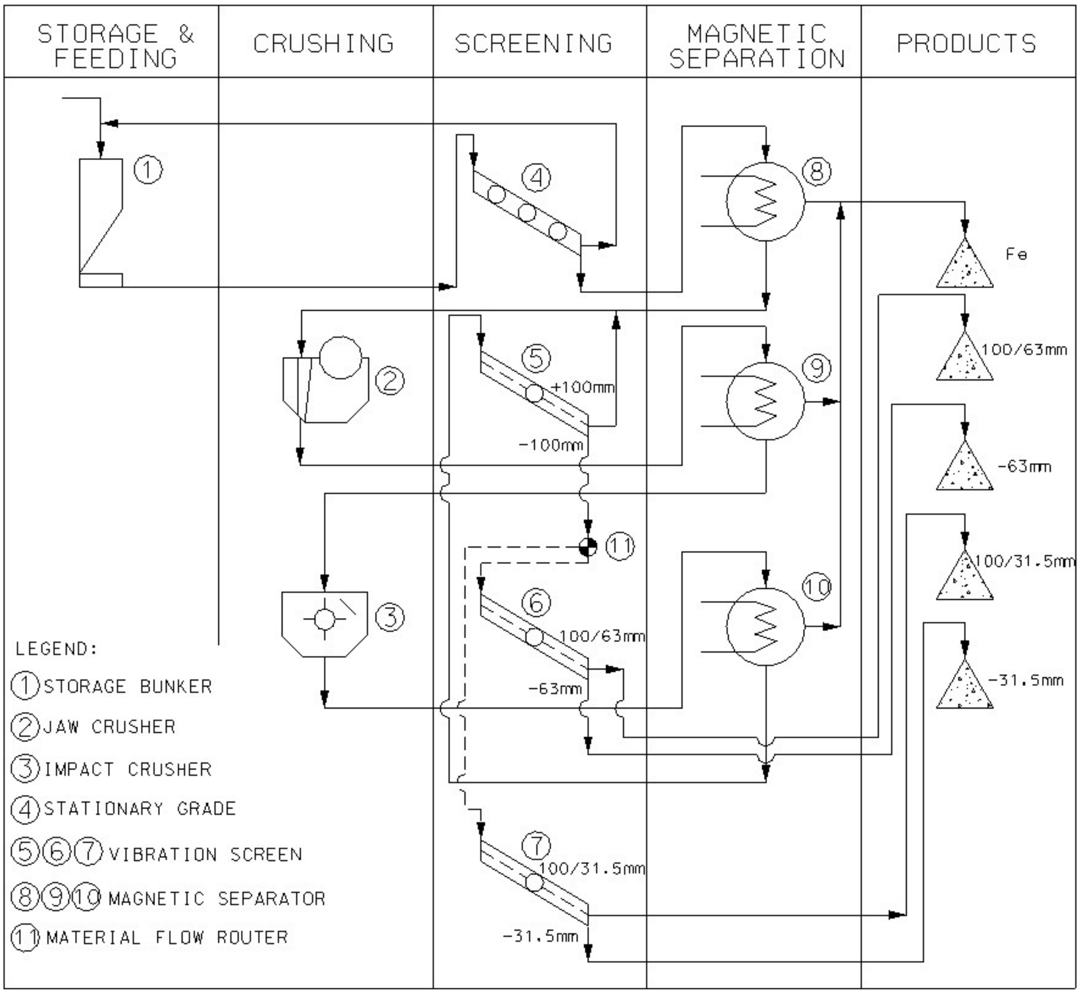


**FIGURE 2:** The flow sheet of the C&D waste recycling plant.

cling plant is a dedusting system. All belt conveyors are covered to prevent dust spreading into the surrounding area. Secondary, the impact crusher (3) and all vibrating screens (5, 6, 7) are included in the dedusting system. The polluted air from the crusher and vibrating screens is led by tubes into the bag filter for cleaning. The efficiency of the bag filter is more than 99% and the clean air is let out through the chimney to the atmosphere. Dust particles remain in the bag filter and are collected in the dust collector below the bag filter. In addition, the recycling plant is equipped with watersprinklers.

Inadequately developed C&D waste recycling is the consequence of insufficient knowledge, as well as the insufficient awareness of the importance and advantages of the use of recycled material in environmental protection and preservation of natural resources, low level of understanding of possibilities of use of those materials, lack of experience, underdeveloped market, inadequate quality standards, variable prices and technical regulations (Bjegović, 2008). A prerequisite for a successful development of recycling as an unavoidable segment of the C&D waste management is a legal, economic and technological framework.

The composition of C&D waste largely depends on the fact whether a new structure is being constructed or an existing one is being demolished. Furthermore, the geographic features of Croatia influence the composition of C&D waste. If the composition of C&D waste in four major regions of Croatia is observed, stone prevails in two of the regions (Dalmatia and Primorje), while in the east of Croatia the C&D waste mostly consists of bricks, i.e. mixed waste, while concrete and brick prevail in the northeast. The composition and quantity of waste partly influence recycling technology. The process may be simpler or more complex, depending on the raw material to be recycled, as well as on the planned use of the recycled material. The secondary raw material, obtained through C&D waste recycling, is mostly used as a material for load-bearing layer for roads and added to asphalt concrete mixtures and different types of concrete.

## 4. RESULTS AND DISCUSSION

On the basis of systemized data on collected C&D waste within LIFE05 TCY/CRO/000114 CONWAS project (Bjegović, 2008), the average quantity of C&D waste is calculated for all counties in Croatia for two previous periods (2001-2005 and 2006-2015).

The estimated quantities of C&D waste in the period 2006 - 2015 included a larger part of C&D waste at so-called uncontrolled landfills, and the intense demolition of old structures along the Adriatic coast in that period, as well as decreasing waste from the war affected areas. The quantities for the period 2001-2005 are based on the data obtained by questionnaires, and the actual quantities for the period 2001-2005 are definitely larger than the ones obtained through questionnaires.

Taking into account the current legal framework for C&D waste management, and based on the indicators of the Croatian Bureau of Statistics on the value of performed construction and demolition operations, a significant increase in the quantities of C&D waste is not expected in the following years. Based on the aforementioned data, it is estimated that the annual increase rate will amount to 1.5% by 2020. According to that estimate, a total amount of C&D waste will amount to 1,281,231 tonnes and non-hazardous mineral waste to 539,861 tonnes by 2020 (Štirmer and Škreb, 2017).

The aforementioned discussion points to the fact that all documents include "estimated" and not "actual" quantities of the C&D waste. Without high-quality data on quantities, composition, sources and waste streams it is not possible to plan and establish efficient logistics, appropriate technology for sorting and recycling, or to design landfills. Therefore, it is necessary to ensure continuous and high-quality data collection in order to draw up plans, strategies and to establish efficient C&D waste management. Collection and transport of C&D waste is mostly carried out by contractors with their co-operators, using their own equipment, and only a smaller part is carried out by authorized collectors and utility companies. According to the data of the Croatian Agency for the Environment and Nature, a total of 136 companies in the Republic of Croatia have a permit for the C&D waste management. Although the Ordinance on Classification of Waste prescribes registers for waste handling (type, quantity, source, handling) and transfer notes certified by a waste producer, collector and treater, the systematic monitoring of quantities, properties and waste streams has not been established yet, therefore the quantities of C&D waste can only be estimated.

Since the C&D waste is not always registered under the correct catalogue number, and some construction companies and quarries recycle C&D waste in their own facilities, and the C&D waste is occasionally disposed in uncontrolled landfills with other waste, it is not possible to determine actual quantities and composition of C&D waste. Despite relatively good legal framework and regulations, it is necessary to invest additional efforts into their consistent implementation in order to improve the current state. The importance to the issue of the C&D waste management is given by exceptionally large quantities of that waste that proportionally increase the problem in the case of inappropriate handling. Some of the problems still waiting to be solved are uncontrolled disposal of C&D waste, lack of separation system at the source or in sorting centers, mixing hazardous and non-hazardous waste, ignoring C&D waste issues during the design of the building and low level of communication between participants included in the C&D waste management process.

## 5. CONCLUSIONS

The importance of C&D waste management is the result of its large quantities compared to other types of waste, and its high potential for recycling. Additionally, besides the impact on the environment as a whole, plants and animals, the adequate waste management in the Republic in Croatia is important for keeping the international perception of the country of preserved environment and attraction as a tourist destination. In order to make waste management efficient, it is necessary to establish an efficient waste management system. A prerequisite for efficient waste management is an adequate legal framework. Croatia progressed a lot during the last 15 years, and after the accession to the EU, it assumed the obligations resulting from various EU directives. In addition to that, numerous studies and projects related to the C&D waste management issues have been carried out, but they are all based on estimated quantities of waste. The essential prerequisite for the efficient waste management is data on actual quantities, types and waste streams of the C&D waste. The registered quantities of the C&D waste at the county level, which are obviously lower than the actual quantities, emphasise the weakness of the system, particularly in less developed areas. Therefore, it is essential to invest additional efforts with the goal of high-quality monitoring of quantities, composition and C&D waste streams as a basis for planning technologies, plants and facilities for C&D waste management. Legal framework is relatively well established, but it is not entirely implemented, and we have to insist on its implementation in the future. In addition to the advantages of recycling compared to the exploitation of primary raw materials in the production of aggregates, the success of the C&D waste recycling will depend on its market value, i.e. the difference in the price of recycled and natural aggregates, as well as on the waste disposal costs. Taking into consideration the quantities of waste and inadequate infrastructure for C&D waste management, mobile treatment facilities seem to be the best option for solving the C&D waste management issues in Croatia.

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# BIG DATA IN CONSTRUCTION WASTE MANAGEMENT: PROSPECTS AND CHALLENGES

Weisheng Lu [1,*], Chris Webster [2], Yi Peng [3], Xi Chen [1] and Ke Chen [1]

[1] Department of Real Estate and Construction, Faculty of Architecture, The University of Hong Kong, Pokfulam, Hong Kong
[2] Faculty of Architecture, The University of Hong Kong, Pokfulam, Hong Kong
[3] School of Public Administration, Zhejiang University of Finance & Economics, Hangzhou, PR China

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## ABSTRACT

'Big data' has been rapidly sprawling in various research disciplines such as biology, ecology, medical science, business, finance, and public governance but rarely in construction waste management (CWM). The CWM community around the world generally relies on 'small data' collected via active solicitation such as sampling and ethnographic methods. This small data is intrinsically limited by its inability to account for the totality of CWM and research findings generated from the small data cannot be accepted with a high level of confidence. With the growing interests in big data, it can be reasonably expected that the waste management community will augment efforts to develop big data and its analytics. However, the efforts are currently constrained by the limited knowledge to do so. This research aims to provide a synoptic overview of the prospects and challenges of big data in CWM. It adopts an inductive, qualitative case study method whereby the empirical data is collected using an ethnographic–action-meta-analysis research approach and triangulated with data from literature, ongoing debate, and other sources. The paper offers some insights on big data acquisition, storage, analytics, implementation, and challenges. Although having a focus on waste management in the construction sector, the insights generated from this study can be of value to general waste management research, which suffers from the same problems of erratic and poor quality data as CWM.

## 1. INTRODUCTION

A consensus has yet to be reached on what is meant by 'big data'. According to Padhy (2013), big data is a collection of data sets so large and complicated that it becomes difficult to process using traditional data management tools. Likewise, Schönberger and Cukier (2013) proposed big data as "things one can do at a large scale that cannot be done at a smaller one, to create a new form of value". Researchers tend to adopt Gartner's three defining characteristics of big data, namely, volume, variety, and velocity, or the three 'Vs' (McAfee et al., 2012). Volume is the quantities of data in the forms of records, transactions, tables, or files; velocity can be expressed in batch, near time, real time and streams; and variety can be structured, unstructured, semi-structured and a combination thereof (Russom, 2011; Zaslavsky et al., 2013). Data is relentlessly generated from such sources as web logs, sensor networks, unstructured social networking, and streamed video and audio. Analytics have been developed to analyze big data in order to uncover hidden patterns, unknown correlations and other useful information that will guide better business predictions and decision-making (Shen et al., 2016); in effect, value is advocated as the fourth 'V'.

Notwithstanding the disagreement on terminology, big data has rapidly become the new frontier across a wide variety of fields, including biology, medical science, ecological science, business, urban planning, public governance, innovation, competition, and productivity. "Government agencies use big data to generate statistics, to help them understand local and global patterns and trends, in order to improve their services" (Shen et al., 2016). Using its ability to harness information in novel ways to create insights and services, big data can become a crucial source of innovation (Schönberger and Cukier, 2013). Through analyzing big data, researchers aim at identifying some 'latent knowledge' (Agrawal, 2006) or 'actionable information' (World Economic Forum, 2012), which can be utilized for future decision-making.

However, the euphoria of big data is yet to be seen in the waste management research community. This is particularly held in construction waste management (CWM), where research is suffering from notoriously erratic 'small' data. One explanation for this is the temporary nature of

* Corresponding author:
Weisheng Lu
email: wilsonlu@hku.hk


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construction projects (Senaratne and Rasagopalasingam, 2017), whereby once a project is completed it ceases to generate construction waste and the window of opportunity to collect the data closes. The data collection methods adopted by previous CWM studies involved sampling and ethnographic methods during construction processes, such as: direct observation (Poon et al., 2001); questionnaire survey (McGregor et al., 1993); sorting and weighing the waste materials on-site (Bossink and Brouwers, 1996; Kazaz et al., 2018); collecting data through consultation with construction employees (Treloar et al., 2003); tape measurement (Skoyles, 1976); and truck load records (Poon et al., 2004). These data collection approaches are widely perceived as costly, non-value added, and disruptive to the ongoing construction process. Hence, in practice construction companies are not obliged to record and report the characteristics of the waste generated (Fatta et al., 2003; Lu et al., 2017). Most studies have a relatively small sample size or sampled relatively small sites due to the difficulties of covering the whole population. As such, their data has long been limited by its inability to comprehensively represent the totality of waste generated throughout the construction process.

Nevertheless, with the vogue of big data in other disciplines, researchers have started to explore its applications to CWM. For example, Lu et al. (2015) revisited waste generation rates (WGRs) as performance indicators of CWM using big data, which allowed them to say with greater confidence that there is a notable CWM performance disparity between the public and private sectors (Lu et al., 2016a); Chen and Lu (2017) identified factors influencing demolition waste generation in Hong Kong through big data analytics; Bilal et al. (2016a) proposed a conceptual framework of big data architecture for construction waste analytics. However, the general sentiment is that understanding of big data in CWM is still rather superficial. There is a plethora of bestsellers and online articles eloquently promoting the use of big data, but impartial, skeptical insights preferred by researchers are rare. A succession of questions remains unanswered, such as 'What are the potentials of big data for CWM?'; 'Is there a definite size over which a dataset can be called big data?'; 'Will it be financially viable to purposely develop big data for CWM?'; and 'What are the main challenges of big data in CWM?'.

This paper explores the prospects and challenges of big data in CWM, with a view to facilitating pursuit of the research agenda related to big data and its analytics in the realm of CWM and beyond. The remaining sections of the paper report: the research methods used in the study, organized in the form of a CWM case study; a description of big data as a basis as used in our case study; a presentation of the results and findings; an in-depth discussion; and, conclusions. Although a particular big data set of CWM in Hong Kong is described, it is suggested that the analysis yields generalizable insights that are independent of this data set and the CWM setting per se.

## 2. RESEARCH METHODS

Since only a limited number of studies had been reported with this focus at this point in time, this paper adopts a mixed-methods approach with an inductive, qualitative case study (Yin, 1989) at the kernel of the research methodology. Unlike the stereotype that it may have the problems of generalization, case study approach is widely used in management research and considered useful to promote scientific development through deepening understanding of the context and relevant experiences. Over the past five years, the authors have endeavored to investigate CWM performance by taking real actions in acquiring and analyzing CWM big data in the specific context of Hong Kong. Several papers about CWM performance have been published. During the research, it is noticed that some of the insights of big data analytics can be drawn from the action research and contribute to the wider knowledge body of big data in a more general setting. Therefore, other members of the research team, with a strong humanity and sociology background, took an ethnographic approach to observe the "action researchers", e.g. how they collect the data, interacting with practitioners or other researchers. They observed and analyzed from a distance. They conducted meta-analyses of the published papers by "hovering" from the specific research findings on CWM performance but induced some propositions of the prospects and challenges of big data in CWM as a setting. They triangulated the propositions against new literature, ongoing debate, and finally form the insights that are generalizable to general waste management realm or beyond.

### 2.1 The data set

To effectively manage construction waste in Hong Kong, a Construction Waste Disposal Charging Scheme (CWDCS) was enacted in 2006 based on the polluter pays principle (Lu and Tam, 2013). According to the Scheme, a contractor should pay HK$125 per ton for non-inert construction waste that is accepted by landfills; HK$100 per ton of mixed inert and non-inert waste material received by off-site sorting facilities; and HK$27 per ton of inert construction waste material ending up in public fill reception facilities (Hong Kong Environment Protection Department - HKEPD, 2014). Under this Scheme, contractors must send their construction waste to the government-run facilities if not otherwise reduced, reused, or recycled. Every truckload of construction waste ending up in any of the facilities leaves a record with the HKEPD. Waste disposal facilities record information on every load of construction and demolition (C&D) waste received from every construction/demolition site. This practice leads to a database of more than one million transaction records in a year, which is considered a full coverage of the waste generated from all construction sites in Hong Kong. The Scheme also requires all contractors involved in C&D activities to open a billing account with the information of the activities also recorded by the HKEPD. These records form the account information database, which includes account number, construction name, category, site, and contract sum of all C&D projects in Hong Kong. A third database is information about the disposal facilities, which includes facility name, received waste type, and facility address, and a fourth database is the information of all the vehicles, including their license plate number and the permitted gross weight they can carry. The links between the four databases are shown in Figure 1.

The three defining characteristics of big data, i.e. volume, velocity, and variety seem evident in the data set. The data is of considerable volume. The main database contains more than 6 million well-structured waste disposal records, recording almost every truck load of C&D waste generated from construction sites and disposal at the designated CWM facilities over the past six years. Although the total physical volume only slightly exceeds 700 megabytes, we argue this is 'big data', given it is a well-structured data set that may contain much more meaningful information than the same volume of messy, raw data. The data has significant velocity. The data in the main database is incoming as a rate of about 4,000 records per day. In addition to the rich data fields exhibited above, the variety of the data set is still expanding, e.g. by collecting more details on new, renovation, or demolition projects from the government Buildings Department (HKBD) and linking green building information publicly available in the Hong Kong Green Building Council (HKGBC) and other potential databases to the main databases in the future (See Figure 1). Therefore, volume, and velocity and variety are all significantly high and dynamic in this data set.

### 2.2 Obtaining and analyzing the big data

To obtain the data, the research team approached the HKEPD through its general inquiry service, followed by emails clarifying the specific data requested and what it will be used for. After the initial request had been granted, the HKEPD advised the research team that it would be more convenient to obtain further data from its themed website where data is updated every fortnight. For security reason, the data was stored in the cloud data service of The University of Hong Kong (HKU) and mapped on the hard disk drive of two computers for further use. Use of the data is governed by general research ethics and HKU's policy on the management of research data and records.

Over a period of five years the research team conducted a series of studies to analyze the acquired big data using various statistical analyses and data mining, with results published in journals or shared at international conferences. This study used these research experiences as a case study to extract the general prospects and challenges of big data in CWM. In addition, preliminary findings from the case study were triangulated with the literature of big data in other fields. The following sections present the critical reflections from this study.

## 3. ANALYSES AND DISCUSSIONS

### 3.1 "In God we trust; all others please bring data"

The famous quotation is widely attributed to Edwards Deming to reflect his fundamental principle of using data to back up any decisions in production or business. It further reinforces the truism about the importance of data to scientific research, where quantification is believed to generate high forms of knowledge in social sciences (Shelton, 2017). Unlike other pollutants such as dust and noise, C&D waste is easy to see, as well as relatively easy to measure (Formoso et al., 2002), albeit not so easy to sample systematically. No other method is more reliable than directly measuring construction waste generation in order to obtain primary data. However, contractors are usually not mandated to record waste generation data on site and generally perceive doing so as disruptive and as not adding value to the ongoing construction process. Contractors therefore record waste generation sporadically, if at all, and so it is not feasible to expect them to be the source of such data.

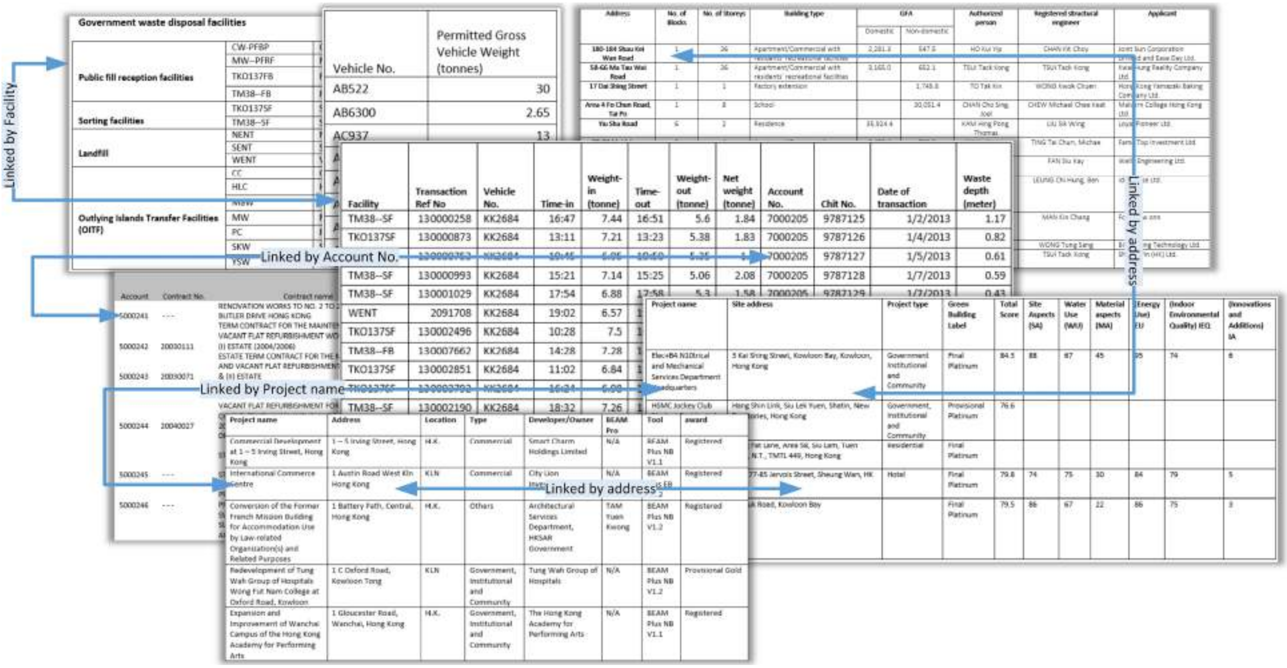


**FIGURE 1:** Links between the databases relating to construction waste management in Hong Kong.

An alternative method of data collection is for researchers themselves to measure tangible waste generation. Research assistants can be dispatched to construction sites to collect primary data by recording actual waste generated, e.g. measuring cement waste, counting bricks ordered, left, and wasted, or checking the materials used against orders. Given the fact that a construction project will last for a relatively long period of time ranging from a few months to years, it is impractical for these research assistants to station on a site to record all the waste generation data relating to a project. Instead, sampling is often adopted to make onsite inspection more tractable, e.g. measuring the waste generation from a typical section or a floor. Aside from the limited data collected, the big issue with sampling is whether the data are sufficiently comprehensive and representative, i.e. the issue of sampling frame and method. Because each construction project is unique, in principle, there is no a priori method or theory for constructing a sampling frame and each sampling effort is thus necessarily ad hoc.

The situation has changed little since Lu et al. (2011b) described a sampling-based data collection approach: "When a trade had finished, the site manager cordoned off an area of the construction site to facilitate the on-site measuring exercises. The area (usually a room plus a section of common walkway) was selected as being representative of a typical floor so that the WGRs derived from that area could be applied to the whole floor." There are two points of weakness with such approaches. First, if too few sampling sites are selected, the estimate cannot be treated as accurate, even if it is an appropriately chosen site. Second, if several or many sites are chosen, then if they are not sufficiently representative of the whole construction process, the estimate would be systematically biased. Neither the degree of accuracy nor the reliability of the measure and the derived WGRs are too variant to be generalized to other projects. Katz and Baum (2011) and Lu et al. (2016b) noted that most previous studies on CWM, even though using objective methods such as weighing waste onsite, had a relatively small sample size or sampled relatively small sites due to the difficulties involved in conducting a full coverage survey, whatever 'full coverage' might mean. These studies are thus limited in their ability to account for the totality of waste generation throughout the construction process, and as a consequence, their results cannot be accepted with a high level of confidence.

In view of the ongoing emphasis on the importance of data to CWM research, it can be expected that researchers and construction companies themselves, will intensify efforts to collect more reliable and representative data (Bilal et al., 2016a). With better sensing and recording technology, CWM systems are expected to emerge that no longer rely on sampling. This is analogous to other big data domains in which routinely sensed and stored data are replacing occasionally collected data. For example, occasional surveys of shoppers at supermarkets to obtain customer profiles has been replaced by data collected at electronic points of sale. We are moving to an era in which the researcher's task is not so much to sample from the real world but to sample from a database that is a complex and voluminous model of the real world. A CWM big data source will provide something approximating full coverage rather than a sample of the population of interest. With continuing advances in data acquisition technologies and the lowering of data processing costs, collecting big data is becoming ever more feasible. In the future, it might be required that "all others please bring big data" to the CWM community. Kitchin and Lauriault (2015) echoed by contending that big data in the future will become as common as small data is in today's research. It is therefore expected that big data regarding CWM in other regions and countries would emerge, although they currently have no such structured databases of CWM big data as the one reported in Section 2 of this paper.

### 3.2 Size does matter

There are two basic premises behind the exhortation of investing in big data. First, the large volume of big data can alleviate the potential bias inherent in small data and provide a fuller picture so as to have a closer claim of objective truth (Bilal et al., 2016a). Second, by analyzing big data it is possible to discover hidden patterns, unknown correlations and other useful actionable information that will help with devising more informed CWM approaches. With its characteristics of volume, velocity, and variety, analyses of big data can lead to actionable information that would not be possible to discover with small data. Our mining of CWM-related big data in Hong Kong illustrates these points.

WGR is widely accepted as a CWM performance indicator, which is calculated by dividing waste generation in volume (m3) or quantity (tons) by per m2 of gross floor area (Poon et al., 2004) or by per million US$'s worth of construction work (Lu et al., 2015). The lower the WGR, the better the CWM performance. Without readily available secondary data relating to waste generation (e.g. volume or quantity of waste), researchers have to use sample and ethnographic methods to collect the data from the project to calculate WGR, as described above in Soibelman (2016) and Lu et al. (2011b). However, in this study, every truckload of C&D waste generated from all the construction sites over the past six years was recorded by the HKEPD. The WGRs (ton/mHK$) of all Hong Kong's 4,062 sites are plotted in Figure 2. Every dot in the figure represents a project with its WGR calculated by summing the truckloads of waste and then dividing that figure by the contract sum of the project.

Using a sampling method, researchers would only be able to collect a limited portion of the dozens or hundreds of loads of waste as depicted in Figure 2, with the burden to justify whether the sample represents the holistic waste generation pattern of the project. This leads to a situation similar to the ancient parable of "blind men and an elephant" – the researcher is at the risk of probing into only a small scope of actual waste generation from all the construction projects. This leads to uncertainty over whether the data collected is sufficiently comprehensive and representative.

Size definitely does matter in this case because big data can portray a fuller picture of C&D waste generation, and the calculated WGRs converge to a range. This is supported by the law of large numbers: the average of the results obtained from a large number of trials tend to become convergent to a certain value as more trials are performed (Sen and Singer, 1993; Shen et al., 2011). The advantage of big data over small data allows more in-depth analyses of WGRs. Given the abundant data covering the waste generation from all the projects, it might be considered rigorous to average the WGRs and derive a mean to represent the general CWM performance. However, after having calculated and plotted the frequency distribution WGRs of all the projects (see Figure 3), it was found that the distribution is far from a normal one but rather a heavily skewed distribution. The median of the group of WGR, 15 t/mHK$ is much lower than that of the mean of 76 t/mHK$ (see Table 1). Using the mean to represent the general CWM performance is thus very misleading, which however is a common problem in existing CWM research with small data. Without big data covering the whole population, this insight would have been difficult to discover.

Bigger data size also allows some hidden patterns, unknown correlations and other useful information to be discovered (Zhou et al., 2016). For example, by analyzing one day's waste disposal records randomly selected from the 6 years' pool, it is discovered that a considerable number (734 out of 4780) of waste haulers tend to overload than their permitted load weight (See the red dots in Figure 4). Transporting the waste is charged by trips and it is often costly, sometimes costlier than the waste disposal charge itself. Tracing individual lorries may reveal the ones that are consistently involved in this overloading as an unsafe behavior so that they can be more closely monitored or possibly be subjected to legal action. Meanwhile, as shown by green dots in Figure 4, often lorries are underloaded (the

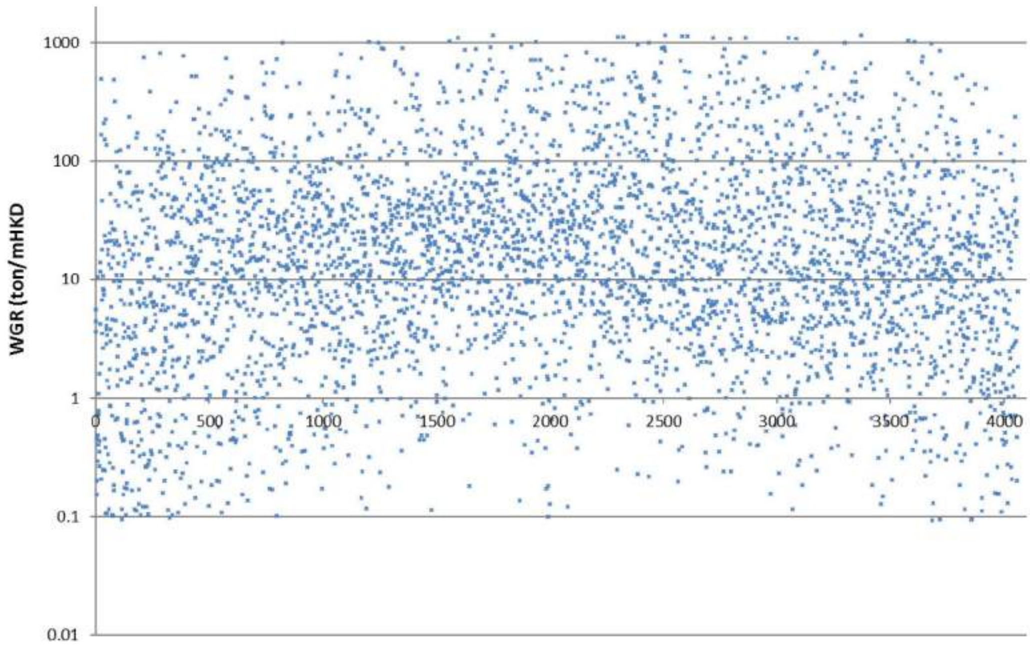


**FIGURE 2:** WGRs of the individual projects (Sample size=4,062).

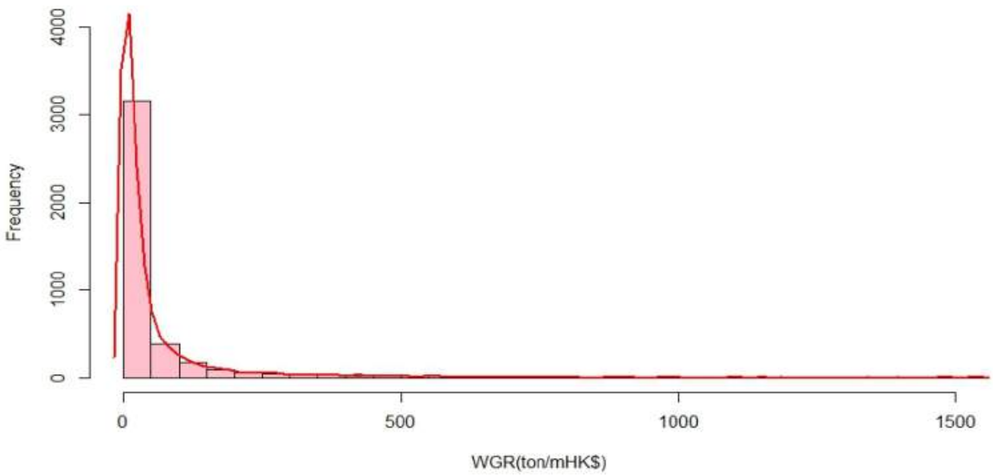


**FIGURE 3:** Frequency distribution of WGRs of all projects (Sample size=4,062).

**TABLE 1:** Means, standard deviations (SD), and medians of WGRs of the projects.

| Projects | Sample size | Mean (t/mHK$) | SD | Median (t/mHK$) | Range (t/mHK$) |
|---|---|---|---|---|---|
| Overall | 4062 | 76 | 192 | 15 | 0.13~1793.33 |

lower the point, the more underloaded a lorry is), which is more likely than due to poor fleet management. Likewise, by further analyzing the WGRs of the individual projects, it is found that a handful of companies achieved consistently low WGRs, such as Company A in Figure 5. Perhaps these companies are truly good at managing C&D waste, in which case their experiences should be disseminated to the whole industry. On the other hand, the WGRs of Company B are consistently high suggesting that a review of the company's poor performance might be advisable. This kind of useful actionable information can only be revealed with big data.

### 3.3 How big is big data? The relativeness of big data

There is a misconception among the CWM community that to be considered big data, a dataset should be in

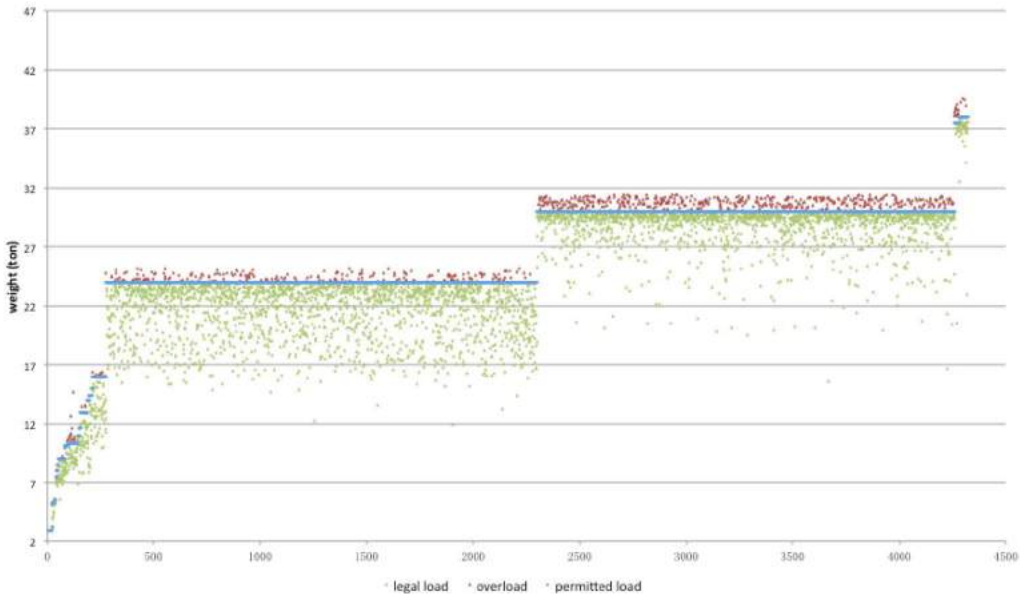


**FIGURE 4:** Pattern of overloaded or underloaded of waste haulers in one day (Sample size=4,780 truck loads).

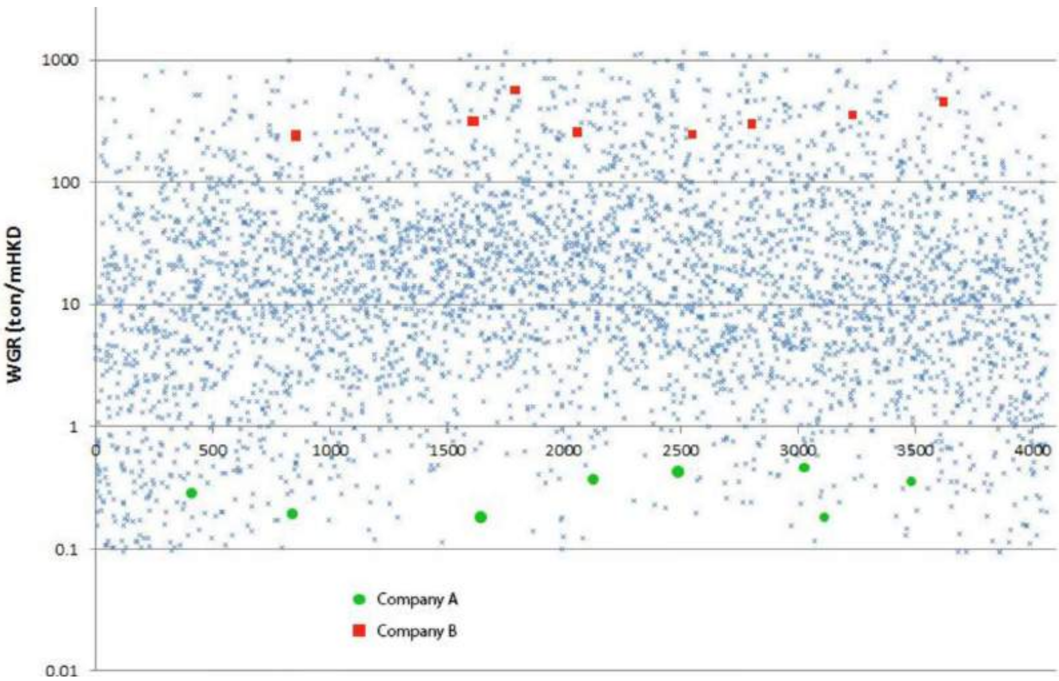


**FIGURE 5:** Unusual construction waste management performance using big data analytics.

terabytes or petabytes. This misconception is evident in debates in international conferences, comments from journal paper reviewers, and reviewers of research grant applications. These commentators and reviewers at times simply judge that some datasets are not big data on the basis of the data's size in electronic format (i.e. megabyte, gigabyte, or the like). Press (2013) asked, "Is there a definite size over which data becomes big data? How big is big data?"

It is argued that big data is a relative concept. In particular, big data is time relative. A dataset that appears to be massive today will almost surely appear small in the near future (MIT Technology Review, 2013). Current big data may be considered small data in the future due to the rapid development of technology, particularly in the area of cloud-based data storage and retrieval. Here, the CWM data compared to the 1990s is 'big', and the data in the 1990s was considered big data compared to the 1970s. Big data is also user relative. A dataset treated as big data by one entity may be considered 'small' by another depending on its intended use. For example, the CWM dataset in Hong Kong may be treated as big data by researchers interested in construction management, urban planning, or transportation, since it can provide many useful insights. However, it may not be considered as big for the purpose of either estimating total construction waste generation in China. Some researchers (e.g. Sivarajah et al., 2017) thus highlighted the intricacy of a dataset as a significant factor in determining whether it is big. Big data does not necessarily always mean better data (Taylor and Schroeder, 2015). Leek (2014) pointed out that in general the bigger the sample size, the better, but that meaningful sample size and raw data size are not always tightly correlated (Akter and Wamba, 2016). Large datasets from Internet sources are often unreliable, prone to outages and losses, and errors and gaps are magnified when multiple datasets are used together (Boyd and Crawford, 2012). There is also a lot of noises in this CWM dataset that must be excluded (Lu et al., 2015). Data cleansing helps detect and correct incomplete, incorrect, inaccurate or irrelevant parts of the raw data and allows users to perceive a dataset's true size, value, and relevance to a particular research inquiry.

It can be concluded that although size does matter, there is no definitive size at which a dataset can be called big data and it most definitely does not mean that a dataset must be in the volume of a terabyte or petabyte. The point is to examine the dataset's ability to account for the totality of the subject under investigation and determine whether it allows values to be created that could not be arrived at with data on a smaller scale. Conference discussants and paper reviewers should not take the automatic position that the data smaller than terabytes or petabytes is definitely not big data. One more useful definition of big data suggested by this research so far, is that big data can be perceived as a data model of a system, its dynamics in particular, that can be analyzed in totality or by systematic or random sampling to identify and interrogate trends. The 'big', in this definition, refers not to absolute size of data stored but to the degree of coverage of the data model vis-a-vis the system being represented.

### 3.4 Big data analytics: applied statistics vs data mining

Another misconception found in the big data literature is that big data analytics is equated to 'pattern finding algorithms', 'unattended machine learning', 'deep learning', 'artificial intelligence', NoSQL database, Hadoop, and other fascinating methods (Bilal et al., 2016b). Traditional applied statistics, as Leek (2014) argued, has been largely left out in the discussion. Traditional statistical analyses can be used for multiple purposes, e.g. describing the nature of the data, exploring the relation of the data, creating a model, proving or disapproving the validity of a hypothesis, and so on (Moses, 1986). Arguably, one of the purposes of sophisticate data mining techniques is to search and structure a large and unwieldy data base in such a way that makes possible the use of traditional statistical methods designed to formally describe data in ways that are scientifically well understood.

Data mining is useful for automatically discovering valuable information from a large collection of data and transforming it into organized knowledge (Han et al., 2012). Rather than simply locating, identifying, understanding and citing data, data mining serves as a computational process where patterns in large datasets can be discovered (Clifton, 2010). Other approaches such as pattern finding algorithms and unattended machine learning are also useful, although they have been over stated by the media to such an extent that it gives the illusion that they are the only approaches appropriate for exploiting the value of big data. It is the experience of the research team, when mining the CWM-related big data in Hong Kong, that purely relying on machine intelligence is ineffective at best. Predictions and human intervention can save a large amount of computational time and increase the effectiveness of data mining. As a formalized procedure it is advisable to plot big data, using 'data visualization' methods, before engaging in any data mining techniques (Kostelnick, 2007). The intention is to observe potential patterns and provide a direction for the subsequent data mining. This is essential a human intervention process.

Both traditional statistics and data mining are indispensable means of harnessing the power of big data, although there are challenges with both techniques. A typical challenge is to select statistical indicators to interpret the results (Ekbia et al., 2015). Traditionally, in small data analysis, p-value is commonly used in the context of null hypothesis testing to indicate the statistical significance of evidence. If the p-value is less than or equal to the chosen significance level, either 5% or 1%, the test suggests that the observed data is inconsistent with the null hypothesis, so the null hypothesis must be rejected. Although this method is disputed (Goodman, 1999; Wasserstein and Lazar, 2016), it is commonly used as a license for making a claim of a scientific finding or implied truth in numerous fields including CWM (Wetzels et al., 2011). However, in very large samples, p-values go quickly to zero, and solely relying on it can lead the researcher to claim support for results of no practical significance (Lin et al., 2013). In this context, data interpretation and discussion become more sophisticated and care is need in making claims.

Researchers should also keep a wary eye on the data saturation, which is under explored in big data analytics. Data saturation is generally used to refer to the process of gathering and analyzing data to the point at which no new insights are added (Wray et al., 2007). In analyzing waste haulers' transportation behavior, one day's data was randomly selected and plotted (Figure 4). When the analyses were gradually extended to more days, it is noticed that the patterns are largely stable without new insights added (see Figure 6), i.e. it has reached a point of data saturation. With really big data the computational power, energy and time savings can be very high. Leek (2014) suggested that in big data analytics, it is best to define a metric for success up front and stop wasting resources when the data is saturated. It is an analogous issue to the question of parsimonious sample size in small-data research. For example, a political polling researcher will rarely sample more than 2000 voters under normal expectations of the distribution of votes, since the reduction of the standard error of the estimate beyond that number is tiny compared with the cost of surveying additional people. Data saturation challenges the orthodox view that the bigger the data, the better.

### 3.5 "The gold mine" to be protected or to be shared?

Currently, many big data sets are left over unintentionally when businesses are done (Ekbia et al., 2015). For example, they are created as by-products of people travelling around, communicating using smart phones, or purchasing from supermarket or through e-commerce. Likewise, the Hong Kong CWM big data set is a by-product of measuring and monitoring CWM flows. The amassed data can be a corporate asset, the mining of which allows companies to make better business predictions and decisions. Big data is thus like a gold mine; researchers and data analysts gather around potentially rich sources like a 'gold rush'. Since it is incidentally created and describes natural business processes and captures revealed behavior, big data tends to be considered better than experimental data or simulation data as it potentially contains more ground truth with respect to social reality than traditional instruments (Hand, 2015). Big data portrays a fuller picture of a subject matter, which allows for a stronger claim to objective truth; as Anderson (2008) put it, "with enough data, the numbers speak for themselves". Researchers and data analysts are therefore abandoning carefully curated small data and are rushing to discover big data sources to exploit. It can therefore be predicted that data owners will become more protective of their big data and reluctant to share their gold mine with others. Facebook accumulates big data from its users but only a few individuals have free access to it. Some companies restrict access to their data entirely, others sell access for a fee, and others offer small datasets to university-based researchers (Boyd and Crawford, 2012). The big data on CWM in Hong Kong was granted to the research team for free as the request was made at a time when big data was not as highly sought after as it is today.

The open data movement around the world may offset the effects of this trend to a certain extent by calling for big data to be openly available. Open data is the idea that some data should be freely accessible to everyone to use and republish as they wish, without restrictions from copyright, patents, licenses or other mechanisms of control (Auer et al., 2007) exerted by both public and private organizations. The movement argues that these restrictions are at odds with the communal good and hence data should be made available without restriction. For some public organizations, such as the United Nations, the World Bank, statistics bureaus, or government agencies, it is their obli-

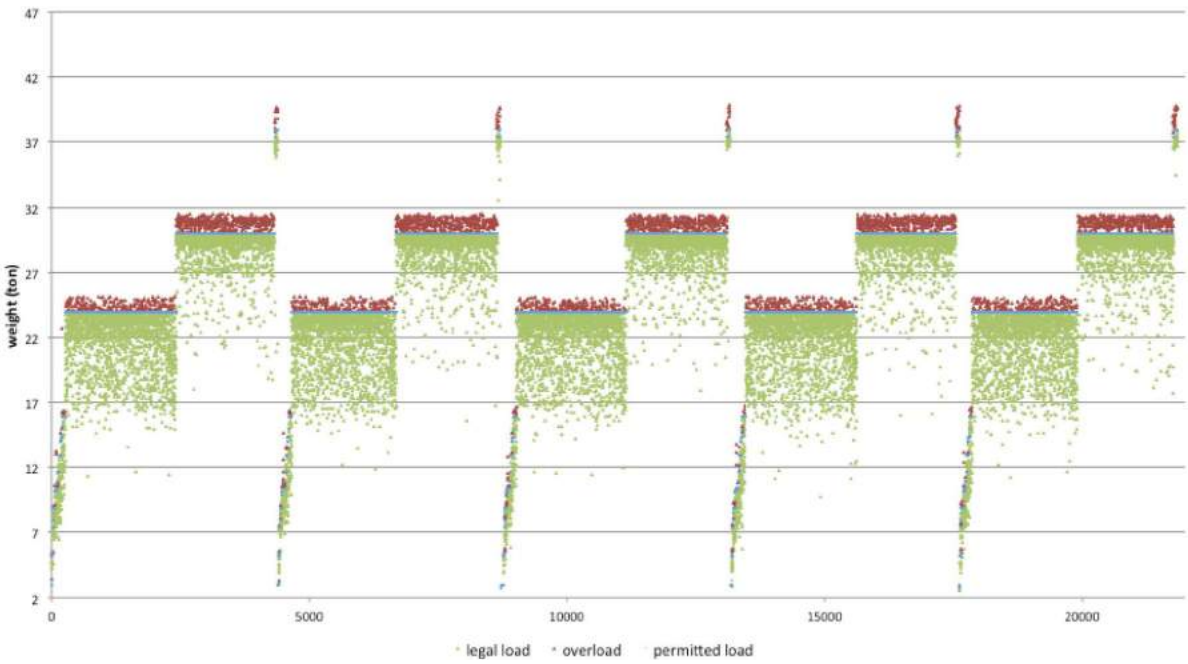


**FIGURE 6:** Pattern of overloaded or underloaded of waste haulers in five consecutive days (Sample size=20,841 track loads).

gation to make their data available to the public. These can all be sources for researchers to form big data. To enrich the Hong Kong CWM big data set, publicly available data from the HKBD and HKGBC were accessed and linked to the CWM process data. Enormous efforts went into data searching, slicing, stitching, and cleansing; a process similar to putting jigsaw pieces together. Manual interventions, string matching algorithms, and address geocoding were all employed to link the various data sets together.

### 3.6 Proactive big data strategies, "developing a mine to mine?"

Currently, many big data sets are largely left not intendedly, but discovered a 'gold mine' by mining which many meaningful, previously uncovered findings can be revealed (Terranova, 2000). Many institutions have adopted proactive strategies to develop big data. For example: statistical organizations such as Eurostat, United Nations Economic Commission for Europe, have formulated their big data roadmaps (Kitchin, 2015); Chicago has launched the 'Array of Things', a connected network of sensors that will be deployed throughout the city to collect data on environmental factors such as air quality, noise, and climate, which can then be used to discover hidden problems and develop targeted policies to improve city life (Thornton, 2015); and Barcelona deployed responsive technologies across urban systems including public transit, parking, street lighting, and waste management, which are intended to yield significant cost savings, improve the quality of life for residents, and provide better urban governance (Adler, 2016). The foregoing is essentially "developing a mine to mine", which is possible with increasing accessibility to ubiquitous and affordable sensing and communication technologies. An example relating to CWM in Hong Kong is probing into the behavior of waste haulers. The CWM big data captured for this study did not include the time when haulers left a site or their behavior en route to waste disposal facilities. Now, in addition to Radio Frequency Identification (RFID) technologies (Lu et al., 2011a; Flanagan et al., 2014), smarter technologies have been developed and embedded in the lorries to track their geographical positions (Niu et al., 2016; 2017). Such proactive big data strategies are similar to finding the missing pieces of a jigsaw puzzle.

Adopting proactive big data strategies raises many data platform design issues. For example, where to install the data capturing and communication devices such as sensing, RFID, laser scanning, webcam, and wireless. CWM researchers have suggested that new technologies should be unintrusive to ongoing construction processes otherwise their use is doomed to failure (Niu et al., 2017). Although data capturing and communication technologies are more accessible than ever, they still incur significant costs that need to be optimized against the value of data collected. In addition to the capital cost of installation, there is the ongoing cost of maintaining and renewing the data infrastructure. The lifecycle cost can be formidable, particularly where the system monitors performance over a large city. Furthermore, proactive data collection relies on a network of devices where the malfunction of one device could potentially cause the whole system to fail. Data infrastructures therefore need to be designed with a high degree of resilience.

### 3.7 Last but not the least: a little touch of big data ethics

The capture and use of big data has both benefits and risks. Ever since the advent of big data, there has been concern over the ethical ramifications of data analysts misusing its power, e.g. Facebook's data privacy scandal in March 2018. Although the conceptual, regulatory, and institutional resources of research ethics have developed greatly over the past few decades and have become familiar to researchers, there are always many unaddressed issues with respect to the ethical implications of the big data phenomenon (Boyd and Crawford, 2012). Existing norms governing data and research ethics have difficulties accommodating the special features of big data. The ethics of using big data are intimately tied to questions of ownership, access and intention, all of which are often disputed. Social media such as Facebook claim to own their big data and have exclusive access to it, even though the data itself is actually contributed by their users. It is also problematic for researchers to justify their actions as ethical simply because the data are accessible, let alone respond to the accusation that "limited access to big data creates new digital divides" (Boyd and Crawford, 2012).

Consent, in particular informed consent, premised on the liberal tenets of individual autonomy, freedom of choice and rationality, has been the cornerstone of personal data regulation and ethics (Cheung, 2016). However, it becomes impossible to ask researchers to obtain consent from every waste hauler who left the data passively as a part of the business process. Traditional de-identification approaches (e.g. anonymization, pseudonymization, encryption, or data sharding) to protect privacy and confidentiality and allow analysis to proceed are now problematic in big data, as it is the power of big data analytics that even anonymized data can be re-identified and attributed to specific individuals (Ohm, 2009). For example, analyzing the CWM big data can tell which companies performed well in CWM and which did not. De-identification is not always helpful, one can re-identify the companies which left their records in other databases, e.g. the one in the HKBD. Researchers thus need to start thinking more clearly about accountability of big data analytics; identifying methods, predictions and inferences that can be considered ethical, and those that are not.

The big data revolution has seen its ramifications including a series of ethical issues as listed above, none of which is resolvable with an easy answer. Metcalf et al. (2016) suggested that to get a grasp of the ethics of big data requires theorizing big data as something more than a technological artifact. The law is a powerful element in big data ethics, but it is far from able to handle the many nuanced scenarios that arise; organizational principles, institutional statements of ethics, self-policing, and other forms of ethical guidance are also needed (King and Richards, 2014).

## 4. CONCLUSIONS

Big data has rapidly become a game changer in many research realms, including waste management. Using the big data in construction waste management (CWM) in Hong Kong as an inductive case study, this study provides a synoptic overview of the prospects and challenges of big data in CWM. It is argued that big data, in comparison with small data collected from sampling and ethnographic methods, can portray a fuller picture, so that research findings from the big data can be accepted with a higher level of confidence. It is also illustrated that big data analytics can reveal hidden patterns, unknown correlations and other useful information to better inform CWM decisions.

Given the advantages of big data and the increasing availability of routinely-collected data, it is likely that big data will be a standard requirement for CWM research in the near future. However, it is also expected that data owners will become more protective of their big data for reasons of profit, privacy, or security. Some of the main issues using publicly available data have been reviewed, which will probably have to be the source for CWM research in the future. Hong Kong's CWM data, similar to its counterparts in other areas such as social media, e-commerce, or retailing, is left over unintentionally. Given the value of big data, it is expected that many researchers will take proactive strategies to collect big data. This is particularly opportune nowadays as data acquisition and communication technologies are becoming increasingly accessible. The findings provide references for big data in CWM in other regions and countries through specifying the prospects and challenges regarding data collection, analysis and applications.

There are misconceptions that prevail in big data research, one of which is in relation to the definitive size over which a dataset can be called big data. This paper argues that there is no definitive size and that the criteria should be whether the data is able to account for the totality of a relevant subject and whether it allows values to be created. It is suggested that academic arguments and positions in respect what big data is and is not are less important than understanding what large data sets can and cannot do. While many researchers are eager to explore the value of big data, both data mining and traditional applied statistics face challenges in dealing with its volume, velocity, and variety, and there are some researchers who do not consider big data research as representing scientific inquiry. Also, bigger data does not necessarily mean better data and researchers are advised to have a comprehensive and impartial understanding of the phenomena before embarking upon research that involves it.

Although this study has provided many interesting insights, they are just the tip of an iceberg. There is a massive agenda of big data for CWM researchers. Further research along these various lines will drive a shift from a theory-driven to data-driven regime investigations and from searching for correlation and causality to correction. The finer-grained the sensing technology and procedures underlying the database, the nearer to real-time will be the correctional options. As CWM systems are better modeled and understood, researchers will be able to move from basic descriptive analysis to behavioral analysis of CWM. It is also recommended that future studies should take an in-depth look into proactive big data strategies and big data ethics, neither of which has been fully deliberated in this paper.

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# SANITARY LANDFILL COSTS FROM DESIGN TO AFTERCARE: CRITERIA FOR DEFINING UNIT COST

Alberto Pivato [1,]*, Salvatore Masi [2], Diego De Caprio [3] and Anna Tommasin [1]

[1] *ICEA, Department of Civil, Environmental and Architectural Engineering, University of Padova, via Marzolo 9, 35131 Padova, Italy*
[2] *University of Basilicata, School of Engineering, viale dell'Ateneo Lucano, 10, Potenza, Italy*
[3] *Regione del Veneto, Area Tutela e sviluppo del territorio Direzione Ambiente - Unità Organizzativa Ciclo dei rifiuti, Palazzo Linetti, Calle Priuli, 99, 30121 Venezia, Italy*

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**ABSTRACT**

Landfill costs have been estimated for an average case that represents a reference landfill model for Northern Italy. This case is based on the analysis of more than 15 landfill projects per the requirements of the Italian legislation. These projects were based on the suggestions of the national landfill guidelines and best practices. Costs have been analysed through the four phases of the landfill lifetime (i.e.: design and authorization, construction, operation, and aftercare) and expressed per unit of waste volume in order to define the "landfill gate fee" that represents a unit payment for the whole landfill life cycle service. The estimated value, equal to 86.04 €/m$^3$, is in line with the analysed references and it is mainly attributed to the costs of leachate disposal, the staff, and raw materials for liner construction. However, other factors such as localization, climatic conditions, waste quality, landfill geometry, operative procedures, and financial and legal aspects can significantly influence the landfill gate fee. Finally, the results of this study, based on a large data analysis, represent a starting point for a comparative economic analysis to support political and technical decisions.

## 1. INTRODUCTION

Italian (art. 179 of D. Lgs. 152/2006) and European norms (art. 4 of Directive 98/CE/2008) that regulate the prevention and management of waste considers sanitary landfills at the bottom of the hierarchy which includes the following strategies: prevention, preparation for re-use, recycling, other recovery (e.g.: energy recovery), and disposal. In this regulatory framework, sanitary landfills in Italy can only accept pre-treated waste with the exception of few specific cases (art. 7 of D.Lgs 36/2003 and ISPRA, 2016).

However, sanitary landfills can properly meet waste management objectives while ensuring higher safety standards and operating flexibility compared to other final waste destinations. In fact, a landfill can operate within a wide range of potentiality (tons of waste per year) without requiring substantial structural changes and without significantly increasing emissions.

The sanitary landfill technology responds better than any other system to waste composition variability and it can be used, without any particular problems, to dispose of waste fluxes with different chemical compositions.

These characteristics make the landfill an indispensable component of integrated waste management systems that need to meet the following requirements:

- High fluctuations and seasonal peaks of waste streams production, as is the case of tourist areas or exceptional/occasional waste disposals;
- High temporal variability (e.g.: ten years) of the disposal requirement;
- Heterogeneous waste fractions from different waste recovery and stabilization treatments.

The sanitary landfill still represents a reliable technology in areas where waste production is very low and/or where significant distances discourage waste transportation to local facilities. It can represent the only solution for developing countries who find it difficult to implement more complex and often more expensive plants.

However, we have to not think that a landfill is a low technology that is easy to manage and does not require complex engineering components and high-technology materials.

Landfills can ensure that safety standards are implemented that are equal or higher than other waste management technologies at lower costs. Costs thus becomes the

* Corresponding author:
Alberto Pivato
email: alberto.pivato@unipd.it


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"independent variable" whose value is used to select the best option.

An appropriate estimation of landfill costs needs to consider the following key elements:

- Site factors: permeability of the soil; geotechnical stability of the walls (angle of natural stability); morphology and presence of excavations (quarries); presence of roads; presence of network services in the area (power lines, drainage system, etc.);
- Climatic and environmental factors: rainfall intensity (average value and seasonal variability of precipitation); temperatures and potential evaporation; intensity and persistence of winds; exceptional presence of birds and other animals; etc;
- Overall geometry and dimensions of the landfill: area and height of landfills; daily quantity of disposed waste; weekly or seasonal peaks in disposal; planned stops of plant operations;
- Presence of other management utilities: waste water treatment plant; biogas recovery units; etc;
- Operational activities: the rate at which the landfill is filled; costs for daily cover/restoration; etc;
- Financial aspects due to investments and taxes (landfill tax and contribution for landfill environmental annoyance);
- Land acquisition.

In addition to the above factors, regulatory technical requirements (minimum thickness of the clay layer, type of the geomembrane, the composition of the top cover, etc.) contribute significantly to the formation of landfill costs.

The main aim of this study is to define the landfill "gate fee" (typically per tonne) for the whole life cycle service. This analysis can be useful for many purposes such as:

- Defining the disposal costs for official price lists;
- Planning the best waste management system;
- Evaluating the economic impact of innovative technical interventions: energy crops application on top cover (Garbo et al., 2017; Lavagnolo et al., 2016); leachate recirculation; in-situ aeration application (Raga et al., 2015); new environmental compensation or mitigation applications (Pivato et al., 2013; Kunreuther and Easterling, 1996); etc;
- Improving the knowledge of the entire life cycle cost of a consumer good (from procurement of raw materials to disposal);
- Analysing the impact of the reduction of specific taxes or financial guarantees when innovative interventions are applied to the landfill to reduce long term emissions;
- Comparing the cost of the different landfill phases (construction, operation and aftercare) and their optimizations through the application of design alternatives.

## 2. MATERIAL AND METHODS

Costs along the landfill lifetime depend on several factors, strictly correlated to the design of the landfill itself. In this view, key points regarding the landfill type (for inert, non-hazardous or hazardous waste, according to the EU and Italian laws Council Directive 99/31/EC and D.Lgs. 36/2003, respectively) and the quantity and quality of the disposed waste (e.g.: moisture content, biodegradable fraction). Their influence on costs is significant, since they affect the constructive aspects (e.g.: total landfill height, planned landfill lifetime), management aspects (e.g.: use of pre-treatments, waste compaction, in-situ leachate treatment), required capital investment (e.g.: area acquisition, machinery, devices, materials, energy), labor force employed (e.g.: number of staff workers and managers), financial expenses, insurance policies, and taxes. Site planning is also fundamental in the cost definition. For instance, the construction costs for a landfill in a mountain area will be much higher compared to an equivalent one built on plain land, or, in some cases, in hill land, without significant physical constrains.

The definition of a "representative" landfill case becomes a key issue in order to generalize the results as much as possible and, therefore, to avoid the analysis of a specific landfill, which could result too much restriction for comparison and too much site dependency.

In this study, cost estimations are referred to as an average case that represents a reference model landfill for the Northern-Italy region. These estimates are based on the analysis of more than 15 landfill projects per the requirements of current legislation (D.Lgs. 36/2003) as well as suggestions of national landfills guidelines (CTD, 1997; DGR n. X/2461/2014) and on best practices.

Landfill cost and benefit analyses were performed throughout the four phases of the landfill lifetime (design and authorization, construction, operation, and aftercare) using reference unit prices from official price lists and market surveys.

Finally, all costs were grouped into economic or functional categories and correlated to the available waste volume of the landfill, which were assumed as a reference unit. In this view, it is important to note that landfill cost analysis usually refers to the total price ratio to the weight of waste disposed and not to the volume. The reasons behind this choice are explained in the following paragraphs. Figure 1 shows the methodology adopted in this work.

### 2.1 Model landfill definition

The statistical analysis of several case studies, mainly from the Veneto and Lombardia regions, were used to develop the definition of the geometry (volume and surface) and the most important landfill constructive and operational characteristics.

The model landfill is defined as a non-hazardous waste and underground landfill developed in a pre-existing excavation of an exhausted borrow pit (as 60% of the investigated landfills are underground); therefore, construction and management characteristics rely on these basic assumptions.

The total landfill volume after settlement is about 1,000,000 $m^3$, which corresponds approximately to a height of 27 m (17 m underground, 7 m above the ground level and almost 3 m considering both temporary and final top covers) and a landfill surface of 50,000 $m^2$ at the ground level. The volume related only to waste is about 800,000

$m^3$, and therefore daily, temporary, and final top covers are not included in this value. Four hydraulically independent sectors are considered and the bottom liner and top cover are defined in accordance with D.Lgs 36/2003 and Lombardia regional landfill guidelines (DGR n. X/2461/2014). The fixed operation time is 10 years and the aftercare period is 30 years.

The dimensions of the devices needed for leachate, landfill gas (LFG), and rainwater management rely on calculations obtained by the implementation of specific models. In particular, for leachate production estimation, a hydrological balance model was performed (Canziani et al., 1989):

- L = P – E, for the operational phase;
- L = P – R – ET, for the aftercare phase.

Where: L is leachate production, P is precipitation, E is evaporation, R is runoff and ET is evapotranspiration. The terms are estimated by the implementation of appropriate models.

Precipitation is estimated by taking the monthly average data (10 years monitoring) provided by the Villafranca (VR) weather station located approximately in the center of the area covered by the statistic.

The evaporation term relies on the application of the Turk formula and the evapotranspiration on the Thorntwaite formula (Canziani et al., 1989). The runoff refers to the work of Blakey (1992).

The most important results obtained from the application of this model are:

- Maximum yearly leachate production: 5,549 $m^3$/year in operational phase;
- Yearly leachate production in aftercare: constant and equal to 2,027 $m^3$/year;
- Cumulative leachate production: 42,540.31 $m^3$ in operational phase and 60,826.15 $m^3$ in aftercare, on a total of 103,366.46 $m^3$ in 40 years (see Figure 2).

Leachate collection is performed by a leachate drainage system, made of a drainage gravel layer on the bottom liner and slotted pipelines. The pipelines are arranged in four sectors and it consisted of a primary pipe and secondary pipes. The latter organized in a herringbone pattern. The pipelines slope is assumed to be 2%. The leachate is then collected from the South side of the landfill and delivered, through submersible pumps, to the leachate collection pipe (the removal pipe) and finally to the leachate storage system. The leachate is stored in three 100 $m^3$ fiberglass tanks. An extra tank with the same dimensions is assumed as a reserve. The tanks are placed on a concrete-made containment basin of 420 $m^3$ (internal length of 21 m, internal width of 8 m, and internal height of 2.5 m). The containment basin represents a safety measure in case of a failure in the fiberglass storage tanks. Leachate tanks are emptied periodically, on a schedule defined in

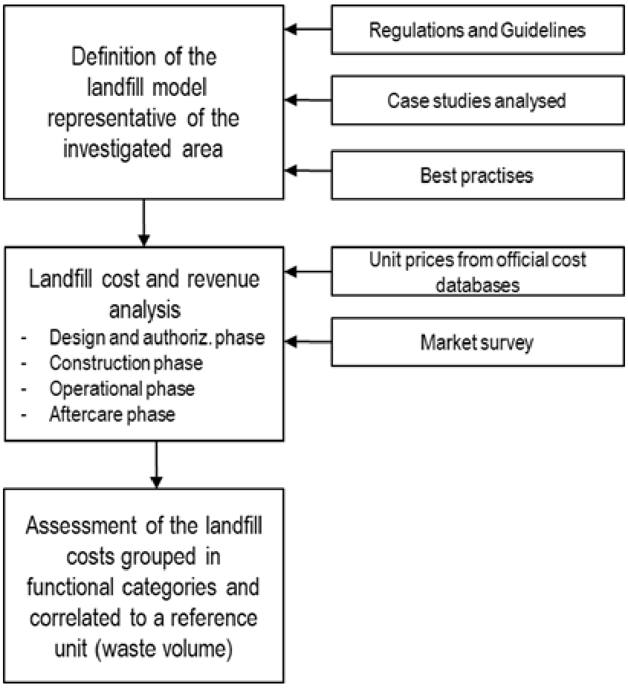


**FIGURE 1:** General scheme of implemented procedures.

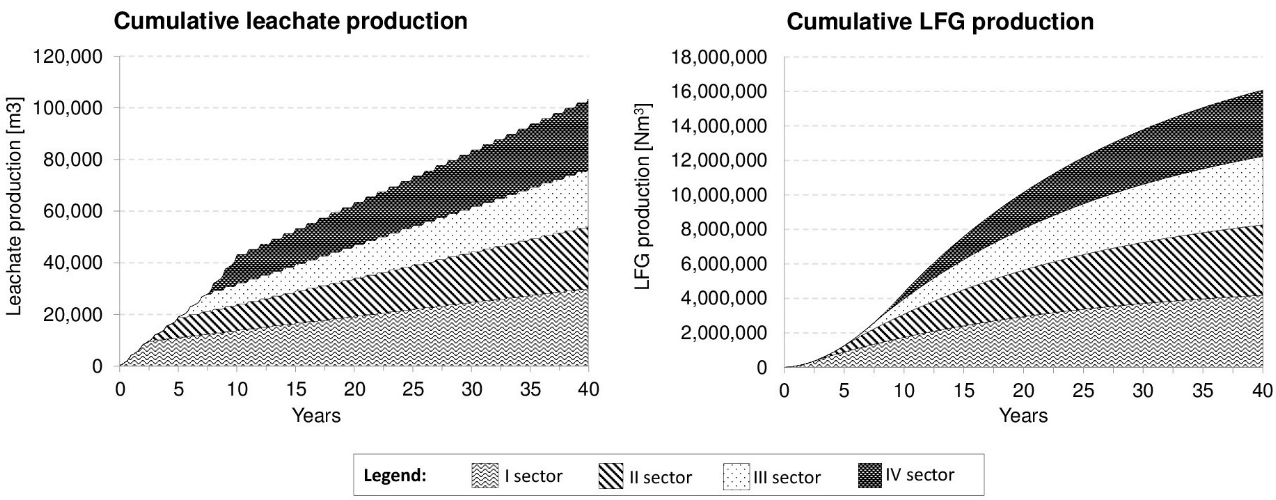


**FIGURE 2:** Cumulative leachate and LFG production over 40 years (10 years of operational phase and 30 years of aftercare phase).

accordance with the specific leachate production period. The leachate is then sent to an external treatment plant. No in-situ leachate treatment is assumed.

The estimation of LFG production is based on the work of Cossu et al. (1992). The methane content in the LFG was assumed as 50% and the LFG collection efficiency was assumed to be 70% of the total produced (DGRV n. 995/2000). The estimated values are reported in Figure 2 and summarized as follow:

- Maximum yearly LFG production: 743,034 $Nm^3$/year occurring in the 11th year;
- Cumulative LFG production: 3,955,642 $Nm^3$ in the operational phase and 12,025,537 $Nm^3$ in aftercare, for a total of 15,981,180 $Nm^3$ over 40 years.

The LFG collection system consists of 42 vertical wells (external diameter 600 mm, internal slotted pipe 160 mm) installed in the landfill. Each well has a specific maximum influence radius according to its location: 20 m near the border and 35 m in the center of the landfill. Wells are linked to four regulation stations, which are connected to the extraction station. Because of the low quantity of LFG produced, energy recover is not worthwhile (DGRV n. 995/2000). A flare is included and is designed in accordance with D.Lgs. 36/03 and CTD (1997).

The landfill was built in an area of about 94,500 $m^{2,}$ which includes the auxiliary buildings and services needed for its proper operation and management. These buildings and services include: leachate storage system, LFG extraction station and flare, internal roads, and service area. The service area (3,950 $m^2$) includes temporary storage, an office building, car and truck parking areas, a tire washing system, and a platform scale. The dimensions of these areas are defined through specific calculations, which considers possible law restrictions and other similar designs solutions.

The service area rainwater drainage system was designed using a climatic curve for a return period of 50 years. The first rain falling in the service area, represented by the first 5 mm of precipitation, is collected and treated in the first rainwater treatment system. The treated water is then stored in an underground water storage tank with a volume of 31.5 $m^3$. This water storage tank also receives rainwater from the service building roof. The stored water is reserved for the tire washing system and other uses, including the fire prevention system. The excess rainwater is released into a rainwater drainage trench located close to the first rainwater treatment system. This service area drainage trench is 18 m long and it has a trapezoidal shape (larger base 3 m, shorter base 2 m and height 2 m). The wastewater from the service building is treated with a grease separator and an Imhoff tank. The residual wastewater flow is then sent to a phytodepuration system with a surface area of 20 $m^2$ that was obtained by assuming 4 p.e. (population equivalent) and 5 $m^2$/p.e (ISPRA, 2012).

Rain falling on the landfill top cover is collected by a perimeter channel (trapezoidal shape, larger base 1.5 m, shorter base 0.5 m, height 0.5 m). In this case, according to D.Lgs 36/2003, this design relies on a climatic curve for a return period of 10 years. Rainwater collected is then released into another drainage trench, with the dimensions of the service area drainage trench except with a different length. This trench is 110 m long and located along the East side of the landfill.

Eight people are assumed to work at the landfill during the operational phase: an operations director, a technician responsible for the plant, two technical-administrative employees, three workmen, and a supervisor.

Figure 3 shows the landfill model as projected with the the aforementioned services, the leachate and LFG collection system configuration, and the defined liner system.

The characterization of the disposed waste was assumed as follows (percentages are expressed on wet weight base): paper 1.5%, cardboard 1.5%, glass and inert 52%, plastic 12%, metals 3%, stabilized inert 15%, and sludge 15%. These values refer to the residual and pre-treated fractions and are consistent with those assumed in the investigated cases. However, when designing a landfill, the waste characterization is not a simple straightforward definition and sometimes it is estimated without any further lab investigations. It is also difficult to know the temporal and spatial dynamics related to the quality and quantity of the incoming waste. For example, the European Union's Directives are working to limit landfilling and avoiding any recyclable material being landfilled. Consequently, it is expected that the amount of recyclables will decrease in future landfills which could increase the expected life of the landfill but also could modify the assumptions made regarding the waste composition entering the landfill.

Today, most of the waste received by non-hazardous waste landfills is represented by special waste produced by economic activities due to the difficulty to fill landfills only with MSW.

In Italy, the analysis of non-hazardous special waste production data (Laraia, 2017) by economic activity highlights that the construction and demolition industries represent the highest percentage (42.3%). Waste treatment and recovery activities follow (27.2%) along with manufacturing (19.2%), which include the quantities resulting from mechanical biological treatment of MSW. All of the remaining activities (e.g.: services, trade, transport; administrative, education and health services; agriculture, forestry, hunting and fishing; mining and quarrying; electricity, gas, steam and air; water supply; sewerage) accounted for 11.3% of the overall production of non- hazardous waste.

Sector experts estimate that in these landfills MSW does not exceed 30% of the total waste disposed. According to this assumption, the classification of waste disposed in the landfill model is 30% MSW and 70% Special Waste. It is also assumed that the waste characterization already takes into account this waste type ratio.

In the waste characterization assumed, unsorted and putrescible waste are not included as waste fractions categories. However, biodegradability is considered in the other fractions: paper and cardboard are considered slowly biodegradable and the sludge fraction is considered highly biodegradable. The choice not to define a separate category for putrescible material, comes from the landfill disposal trends over the last years. As a matter of fact, over the last

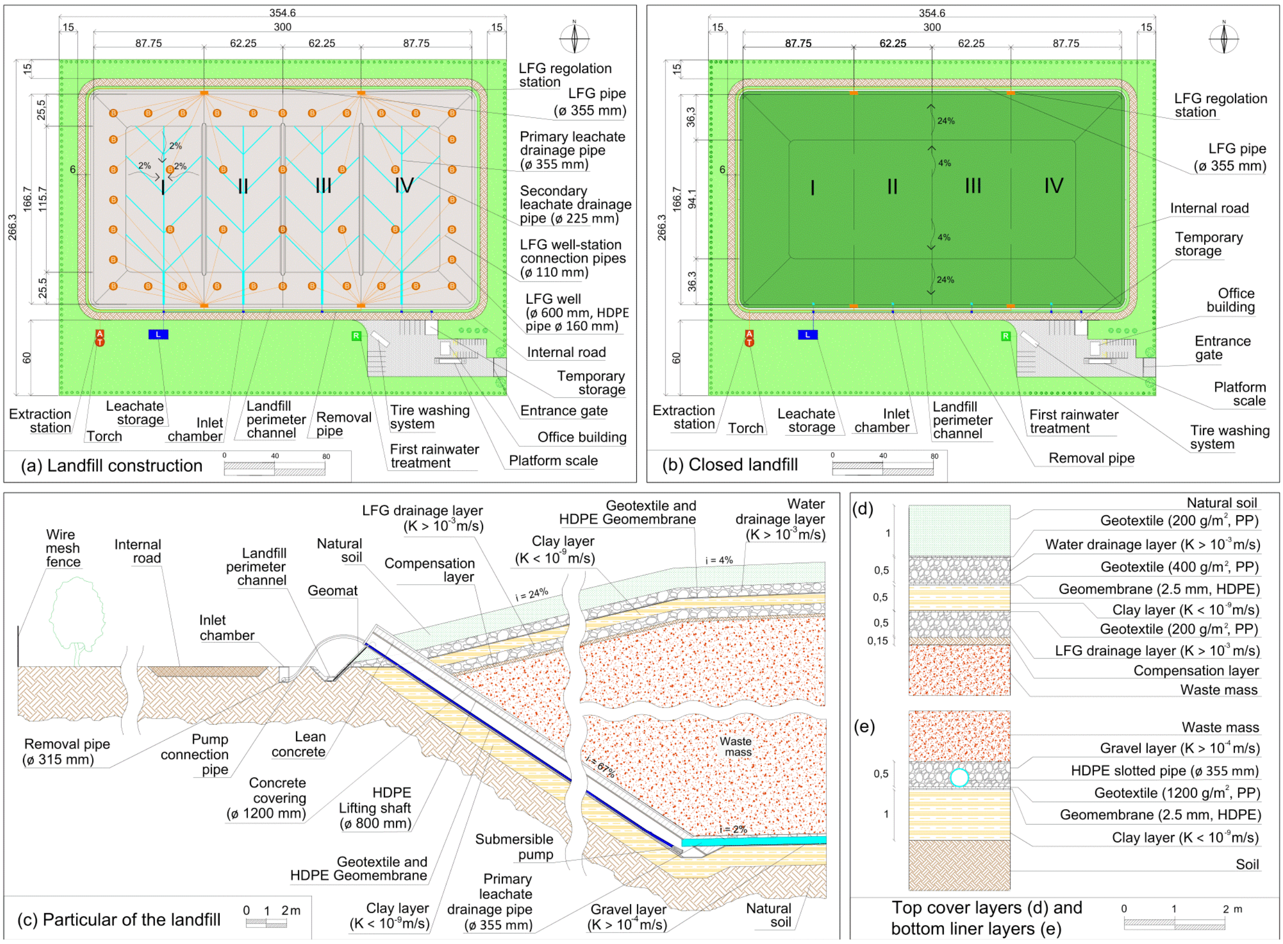


**FIGURE 3:** Detailed landfill model schematic

15 years the increased restrictive legislation targets caused a significant reduction in the biodegradable fractions disposed of in landfills. In Italy, the greatest reduction of 78% was achieved in the Veneto Region (DGRV n. 1245/2016). The biodegradable fraction disposed into the landfill, calculated as kilograms of biodegradable waste per inhabitant per year, decreased from the value of 133 kg/year/inhabitant in 2002 to 29 kg/year/inhabitant in 2014 (Annex A, DGRV n. 1245/2016). It is important to remember that, these data only refers to the MSW portion of the waste disposed. Therefore, in the case of non-hazardous waste landfills the real value of biodegradable waste disposed may be higher, since it also includes the fraction from special waste.

Waste characterization is also strictly connected with waste density and indirectly to the degree of utilization of the landfill. Waste density could be calculated considering single waste fraction densities however this would be an oversimplification of reality because its value is reflective of site specific conditions and dependent on time and the processes occurring during the landfill lifetime. In the present study, in order to follow the goal of generalization of the landfill model, a waste density of 1.1 t/m$^3$ is assumed. This value aligns well with the other case studios considered. Hence, considering the total volume of the waste disposed, the waste inflow calculated is 88,000 t/year.

## 2.2 Sanitary landfill life cycle

The landfill costs must consider all the landfill life phases:

- Design and authorization phase (1-3 years): waste characterization study, projects, construction management, testing and security coordination, meetings;
- Construction phase (1-2 years): site operation (excavation, backfilling of soil, etc), construction of the main parts of landfill body (barrier layer, leachate and LFG collection) and construction of other facilities at the landfill site (such as monitoring system, internal road and office building);
- Operational phase (10 years): operation of the landfill, for example the transport of waste from the outside, the placement and compaction of waste, the daily coverage, the environmental monitoring (groundwater, surface water and air around the plant), and leachate and LFG management;
- Aftercare phase (30 years): operations planned for this phase consist of monitoring and maintenance activities, which are mainly: top cover maintenance and monitoring, leachate collection system operation and maintenance, LFG collection system maintenance and monitoring, LFG migration control and monitoring, groundwater and surface water monitoring, security and ground stability maintenance.

A proper estimate of the duration of each phase is fundamental and is importance especially from an economic point of view. In fact, costs of each phase should be referenced and homogenized to a specific time frame (temporal homogenization of the costs). Further considerations of this particular subject are discussed in the following section.

### 2.3 Criteria for economic analysis

The economic analysis for the determination of the landfill gate fee is based on the technical procedures used in the bill of quantities and in the financial plans.

Regarding the bill of quantities, a generic item of cost/profit is determined applying the following expression:

Item cost [€] = Quantity [reference unit] x Unit price [€/(reference unit)]

The cost of this item refers to the cost of the specific intervention or operation considered (e.g. activity, equipment, material) and the reference unit of each item is selected each time following the most suitable unit of measure (e.g. $m^3$, $m^2$, m, t, body). In this work, quantities assumed are those defined above for the model landfill, whereas unit prices come from the official price lists of the Veneto (Veneto Region Price List, 2013) and Lombardia regions (Regione Lombardia, 2011) as well as from market analysis.

Regarding financial plans, interest rates for investments commonly applied in public projects were used.

Specific assumptions were applied for the following items:

- **Actualization of cost and revenue**
  Cost and revenue of activities/works realized at different times should be actualized. However, in real cases assumptions are made to consider unit costs equal for all of the landfill lifetime. This is also the case for the present study.
- **Tender discounts**
  Tender discounts for public contracts should be clarified. For those generally in question, average discounts for engineering services without work execution is 38.2% with peaks of 70%; for engineering services with work execution the average is 16.3% with a peak of 50.6% (CNI, 2016).
  Generally, these discounts don't depend on technical variables and are difficult to forecast. However, if they are applied, often the reduction in cost is compensated by other costs, such as unexpected events, design errors, or appraisals. In the work, these discounts were not considered.
- **Reference units**
  Costs usually refer to the unit of weight of waste disposed. Reasons are mainly practical, since the quantity of waste disposed in a landfill is measured by weighing the trucks. However, referring to unit of volume better takes into account the waste volume which avoids the influence of waste quality and type.

For example, the landfill cost referred to as a unit of weight of glass swarf (specific weight greater than 1.6 t/$m^3$) is double relative to the cost referred to normal MSW (0.8 t/$m^3$ after compaction), even if they occupy the same volume in the landfill; glass swarf however does not generate biogas or release pollutants in the leachate. In this sense, the cost correlated to weight results are restrictive.

Therefore, in accordance with the above considerations, the costs of the model landfill refers to the unit of volume. The volume considered as reference is the volume for waste equal to 800,000 $m^3$.

The weight/volume reference issue is similar to the concept of degree of utilization of the landfill (or utilization factor of the volumes). It represents the ratio between the quantity of waste disposed and the volume. Therefore from a dimensional point of view, the unit of measure is t/$m^3$. This dimensional similarity to density must not be misleading, because the meaning is different. Several factors affect the degree of utilization of the landfill. Besides the specific landfill site characteristics and the quantity and quality of waste disposed, the most important are:

- Density: the increase in the waste density value allows taking full advantage of landfill volumes. Considering the same volumes, higher density values mean larger quantity of waste that can be dispose of;
- Waste pre-treatments: they can maximize the usable landfill volume. Compaction maximization became fundamental, especially for some materials for which it is difficult to reach large compaction levels (and then high values of density) after disposal in the landfill. This means that, for these materials, good compaction before disposal allows for some kind of landfill volume "recovery". This concept fits particularly well to the case of plastics, where the density can reach 0.9 t/$m^3$ with a compactor roller, and it is around 0.4-0.5 t/$m^3$ with simple mechanical shovel. In general, the compaction density has a physical limit, which cannot be exceeded even with stronger compaction. In case of the aforementioned plastic fraction, the maximum density values can reach close to 1 t/$m^3$, while it is around 2.5 t/$m^3$ for glass swarf, 7.5 t/$m^3$ for ferrous waste, 2-2.5 t/$m^3$ for stones inert, and 1.5-2 t/$m^3$ for combustion residues;
- Settlements: this factor strongly influences the degree of utilization of the landfill volume. In this view, landfill closure timing is important, since settlement that occurs before the closure determines the possibility of disposing other waste within the volumes, that otherwise would be "lost" (if the landfill is closed, it is impossible to dispose other waste). The biodegradable fraction has a significant volume reduction potential since its tendency to lose water;
- Daily cover: it can significantly decrease the landfill volume available, since it can occupy a significant volume (5-10% of the total landfill volume). Considering also that the other soil covers can occupy about 20%-33% of the total landfill volume (Ziyang et al., 2015). The daily cover, if made with layers of soil or biostabilized material, usually has thickness of 10 cm and is placed every one to two meters of disposed waste. Through rough calculations it means that for the landfill model, where

the waste height is around 23 m, the daily top cover has a total thickness of 1.2-2.3 m. The corresponding volume for the daily cover layer must be subtracted from the usable landfill volume. Furthermore, this value can also increase, since settlement affects the waste layer thickness to greater extent than daily cover layers.

It is worth mentioning that in the case of slower disposal processes, the achievement of a minimum daily thickness of 1 m of waste needed for the placement of the daily cover can be difficult. For instance, with a waste inflow of 50 t/day (density 1 ton/m$^3$), a meter of waste thickness means the formation of an ideal solid with an area of 7 m by 7 m which is too small for compactor operations. Increasing that area to 10 m by 10 m, the incidence of the daily cover on a height of 50 cm of waste becomes equal to about 20%.

These considerations are important since they are a starting point for the optimization of the use of landfill volumes. However, these factors are really site specific and their influence on costs is difficult to evaluate.

### 2.4 Sensitivity analysis

The sensitivity analysis applied to the economic analysis is carried out using the "tornado" application of the program Crystal Ball©. The "tornado chart", a type of deterministic sensitivity analysis result output, was chosen to provide a graphical representation of the degree to which the results (in terms of unit cost in €/m$^3$) are sensitive to the specified Independent Variables (unit prices) with the other variables being held constant.

## 3. LANDFILL COST AND REVENUE ANALYSIS

The following sections detail the main analyses carried out to estimate the costs and revenues for the entire landfill life. The results are reported in Table 1 and are grouped into three levels of economic or functional categories: level one corresponds to the landfill phases plus an economic category (general expenses and net income) not correlated to a specific landfill phase but required at this level by specific national guidelines; level two corresponds to the main works/activities/services; level three corresponds to a more detailed description of the previous categories.

### 3.1 Design and authorization phase

The costs for design and authorization phases include the following technical activities: preliminary studies and surveys (topography, geology, geotechnical, etc.), waste characterization study, project management at different levels (preliminary, definitive and executive), plan for the operative and aftercare phases, security plan, site engineering, administrative and technical testing, and environmental impact assessment.

These aspects can be easily evaluated as a percentage of the construction cost (8% in this study). More sophisticated approaches such as the one suggested by Italian law DM 143/2013 use a parametric calculation based on three parameters: construction cost, performance complexity (buildings, structures, facilities, roads, etc.), and performance specificity (preliminary project, definitive project, executive project, etc.). However, these results are too high if compared to the ones from the selected statistic and they are not representative of applied costs.

Other costs for different authorization procedures such as investigation, notices, meeting, etc., can be estimated on the basis of worker hourly rates for the public sector.

### 3.2 Construction phase

The costs for the construction phase include the investment costs (area acquisition, construction cost, and machinery purchase) and the financial expenses due to investment and indirect costs.

Concerning construction cost, it is important to underline that some costs were included here even if not actually incurred during the construction period. This is necessary for an easier presentation of the results. This method was also followed in the majority of bill of quantities and financial plans analyzed. For example, in this case, we consider the temporary and final top covers, which are realized in the operational phase.

#### *3.2.1 Area acquisition*

Within a given country, as well as across countries, acquisition costs are difficult to specify in any formulaic manner. In some cases, the site may be acquired outright for a fee, in others, a royalty might be paid, or the site may be leased. It is difficult to generalize about the costs of acquisition and much depends upon the landowner in determining these costs.

In this work we adopted the methodology used for the compensation of a motorway called "Pedemontana Veneta" (2011) which reimburses voluntary transfer of non-building areas with a payment three times the Average Agricultural Value, assumed equal to 5 €/m$^2$. Therefore, area acquisition costs amounts 3 x 5 €/m$^2$ x 94,500 m$^2$ = € 1,417,500.

#### *3.2.2 Construction cost*

The construction cost includes all of the works planned by the project. It is important to underline that the costs are not considered in a chronological order, but they are grouped according to the type of work or area of interest.

Preliminary work include the construction and dismantling of the construction site and the excavation of general cleaning, while morphological shaping includes the cost for remodeling the borrow pit.

The costs of the bottom liner system and the top cover system arises due to the cost of supply and installation of the different liners required by regulations.

The cost related to the leachate system includes the slotted pipes for drainage, the lifting shafts for collection, the removal system, and the containment basin consisting of four storage tanks.

LFG system cost is due to the cost of wells and slotted pipes for the collection, connection pipes and regulation stations for the transport, and extraction station and flare for the disposal.

Monitoring includes the supply and installation of the piezometers for groundwater control, the weather station

**TABLE 1:** Landfill gate fee calculation. The economic categories have been grouped in three levels: level one corresponds to the landfill phases plus the general expenses and net income; level two corresponds to the main works/activities/services; level three corresponds to a more detailed description of previous categories. The range of unit costs (where available) were estimated on the analysis of a statistic of more than 15 landfill projects designed on the basis of current legislation (D.Lgs. 36/2003), on the suggestions of national landfills guidelines (CTD, 1997; DGR n. X/2461/2014), and on best practices.

| Items | | Estimated values for the reference model | | Unit Cost (€/m³) | |
|---|---|---|---|---|---|
| | | Tot for item (€) | % | Estimated values | Range from statistics (Min - Max) |
| **1** | **Design and Authorization Phase** | **870,303.61** | **1.26%** | **1.09** | **0.35-2.20** |
| 1.1 | Design and Authorization Cost | 870,303.61 | 1.26% | 1.09 | 0.35-2.20 |
| **2** | **Construction Phase** | **18,031,437.69** | **26.20%** | **22.54** | **9.01-34.19** |
| 2.1 | Area Acquisition | 1,417,500.00 | 2.06% | 1.77 | 0.87-4.07 |
| 2.2 | Construction Cost | 10,503,795.11 | 15.26% | 13.13 | 6.50-25.87 |
| 2.2.1 | Preliminary Works | 164,987.20 | 0.24% | 0.21 | n.a. |
| 2.2.2 | Morphological Shaping | 271,455.00 | 0.39% | 0.34 | n.a. |
| 2.2.3 | Bottom Liner System | 2,849,076.45 | 4.14% | 3.56 | n.a. |
| 2.2.4 | Top Covers System | 4,671,169.43 | 6.79% | 5.84 | n.a. |
| 2.2.5 | Leachate System | 409,841.30 | 0.60% | 0.51 | n.a. |
| 2.2.6 | Landfill Gas System | 458,982.86 | 0.67% | 0.57 | n.a. |
| 2.2.7 | Monitoring | 32,135.10 | 0.05% | 0.04 | n.a. |
| 2.2.8 | Landfill Hydraulic Settlement | 25,386.66 | 0.04% | 0.03 | n.a. |
| 2.2.9 | Underground Utilities | 170,621.27 | 0.25% | 0.21 | n.a. |
| 2.2.10 | Internal Road and Service Area | 274,463.58 | 0.40% | 0.34 | n.a. |
| 2.2.11 | Facilities | 179,000.00 | 0.26% | 0.22 | n.a. |
| 2.2.12 | Environmental Restoration Works | 137,746.59 | 0.20% | 0.17 | n.a. |
| 2.2.13 | Final Works | 80,870.77 | 0.12% | 0.10 | n.a. |
| 2.2.14 | Safety | 778,058.90 | 1.13% | 0.97 | n.a. |
| 2.3 | Machinery Purchase | 1,350,000.00 | 1.96% | 1.69 | 0.66-1.95 |
| 2.4 | Financial Expenses | 4,760,142.58 | 6.92% | 5.95 | 1.52-9.80 |
| 2.5 | Indirect costs | 0 | 0.00% | 0.00 | n.a. |
| **3** | **Operational Phase** | **29,325,233.76** | **42.60%** | **36.66** | **4.90-42.87** |
| 3.1 | Operating Costs | 10,039,565.91 | 14.59% | 12.55 | 2.99-19.97 |
| 3.1.1 | Staff | 4,598,350.00 | 6.68% | 5.75 | n.a. |
| 3.1.2 | Utilities and Materials | 400,000.00 | 0.58% | 0.50 | n.a. |
| 3.1.3 | Leachate Management | 1,917,973.05 | 2.79% | 2.40 | n.a. |
| 3.1.4 | Landfill Gas Management | 458,982.86 | 0.67% | 0.57 | n.a. |
| 3.1.5 | Daily Top Cover | 464,760.00 | 0.68% | 0.58 | n.a. |
| 3.1.6 | Monitoring | 344,500.00 | 0.50% | 0.43 | n.a. |
| 3.1.7 | Maintenance | 750,000.00 | 1.09% | 0.94 | n.a. |
| 3.1.8 | Other Services (technical costs, etc.) | 1,105,000.00 | 1.61% | 1.38 | n.a. |
| 3.2 | Pollution Liability Protection in Operation | 180,000.00 | 0.26% | 0.23 | 0.12-0.59 |
| 3.3 | Financial Guarantees in Operation | 118,787.86 | 0.17% | 0.15 | 0.02-4.95 |
| 3.4 | Contribution for Env. Annoyance and Landfill Tax | 18,986,880.00 | 27.58% | 23.73 | 2.14-32.89 |
| 3.4.1 | Contribution for Environmental Annoyance | 5,807,120.00 | 8.44% | 7.26 | n.a. |
| 3.4.2 | Landfill Tax | 13,179,760.00 | 19.15% | 16.47 | n.a. |
| 3.5 | Operating revenue | 0.00 | 0.00% | 0.00 | n.a. |
| **4** | **Aftercare Phase** | **7,149,570.29** | **10.39%** | **8.94** | **2.74-12.20** |
| 4.1 | Operating Costs | 6,557,113.38 | 9.53% | 8.20 | 2.06-11.74 |
| 4.1.1 | Staff | 2,035,570.00 | 2.96% | 2.54 | n.a. |
| 4.1.2 | Utilities and Materials | 244,000.00 | 0.35% | 0.31 | n.a. |
| 4.1.3 | Leachate Management | 3,068,401.95 | 4.46% | 3.84 | n.a. |
| 4.1.4 | Landfill Gas Management | 229,491.43 | 0.33% | 0.29 | n.a. |
| 4.1.5 | Monitoring | 422,250.00 | 0.61% | 0.53 | n.a. |
| 4.1.6 | Maintenance | 512,400.00 | 0.74% | 0.64 | n.a. |
| 4.1.7 | Other Services (technical costs) | 45,000.00 | 0.07% | 0.06 | n.a. |
| 4.2 | Pollution Liability Protection in Aftercare | 540,000.00 | 0.78% | 0.68 | 0.05-1.76 |
| 4.3 | Financial Guarantees in Aftercare | 52,456.91 | 0.08% | 0.07 | 0.02-2.05 |
| 4.4 | Aftercare revenue | 0.00 | 0.00% | 0.00 | n.a. |
| **5** | **General Expenses and Net Income** | **13,456,500.52** | **19.55%** | **16.82** | **8.06-19.03** |
| 5.1 | General Expenses | 7,198,950.90 | 10.46% | 9.00 | 2.77-9.00 |
| 5.2 | Net Income | 6,257,549.62 | 9.09% | 7.82 | 3.75-9.17 |
| **TOT** | **TOTAL COST - NO VAT (22%)** | **68,833,045.87** | **100.00%** | **86.04** | **38.28-86.95** |

*n.a.: not applicable or not available*

for meteorological data measurements and plates for settlements and diffused LFG monitoring.

Landfill hydraulic settlement refers to works necessary for the management of rainwater falling on the landfill surface: the perimeter channel all around the landfill and the rainwater drainage trench.

The cost of underground utilities grouped all the following systems: service area rainwater drainage system, first rainwater treatment system, water storage tank, wastewater system, service area drainage trench, electric network.

Internal road and service area cost is simply the cost due to the realization of the earth made road all around landfill perimeter and the asphalt for the service areas.

The costs for the office building, temporary storage basin, platform scale, and tire-washing system are all considered in facilities.

Environmental restoration works is the cost for landfill cover grass.

Final works include the cost for the fence, to prevent the uncontrolled access to people and animals; the gate, to permit the access for cars and trucks; and the area surrounding hedge, to hide the construction site.

Finally, safety cost contributes to the estimation of the construction cost and it represents approximately 8% of the cost due to all the other categories of this paragraph.

#### *3.2.3 Machinery purchase*

The investments for acquisition of new machinery includes: 1 tracked digger (€ 210,000.00); 2 tracked operating machines for waste placement (2 x € 250,000.00); 1 agricultural tractor with brush-cutter arm (€ 160,000.00); n°1 compactor (€ 480,000.00). This cost is extremely variable and has to be estimated on a case by case basis.

#### *3.2.4 Financial expenses due to investments*

The financial expenses associated with the investments (area acquisition, construction costs, machinery purchase) are calculated assuming full debt financing at an yearly interest of 6% (range 5-7%) (Hogg, 2001; Florio, 2003) for a period of 10 years, equal to the duration of the operational phase. Constant rate amortization (Followill, 1998) was used for calculations, considering compounded interest.

#### *3.2.5 Indirect costs*

Indirect costs are difficult to quantify but they can potentially have a decisive role on the economic feasibility of the landfill.

An important indirect cost is the devaluation of the areas closed to the landfill. Devaluation of surrounding areas can reduce their original value (5.00 €/m$^2$) by more than 50% compared to before landfill construction. In addition, the effects of devaluation can extend for hundreds of meters (even a kilometre) from the landfill site, even in the case of well-designed and managed landfills.

In this case, considering the landfill model, some rough calculations can be made. Assuming an average value of the areas of 50,000.00 €/ha (before landfill construction) and considering 1 km radius of influence area, for a total of about 438 ha as the average depreciation (hypothesis that devaluation has linear behaviour, from 50% of pre-landfill value in the nearness of the landfill to 0% at 1 km of distance) can be estimated as follows:

438 ha x 50,000 €/ha x 25% = 5,475,000 €

which correspond to 6.8 €/m$^3$ in relation to the assumed landfill waste volume (800,000 m$^3$). It is important to note how much depreciation can increase in the case of valuable surrounding areas and therefore its incidence on landfill cost.

In order to minimize local devaluation, some general criteria should be followed when choosing the landfill site. These criteria may include a distance from housing greater than 1 km, preference of uncultivated or low-value areas, choice of a site with low visibility from urban centres, and from important landscape attractions and with easy and direct access from primary roads. On the contrary, the devaluation is near to zero in the case of environmentally degraded areas or previous landfill sites. However, in general, these costs are not mentioned in financial plans and they can be included in the cost for the area acquisition or into the contribution for landfill annoyance, as was done in the present study.

### 3.3 Operational phase

The economic analysis for the operational phase includes several different categories: operating costs, pollution liability protection, financial guarantees, contribution for landfill annoyance, landfill tax and landfill operating revenues.

In particular, pollution liability protection and financial guarantees are necessary to insure the site management to the benefit of the public administration in case of pollution, or rather in the case of bankruptcy/transfer of ownership of the company. Financial guarantees are also required for the aftercare phase.

On the contrary, the contribution for landfill annoyance and landfill tax are related only to the operational phase, in fact they have to be paid during the period of waste delivery.

#### *3.3.1* Operating costs

The operating costs are the costs related to staff, utilities and materials, leachate and LFG management, daily top cover, monitoring, maintenance and other services.

*Staff*

The staff cost due to each job position can be obtained from the national tables FISE (2016) of recognition time for every single task and level, constituting a substantial part of the national contracts for environmental professionals. Considering 6 to 8 daily working hours and 250 working days/year, the staff cost according to task and responsibility is the following: n°1 operation director (75 €/hour x 2,000 hours/year); n°1 technician responsible of the plant (40.25 €/hour x 2,000 hours/year); n° 2 technical-administrative employees (2 x 28.65 €/hour x 1,500 hours/year); 3 workmen (3 x 23.53 €/hour x 1,500 hours/year); n° 1 supervisor (25.00 €/hour x 1,500 hours/year).

*Utilities and materials*

Utilities and materials include the cost of electricity, telecommunications, water, fuels, lubricants, reagents, and other consumables. In particular, the costs of electricity used by leachate pumps and LFG stations are not included because they are counted respectively under leachate management and LFG management. The yearly l cost has been set equal to 40,000.00 €/year.

*Leachate management*

Leachate management includes the costs related to utilities and to maintenance of the leachate system and the leachate treatment and/or disposal.

The leachate system cost is due to the electricity consumed by the pumps, while the cost for maintenance is calculated as 5% of the cost of leachate system construction.

Financial plans calculate this cost defining a unit price in reference to the volume of leachate produced, including transport and treatment; the cost can vary according to the leachate quality from 17 €/m$^3$ to 45 €/m$^3$, respectively for contaminated rainwater and for highly contaminated leachate (chemical oxygen demand (COD) > 10,000 mg/L). In this study a unit price of 40 €/m$^3$ has been considered for leachate disposal and, therefore, during the operational phase the total cost is: 40.00 €/m$^3$ x 42,540.31 m$^3$ = € 1,701,612.

*Landfill gas management*

Unlike leachate management, the cost for LFG management is just due to LFG system utilities (mainly electricity consumed by the regulation stations, the extraction station and the flare) and maintenance. In fact, its disposal is not necessary because LFG is treated on site by combustion into a high temperature flare. These two contributions are both obtained considering 5% of the cost for LFG system construction.

*Daily top cover*

Differently from temporary and final top covers, daily top cover is strictly linked with the operational phase: it is placed every day on top of the waste disposed during that day. For this reason, it has been considered in this phase and not in the construction phase, as was the case for the temporary and the final top covers.

*Monitoring*

The monitoring activity follows the indications of the supervision and control procedure prescribed by D.Lgs. 36/03. These activites have to be carried out by qualified and independent staff on: groundwater, leachate, drainage surface water, LFG, air, meteorological data, and landfill topography.

Groundwater controls for both volume and composition are required. Groundwater level is measured monthly in five piezometers with an average unit price of 25.00 €/measurement. For groundwater chemical analyses, the parameters are analyzed with the following frequency: quarterly for 11 parameters (pH, temperature, electric conductivity, Kübel oxidability, chlorides, sulphates, Fe, Mn, ammonia nitrogen, nitrate, nitrite) and yearly for 26 parameters (five-day biochemical oxygen demand (BOD5), total organic carbon (TOC), Ca, Na, K, fluorides, polycyclic aromatic hydrocarbons (PAH), heavy metals, cyanides, organic halogen compounds, phenols, pesticides and organic solvents). An average unit price of 20.00 €/parameter can be assumed.

Additionally, for leachate controls both volume and composition are required. Leachate level (indirect measure of the leachate volume) is measured monthly at the bottom of four lifting shafts with an average unit price of 25.00 €/measurement. For leachate chemical analyses, the samples are taken from the leachate storage tanks in order to obtain an average characterization of leachate extracted by the landfill. Parameters, frequency, and average cost are the same assumed for groundwater, except for BOD5, which is monitored quarterly. Moreover, the analysis of COD is also performed with quarterly frequency.

Chemical analyses on drainage surface water are performed on samples from four representative points, considering the same parameters and frequency assumed for leachate chemical analysis. The unit price considered is the same.

LFG quality analyses on $CH_4$, $CO_2$, $O_2$, $H_2$, $H_2S$, total particulate matter, $NH_3$, mercaptans, and volatile compounds are performed monthly at the extraction station (150.00 €/measurement). The analysis for the assessment of biogas uncontrolled emission from the landfill surface are performed three times a year (1,500.00 €/campaign survey).

Air quality monitoring is made monthly by portable equipment in four points with an average unit price of 150.00 €/measurement; the analyzed parameters should be the same considered for LFG monitoring.

Meteorological data monitoring is conducted automatically by the weather station, where the following meteorological sensors are installed: rain gauge (for precipitation), air thermometer (for temperature), anemometer (for wind direction and velocity), evaporimeter (for the evaporation), air hygrometer (for the humidity). The minimum frequency of monitoring is daily for all the parameters for a total cost of 50.00 €/year.

Landfill morphology monitoring is performed through topographical surveys conducted twice a year for a cost of: 350.00 €/ha x 5 ha= 1,750.00 €/each.

*Maintenance*

The maintenance cost includes the maintenance of machinery, facilities (such as the platform scale and tire washing system), underground utilities, and stations and systems for monitoring. Leachate and LFG system maintenance is not considered in this category because they are included in the leachate management and LFG management costs. The yearly cost for maintenance costs amounts to 75,000.00 €/year.

*Other services*

Other costs necessary during the operational phase are: technical costs for supervision and control procedure (1,500.00 €/year); disinfection and disinfestation (4,000.00 €/year); deratting and bird control (5,000.00 €/year); commercial costs (50,000.00 €/year); administrative costs (50,000.00 €/year).

*3.3.2 Pollution liability protection in operation*

The insurance in case of a landfill project is the pollution liability protection and it is defined as a function of volume of the landfill, typology of waste, and location (site vulnerability).

In case of a landfill for non-hazardous waste (including also MSW), not located in a zone of aquifer recharge, the maximum coverage is of € 1,500,000 for each 200,000 $m^3$ of landfill volume (DGRV n. 2721/14).

On the basis of a volume of 800,000 $m^3$, the calculated maximum coverage is of € 1,500,000 for 4 sectors of 200,000 $m^3$ = 6,000,000 €/year.

The total pollution liability protection cost in operation is calculated for the 10 years of landfill life assuming a percentage of 0.3% of the maximum coverage and the costs amounts to: 6,000,000 €/year x (0.3/100) = 18,000 €/year x 10 years = € 180,000.

This percentage is difficult to establish because it depends on financial solidity of the company, but also on the capital market and waste quality. It can vary between 0.1% and 0.33% of the maximum coverage.

It is important to underline that the financial guarantees can be reduced if the company has environmental certifications: by 50% in case of and Eco-Management and Audit Scheme (EMAS) registration, by 40% in case of certifications in accordance with UNI EN ISO 14001 and still by 50% if the enterprises complies with both of the environmental management systems (DGRV n. 2721/14).

*3.3.3 Financial guarantees in operation*

To obtain the authorization for a landfill, some financial guarantees must be provided in accordance with art. 14 of D.Lgs. 36/03.

In particular, a guarantee is required for the activation and the operational management of the landfill, including closure procedures, withheld for at least 2 years from the closure communication date.

These guarantees are calculated considering a percentage of 0.8% (range 0.7-2%) of the total planned cost for:

- Activation and operation costs;
- Costs for closure procedures and final recomposition of the landfill, such as the realization of the top covers, the construction of the 'embric' channels to drain the fallen rainwater on the closed landfill, and the landfill cover grassing.

In the case of landfills for which the authorization is approved for sectors, a guarantee can be given for them.

*3.3.4 Contribution for landfill environmental annoyance and landfill tax*

*Contribution for landfill environmental annoyance*

The contribution for environmental annoyances is a function of the non-hazardous waste typology (Special Waste or MSW) admitted in the landfill, according to DGRV n. 1104/2013.

In particular, remembering that non-hazardous special waste represents 70% of the total waste mass of the landfill model and the remaining 30% is MSW, the contributions for environmental annoyance are as follows:

- (70/100) x 880,000 t = 616,000 t x 5.00 €/t = € 3,080,000 for non-hazardous special waste;
- (30/100) x 880,000 t = 264,000 t x 10.33 €/t = € 2,727,120 for MSW.

*Landfill tax*

The landfill tax (called ecotax) is a form of tax required by art.3 of Italian Law n. 549/1995 in order to promote a lower waste production and the recovery of raw materials and energy from them. This tax is paid to the regional governments and is used to create a fund for inspection programs and long-term mitigation of environmental impacts related to disposal. It is applied on top of the other costs of the landfill causing a rise in the landfill gate fee. In this sense, it represents an inhibition of the disposal in comparison to preferable alternatives (Cossu and Masi, 2013). The amount of the tax is a function of the waste typology disposal in the landfill and established on a regional basis. In particular, according to LR n. 3/2000, remembering that non-hazardous special waste represents 70% of the total waste mass of the landfill model, and the remaining 30% is MSW, the contributions for landfill tax are the following:

- (70/100) x 880,000 t = 616,000 t x 10.33 €/t = € 6,363,280 for non-hazardous special waste;
- (30/100) x 880,000 t = 264,000 t x 25.82 €/t = € 6,816,480 for MSW.

Biostabilized material used for daily and temporary/final top covers, as the other materials used for landfill construction and management, are not subject to pay such contributions.

If municipalities reach a percentages of separate collection greater than 50 - 65% with respect the total collected waste, the landfill tax can be reduced by 35% - 70% (DGRV n. 288/2014). This discount is not considered in the present case study.

*3.3.5 Operating revenue*

The most common revenue from a landfill is generated by the sale of energy from LFG. In this case the price depends on the regime governing energy sales. The operators may contract out the management of LFG for energy recovery, and where energy prices are favourable, they may take a royalty fee in lieu of the contract.

In the case of plants which collect less than 100 $m^3$/h of LFG (DGRV n. 995/2000), the energy recovery is not considered economically favourable because the product energy has to be treated which is an added cost. This is also the case for the present study. A separate category can be created for the revenue or it can be included in the mentioned economic categories.

## 3.4 Aftercare phase

The aftercare phase includes operating costs, pollution liability protection, financial guarantees and aftercare revenue. In particular, as for the operational phase, these costs represent non-negligible costs allocated to the total quantities deposited in the plant, even in case of plant stoppage. It is important to underline that these costs are collected when waste disposal starts, but they are saved and used after 30 years from the end of the operation activity. This fact introduces a serious management risk, because of the tendency to use these reserves to cover operational cash outflows, and because they can ultimately prove insufficient.

*3.4.1 Operating costs*

The operating costs during the aftercare phase are the costs related to staff, utilities and materials, leachate and LFG management, monitoring, maintenance, and other services.

During the 30 years of aftercare, costs are reduced to take into consideration the reduction of the required landfill activities.

*Staff*

The staff cost during the aftercare phase is obtained from the national tables FISE (2016) which were used for the operational phase. It includes: n°1 technician responsible of the plant (40.25 €/hour x 1,000 hours/year); n°1 workman (23.53 €/hour x 2,000 hours/year); n°1 responsible for supervision and control plan (25.00 €/hour x 210 hours/year).

These costs are reduced by about 30% every 10 years.

*Utilities and materials*

Utilities and materials include more or less the same costs of the operational phase, but the yearly consumptions are lower. As for the operational phase, these costs do not include the cost for electricity used by leachate pumps or biogas stations, which are accounted for in the leachate and LFG management categories.

Unlike the operational phase, the cost is not the same every year. This cost amounts to € 10,000.00 for the first year, and then decreases about 20% every 10 years.

*Leachate management*

Leachate management during aftercare includes the same costs for the operation: consumptions and maintenance of the leachate system and leachate disposal. Differently from the operational phase, the amount of leachate does not vary each year, but it is assumed constant, so the leachate disposal cost amounts to: 40.00 €/$m^3$ x 2,027.54 $m^3$/year = 81,101.60 €/year, for a total of € 2,433,048 in 30 years.

*Landfill gas management*

As for the operational phase, the cost for LFG management is just due to LFG system consumptions and maintenance. These two contributions are both obtained considering the 5% of the cost for LFG system construction for the first 5 years of the aftercare.

*Monitoring*

As for the operational phase, the monitoring follows the directions of the supervision and control procedures according to D. Lgs. 36/03, point 5. The monitored matrices (the parameters and the points) are the same as the operational phase, but the frequency changes. The unit prices for the monitoring activity are assumed to not vary with time.

The groundwater level is measured twice a year whereas the quality composition is monitored half-yearly for the 11 fundamental parameters and yearly for the other 26. The frequencies are the same respectively for the leachate level, fundamental parameters (13 in case of leachate), and the other parameters (25 in case of leachate). Chemical analyses on the drainage rain water are performed considering the same parameters and frequency for the leachate chemical analyses. For LFG, the quality analyses are conducted every six months for the extraction station and yearly (for the first 5 years) for the top covers to analyze the uncontrolled emissions. Air quality monitoring is conducted twice a year using portable unit at four points. The frequency of meteorological data monitoring can be reduced and the monitoring of wind direction and velocity is no longer required. Landfill morphology monitoring is performed through topographical surveys every six months for the first three years after that this monitoring is conducted yearly.

*Maintenance*

The maintenance cost during aftercare includes maintenance of underground utilities and of monitoring stations and systems, and maintenance of vegetative landfill covers and the restoration of depressions and caves. Consequently, the cost per year is lower than during the operational phase.

As for the operational phase, the leachate system and LFG system maintenance are not considered in this category, because they are considered respectively in the leachate management and LFG management costs.

These aspects amount to € 21,000.00 for the first year, and then decreases about 20% every 10 years over the aftercare period.

*Other services*

Different from the operational phase, the only other services necessary in the aftercare phase are the technical costs which amount to 1,500.00 €/year.

*3.4.2 Pollution liability protection in aftercare*

The total pollution liability protection cost in aftercare is calculated for 30 years assuming a percentage of 0.3% of the maximum coverage estimated according to DGRV n. 2721/14 as for the operational phase.

This cost amounts to: 6,000,000 €/year x (0.3/100) = 18,000 €/year x 30 years = € 540,000.

*3.4.3 Financial guarantees in aftercare*

Financial guarantees in accordance with D. Lgs. 36/2003 include the costs for aftercare management of the landfill, and they are withholding for at least 30 years from the closure communication date.

These guarantees are calculated, as for the operational phase, considering a percentage of 0.8% of the planned aftercare cost without a value-added tax (VAT).

It is important to underline that for aftercare the guarantees are withheld for at least 30 years, which is a very long period in the financial market which could give rise to solvency issues. A valid solution to these problems could be a five-year automatic renewal period.

*3.4.4 Aftercare revenue*

Energy production from LFG can represent an important revenue opportunity, especially if the activity was profitable during the operational phase. However, decreasing LFG production makes this option usually not convenient.

Other revenue possibilities can derive from innovative technical interventions on the surface of the final landfill top cover and their practical feasibility and economic convenience must be assessed on case by case basis. The installation of photovoltaic parks represent an already important and widely adopted solution. Furthermore, new opportunities are always considered and under development. This is the case of energy crops, which represent a promising opportunity for the near future. In the present study these aspects are not considered in the operational phase.

### 3.5 General expenses and net income

The general expenses and the net income have to be considered in order to define the landfill gate fee. The first corresponds to the 13% (range 13%-17%) of the all costs calculated without VAT, the second to 10% of all costs plus general expenses (art. 32 of the DPR n. 207/2010).

## 4. DISCUSSION

Results obtained from the cost analysis of the whole life cycle of the landfill model are shown in Table 1. The unit cost is obtained by dividing the total cost by the waste volume: € 68,833,045 / 800,000 $m^3$ = 86.04 €/$m^3$. It represents the landfill gate fee, expressed per volume unit (€/$m^3$) and is the sum of the costs related to design and authorization phase, construction phase, operational phase, aftercare phase, general expenses, and net income. None of these costs in the VAT.

The same calculation is repeated for all the three categories reported in Table 1, to obtain a unit cost for each category.

Economic categories level one and two have been compared with ranges obtained from the mentioned statistics, while this was not possible for level three because the investigated document used a different economic structure. It is worth mentioning that all unit costs fall into ranges, therefore it is reasonable to confirm that the goal to define a representative landfill case was achieved.

Regarding the cost distribution in the different landfill phases, the operational phase is where most of the costs are incurred (42.60% of the total landfill cost); it is followed by the construction phase (26.20%), the aftercare phase (10.39%), and the design and authorization phase (1.26%). Moreover, the last two phases together are referred to as lower costs relative to the general expenses and net income category (19.55%).

Concerning financial expenses, the obtained wide range could be attributed to different assumptions. In some financial plans they are calculated based only on the construction cost (minimum value), while others consider the technical expenses as investment costs (maximum value). Among investment costs there are the construction costs, the area acquisition, and machinery purchases. Concerning area acquisition, this cost is difficult to generalize because it depends on many variables (position, presence of building, etc.). Concerning the machinery purchase, the estimated unit costs are quite high because all machinery is estimated to be bought as new items. Buying used machinery or renting can reduce these costs. Within the construction costs, the sub-category with the highest cost (15.26%) also has a wide range which is a result of the different technical solutions and services adopted and strictly correlates to its geographical position (presence or not of an exhausted borrow pit, rather than mountainous or flat terrain) and to the quantity and quality of the disposed waste.

On the contrary, some of the sub-categories of the construction costs are set by the legislation which establishes many of minimum requirements for landfill construction in order to reduce environmental impacts. In this sense the model landfill was designed in accordance with the laws and the best practices and for this reason the corresponding percentages and unit costs align with the regulatory expectations.

For instance, top covers have the highest impact on construction costs followed by the bottom liner. Since the minimum thickness is fixed by the law, the cost may increase considering large landfill surfaces. In this view, an important cost minimization is obtained considering that the average height of the landfill should be greater than 1/10 of its shorter base. This is usually obtained by designing landfills with a total landfill height of about 20-30 meters, like those realized in exhausted borrow pits, which is also the case of the present case study.

Operating costs in the operation and in aftercare phases both consist of staff time. In particular, the staff costs in the operational phase is higher than the costs in the aftercare phase, even if the aftercare period is three times the operation period (30 years versus 10 years). This trend is due to the fact that the operational phase requires more employers than the aftercare phase.

In the operational phase, which is the most expensive one, the definition of the operation time (statistically between 6 and 13 years) is fundamental because cost amounts are very sensitive to changes in the phase duration. Landfilling must therefore be planned carefully, avoiding operation stops, especially long ones. A stop in operations for one year may be onerous and bring important budget deficit challenges.

A further evaluation was carried out in order to identify the most important detailed costs. The analysis compares all of the single voices, not directly highlighted in Table 1, used in the bill of quantities and in the financial panel. These results are reported in Figure 4 where two classes of costs are distinguished. The latter is related to environmental compensations and taxes that does not represent "real" operations where the technology can be applied and they can differ significantly from region to region and from regulation to regulation. In the present analysis these aspects constitutes above more than 1/4 of the total costs. The latter is related to construction and management operations/

works and the results show that the most "10" relevant costs are correlated to leachate disposal (both in operational and aftercare phases), to staff (also in both phases), and to raw materials for liner construction.

Figure 5 represents the tornado chart for the sensitivity analysis of the most relevant costs. The x-axis represents the values of the total unit cost (€/m$^3$) for different values of the independent unit prices. Each bar represents the range of values produced when each independent variable (unit price) is set to the lower bound, central value, and upper bound (with the other variables being held constant). In the present work, the central value is represented by the mean value of the unit price used directly in the model, while the lower and upper bounds were defined equal to -20% and +20% of the central value.

The dark grey indicates that the value is produced by the lower bound, while the light grey bar indicates that the value is produced by the upper bound (high). The vertical line between the two bars represents the mean value of 86.04 €/m$^3$. This analysis shows that a variation of ± 20% of the most significant unit prices determines a variation lower than 2 € of the total unit cost.

With a reduction of 50% of the landfill tax for MSW (the most sensitive unit price as reported in Figure 5) that can occur if a separate collection greater than 50-60% is reached, the total unit cost is reduced to 73.83 Euro/m$^3$, saving 12.21 Euro per cubic meter of MSW waste landfilled.

In conclusion, the obtained unit costs have been compared to the values of the landfill gate fee for other Italian Regions expressed as a function of the waste weight (Andretta et al., 2010). The average value of the unit cost of Figure 6 is 84.96 €/t relative to 78.22 €/t (density=1.1 t/m$^3$) obtained from the model landfill. This fact points out that the correct average behaviour assumption for the model landfill was reasonable and was also supported by the unit cost point of view.

## 5. CONCLUSIONS AND FURTHER DEVELOPMENTS

Landfill costs were defined for a model landfill representative of Northern Italy that refers to the whole landfill life, including the phases of design and authorization, construction, operation, and aftercare. Results were expressed per unit of landfill volume available for waste and compared with economic values obtained from a statistical analysis of several landfill cases and from literature values of different Italian regions. The obtained landfill fee gate (86.04 €/m$^3$) is in line with the analysed references and depends mainly on the costs of the landfill tax, the contribution from landfill environmental annoyances, the leachate disposal fees, the staff, and raw materials for liner construction. However, other factors can significantly influence the landfill gate fee: location (presence or not of an exhausted borrow pit, rather than mountainous or flat terrain; presence of building, roads, network services, etc.), climatic conditions (mainly for leachate production), waste quality for LFG and leachate production, landfill geometry (in particular the

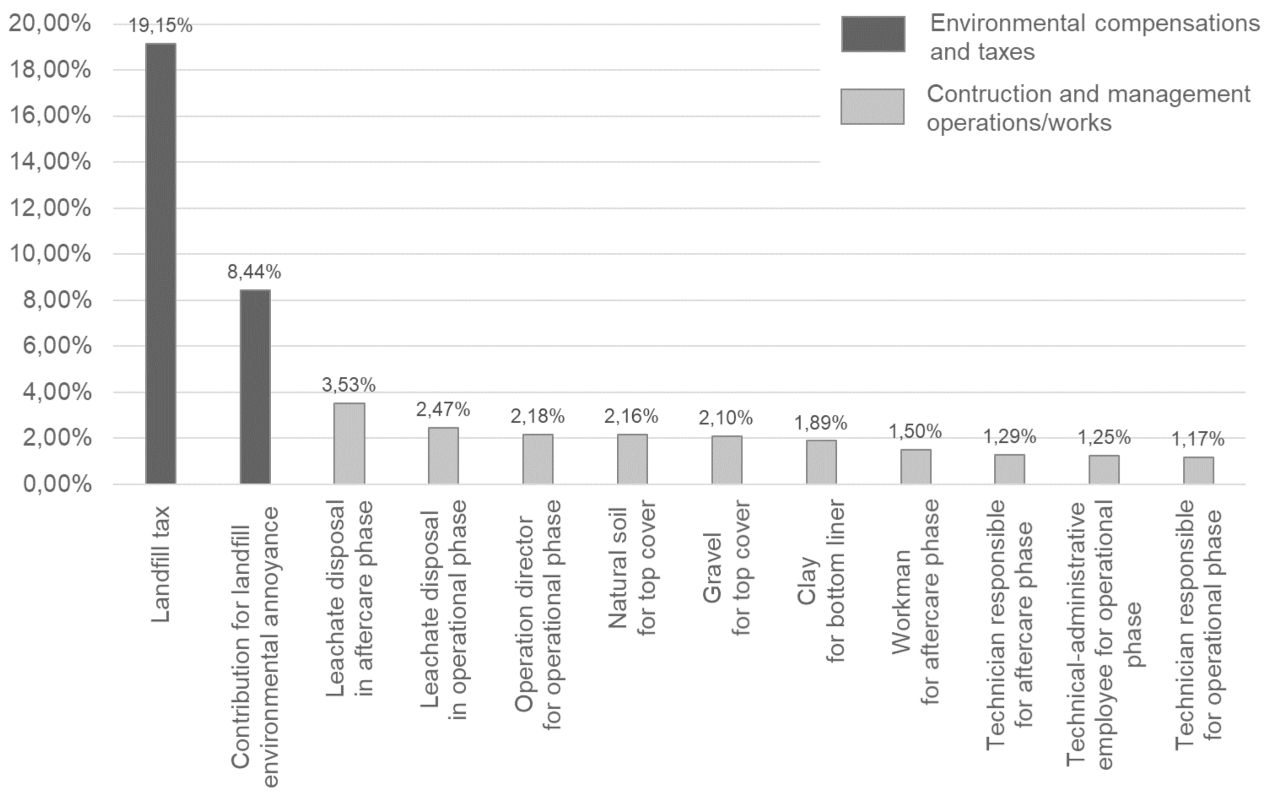


**FIGURE 4:** Top detailed economic voices for the definition of landfill gate fee or the reference landfill model. Two classes of costs were considered: environmental compensation and taxes and construction and management operation/works.

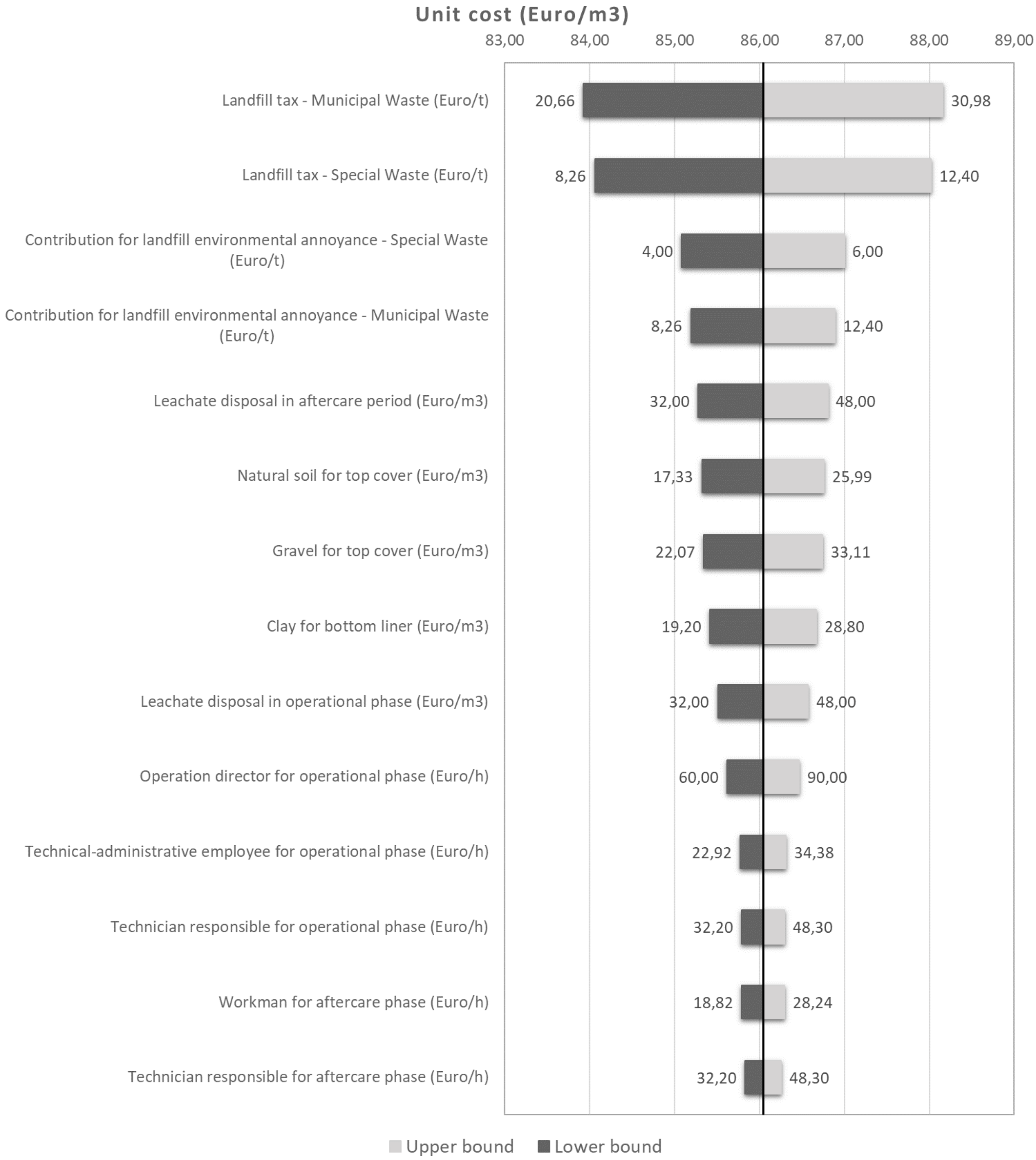


**FIGURE 5:** Tornado chart for the sensitivity analysis of the most significant unit prices.

ratio surface/height of waste), operative procedures (daily cover, utilization factor of the volumes, etc.), financial aspects (area acquisition, interest rate for the investments, financial guarantees, etc.) and legal aspects (pollution liability, contribution for environmental annoyance and landfill tax).

This study can be used as a starting point for a comparative economic analysis. For instance, the assumed model landfill can represent a reference scenario useful for the comparison of different landfill configurations (size, operational time, biogas recovery, etc.) and for the assessment of new innovative technology applications, such as energy crop application, in-situ aeration, and flushing.

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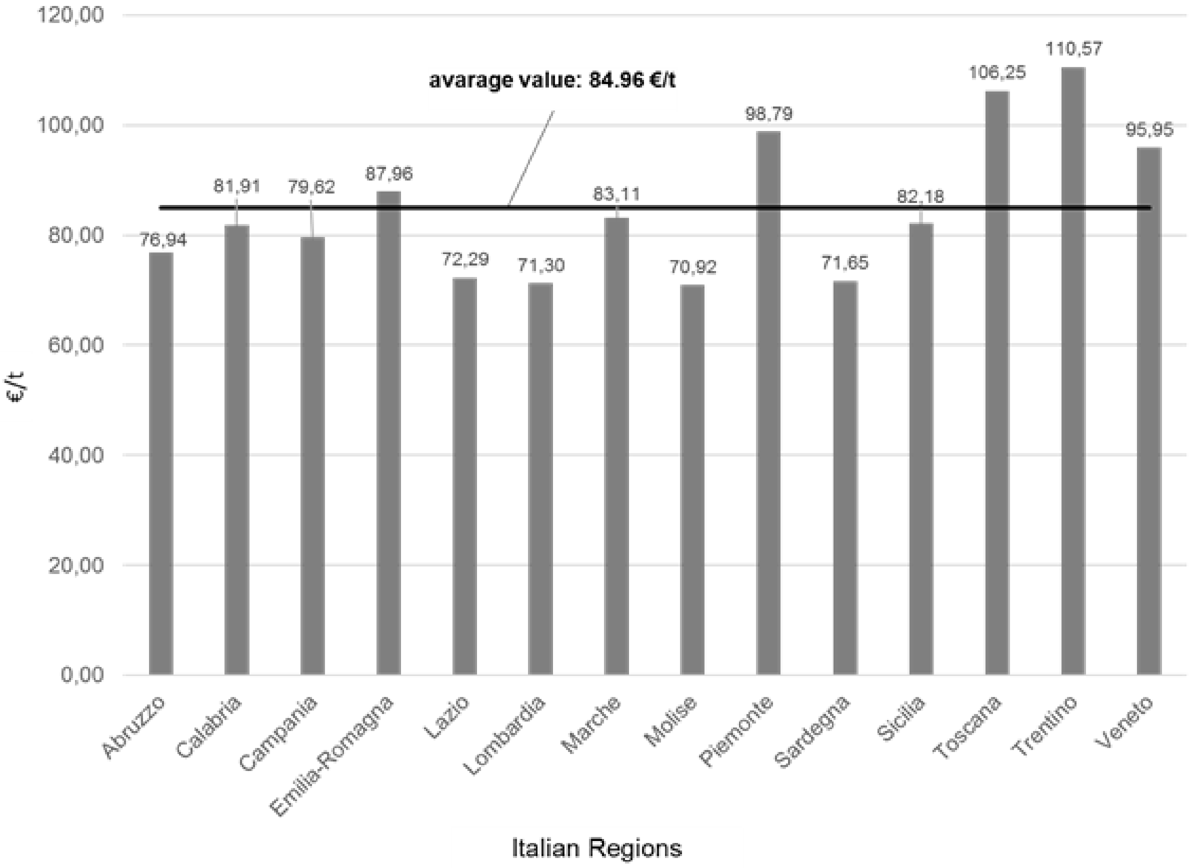


**FIGURE 6:** Average values for landfill gate fee in other Italian Regions.

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# DEVELOPMENT AND APPLICATION OF A PROTOCOL TO ASSESS HEALTHCARE WASTE MANAGEMENT

Leonardo de Lima Moura [1,*], Claudio Fernando Mahler [1] and Heitor Mansur Caulliraux [2]

[1] *Department of Civil Engineering, COPPE, Federal University of Rio de Janeiro, Av. Horácio Macedo, 2030 - 101, Cidade Universitária, RJ, 21941-450, Rio de Janeiro, Brazil*
[2] *Production Engineering Program, COPPE, Federal University of Rio de Janeiro, Av. Horácio Macedo, 2030 - 101, Cidade Universitária, RJ, 21941-450, Rio de Janeiro, Brazil*

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**ABSTRACT**
Healthcare waste management (HCWM) is a current problem in many developing countries. One of the primary steps to implement a proper HCWM process is to know the amount of healthcare waste (HCW) generated, its composition and the logistics flow. However, due to a lack of systematic protocols to facilitate the assessment of HCWM, this information cannot be obtained easily. Therefore, this study, based on a comprehensive literature review, aimed to develop a new tool for assessment of HCWM. This protocol also involves qualitative (questionnaires and observation) and quantitative (quantification and gravimetric characterization) data. Application of the protocol in a maternity hospital showed that it allows complete assessment of HCWM, especially the identification of the most significant sectors and the contribution of each one to the total HCW. We concluded that the protocol can be important for healthcare professionals and researchers to obtain information to improve HCWM.

## 1. INTRODUCTION

Poor management of healthcare waste (HCW) is an increasing problem in many developing countries as populations grow and demand for health services increases to enable improved quality of life and longer life expectancy. These countries have gaps in different stages of HCW management, especially with regard to segregation and final disposal.

Healthcare waste management (HCWM) practices vary from country to country and regionally within countries, since they depend on several factors, such as socioeconomic conditions, human and financial resources available and existing legislation and regulations. Therefore, an important step for implementation of better HCWM is knowledge of the amount of waste generated and its composition.

In developing countries, there is a lack of data about HCW generation and composition. This has encouraged research into HCW assessment in hospitals in several countries, such as Brazil (Andre et al., 2016; Souza et al., 2015; Aduan et al., 2014; Moreira and Gunther, 2012), Ethiopia (Tadesse et al., 2014; Tesfahun et al., 2014; Idowu et al., 2013; Debere et al., 2013), India (Sharma and Gupta, 2017; Hiremath et al., 2016; Kumar et al., 2014), Iran (Ghafuri and Nabidazeh, 2017; Sartaj and Aragbol, 2015; Hadipour et al., 2014; Malekahmadi and Yunesiam, 2014; Oroei et al., 2014; Koolivand et al, 2012), Nigeria (Oyekale and Oyekale, 2017; Anozie et al., 2017; Awodele et al., 2016; Oli et al., 2016; Longe, 2012) and Pakistan (Ali et al., 2017; Kumar et al., 2015).

In general, these studies used a wide variety of data collection tools, which hinders reproducibility. Therefore, this article, based on a comprehensive literature review, proposes a protocol to support the assessment of HWCM in health units.

## 2. THE PROPOSED PROTOCOL FOR HCWM ASSESSMENT

The proposed HCWM assessment protocol is divided into three steps, described next.

### 2.1 Step 1: Observing the hospital routine

The objective is to monitor the flow of patients and HCW. In relation to places to visit, the most important are waste storage rooms and external waste shelters. In these areas it will be possible to see the hospital's infrastructure for HCWM in addition to how waste is segregated, packaged and transported. The researcher is expected to establish initial contacts with the cleaners at the hospital and to develop an understanding of how their activities are related with HCWM.

* Corresponding author:
Leonardo de Lima Moura
email: leonardodelmoura@gmail.com


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Among the cleaning staff, the protocol involves monitoring their activities through listening, observation and annotation of their activities. Furthermore, it is important to obtain other information about HCWM from different professionals, such as doctors, nurses and pharmacists, because their different responsibilities and attitudes directly influence the HCWM, especially with regard to the adequate segregation of waste.

The data should be documented by taking daily notes for later transcription. It is also important to use photo documentation, of course with the permission of the hospital management.

Some documents that are important to analyse are the hospital's waste management plan and documents regarding external collection, treatment and final disposal, as well as the legal and regulatory rules on waste management.

At the end of this stage, the researcher should be able to describe the HCW logistics flow and the flow of patients at the hospital, identifying how much and what types of wastes are generated by each sector.

### 2.2 Step 2: HCW quantification and characterization

The objective of this stage is to obtain information regarding the amount and types of HCW, through quantification and gravimetric characterization. This stage will be carried out concomitantly with characterization and the choice of the setting will depend on the process of each hospital, but it is usually simpler to do this at the external shelter when possible.

Regarding duration of the sampling period, according to Tesfahun et al. (2014), Tadesse and Komie (2014) and Zhang et al. (2013), seven consecutive days is the minimum necessary to obtain reliable data. After this first measurement, many researchers have monitored seasonal flow by averaging the amount of waste per patient, based on patient numbers per day, week or month.

After the mapping the HCW flow and generation sectors, the following steps should be carried out, adapted from the Guide for Internal Solid Waste Management in Health Facilities, published in 1997 by the Pan-American Health Organization (PAHO), as shown in Figure 1, for HCW quantification.

As observed in Figure 1, the first step is to follow a collector. After the bags are collected and labelled, the researcher must separate the bags by sector and leave them in containers.

At the end of the collection, the researcher must weigh each bag from each sector separately by waste group according national legislation. In Table 1, we present the Brazilian HCW classification based on two federal resolutions (ANVISA, 2004; CONAMA, 2005) and some examples of each group.

For gravimetric characterization, we suggest that source separation is the best procedure. In the gravimetric characterization, initial consideration should go to which HCW group will be analysed, in order to minimize occupational risks. In addition, to reduce the occupational hazards related to this procedure, the activity should be performed using all personal protective items (PPIs), in the shortest possible time and in the presence of the smallest number of people.

### 2.3 Step 3: Application of questionnaires and interviews

The objective is to obtain data that may not have been obtained through the other steps, to analyse the staff members' knowledge about practices related to HCW adopted in the institution and to observe the adequacy between discourse and practice.

The best survey instrument for interviews is the Waste Management of Health Services – a rapid assessment tool proposed by the WHO whose Brazilian version was translated and validated by Silva (2011).

Due to the broad scope of this instrument, we recommend only using tool D, which is focused on obtaining data and information from the hospital managers, the nurses responsible for hospital infection control and the HCW manager.

Among the results, the researcher should know the level of knowledge of the respondents regarding current legislation and HCWM.

## 3. EXAMPLE OF PROTOCOL APPLICATION

In this section, a case study conducted in Rio de Janeiro, Brazil, demonstrates the applicability and effectiveness of the proposed HCWM protocol.

### 3.1 Characteristics of the hospital analysed

The hospital analysed is considered a reference facility for obstetrics and has the following sectors: Pre-delivery,

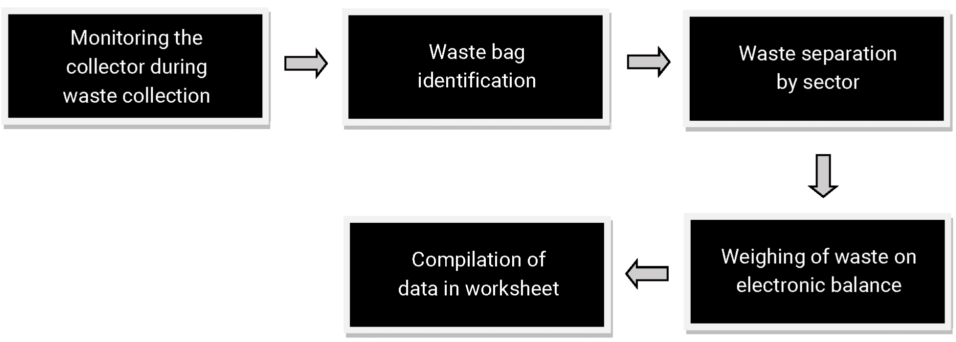


**FIGURE 1:** Steps of HCW quantification.

**TABLE 1:** Classification of Brazilian HCW.

| Groups | Description/examples |
|---|---|
| A | Wastes that represent risk to public health and the environment due to presence of biological agents such as: blood, bodily fluids, drainage fluids or excreta. Ex. surgical gloves, gauze, cotton, bandages; laboratory plates and blades and vaccines discarded. |
| B | Wastes that represent risk to public health and the environment, due to their characteristics of inflammability, corrosiveness, reactivity and toxicity.<br>Ex. some chemical products and pharmacological substances |
| D | Common waste represented by non-recyclable waste (organic waste) and recyclable waste ( paper, plastic etc.) |
| E | Sharp waste (used or unused syringes, butterfly needles, lumbar puncture and cannula needles) |

delivery and postpartum (PPP); Green; Lilac; Reception; Surgery Centre (SC); and Ambulatory. The PPP sector is where normal deliveries occur and it has 12 operating rooms, 2 nursing posts, 1 equipment room, 2 washbasins and 1 new-born room. In turn, SC has 2 operating rooms, 1 nursing post, 1 aesthetic recovery room, 1 equipment room, 1 new-born room and 3 washbasins. This is where caesarean deliveries are performed.

Regarding the Green and Lilac areas, they are where the parturients and their babies remain for about three days until hospital discharge. Green has 34 beds, 1 medication room, 1 nursery, 1 nursing post, 1 darkroom and 1 new-born room, while the Lilac has the same number of beds, 1 medication room and 1 nursing post. Reception is the place where patients register and any necessary initial care is provided. This place has 1 service desk, 1 hall and 1 baby changing room. The Ambulatory sector has 14 offices, 1 hall, 2 new-born changing room and 9 other rooms. It is where outpatients are attended, mainly pregnant in prenatal care and mothers and new-born children.

### 3.2 HCW generation in July and August

For this study, we decided to apply the quantification and gravimetric characterization steps for seven days in July and August, months which had the highest number of deliveries in recent years. However, in July we did not receive authorization for gravimetric characterization. Because of that, we performed quantification and gravimetric characterization in August.

Figure 2 presents the HCW generation (kg/d) in July and August.

The most representative sector in terms of total mass is the PPP, whose generation increased slightly between July and August. From the flow of the patients, we noted that the PPP is the sector where the vaginal deliveries are performed, a procedure that is the specialty of the hospital unit, corresponding to about of 75% of deliveries.

This type of delivery is associated with greater amounts of waste generated compared to caesarean delivery. In addition, the greater need for materials such as gauze bandages can explain the greater significance of this sector in terms of HCW generation.

Based on the results of the HCW quantification (kg/d), we noticed that although waste monitoring for two weeks only is too brief to capture seasonal changes, it is possible to learn the most representative sectors regarding HCW generation.

### 3.3 Contribution by sector - Group D (common waste)

Figure 3 shows the contribution by sector of common waste.

Reception is the sector with the main contribution of non-hazardous waste generation. This sector receives emergency care patients and consequently has a high turnover of pregnant women. Furthermore, this sector is used to store wastes from administrative areas (reception desk and management rooms).

In both July and August, the least significant sectors in the common waste generation were respectively the SC and Ambulatory. This may be related to the types of procedures performed in these sectors. The SC is an isolation area aimed at obstetrical procedures where the most significant waste masses consist of materials that can contain pathogens and pose a risk of disease transmission, such as gauze bandages, compresses contaminated with blood and other body fluids, all of which are classified as infectious waste (if there is no failure in segregation). The fact that the SC sector's contribution is lower than that of the PPP, although they are sectors with the same purpose, can be attributed to the fact that in the SC there are only two surgical rooms while in the PPP there are 12 rooms for deliveries.

In relation to Ambulatory, the main procedures were focused on queries posed to a multi-professional team, so this sector does not generate a significant amount of common waste.

### 3.4 Contribution by sector - Group A (infectious waste)

The same type of evaluation was carried out regarding the participation of each sector in the generation of infectious waste. The results are presented in Figure 4.

Among the sectors, the Reception presented the largest increase in the contribution of infectious waste, from 4% to 9%. This increase was caused by a change in the types of services performed in this sector due to the overcrowding of the hospital and the increase in the number of emergency patients. This meant more frequent deliveries in this sector, changing the types of infectious wastes, in particular increasing the volumes of gloves, gauze bandages and compresses with blood that could contain pathogens and could be a risk of disease transmission if not properly stored.

The predominance of the PPP and SC in infectious waste generation is related to the obstetrical procedures performed in these sectors, with a considerable influence on the characteristics of HCW generated. These wastes are usually composed of materials such as compresses and gauze bandages contaminated with blood and other organic fluids.

### 3.5 Interviews

The interviewees were selected based on the details of tool D, which refers to the director of the hospital, the head

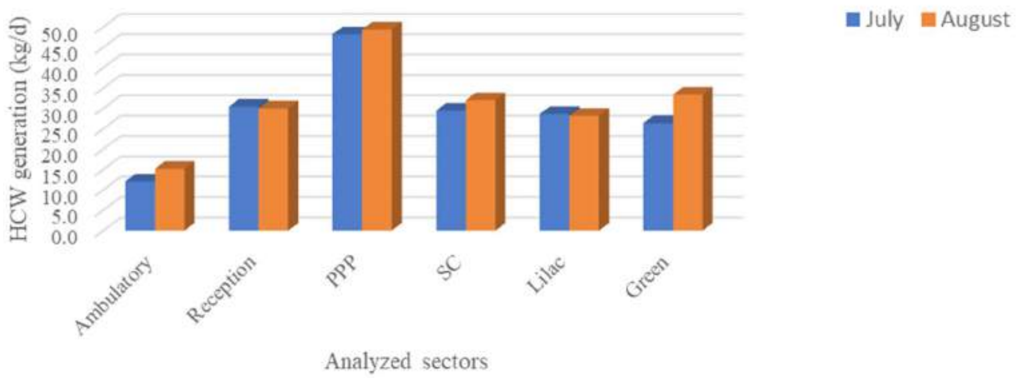


**FIGURE 2:** HCW generation (kg/d).

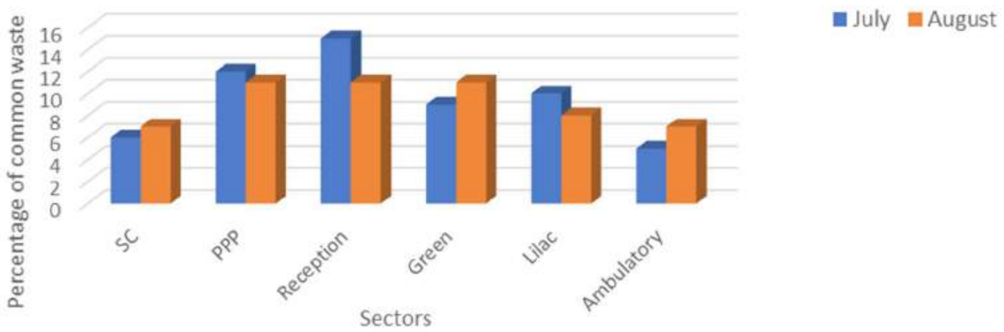


**FIGURE 3:** Common waste generation.

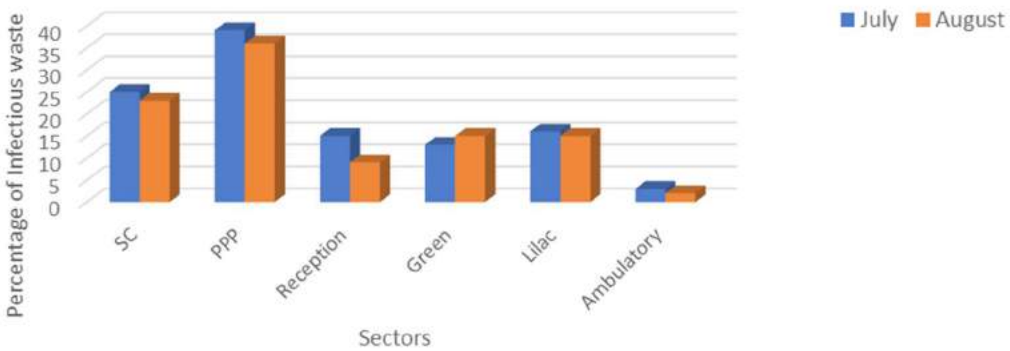


**FIGURE 4:** Infectious waste generation.

nurse for infection control and the HCWM manager. The main results are presented in the sections below.

*3.5.1 Interview with health institution manager*

Table 2 presents some items of the questionnaire in which the interviewee's answers served to corroborate the information obtained by the previous data collection tools.

According to the institution's manager, the hospital is officially classified in the National Registry of Health Establishments as an institution with medium complexity, but in practice it should be classified as having high complexity, based on the following argument:

*"In practice the hospital can be considered highly complex, because it caters to all the patients who come here. It should be a closed-door hospital, which would reduce its complexity, but this is not what happens in practice. There is greater control in the outpatient clinic, but not in the hospital."*

Thus, the hospital ends up being basically an "open door" hospital, especially for pregnant women in the region, and this may be one of the explanations for the manager's response about the number of beds occupied. In this statement, the respondent established in approximate numbers what was observed during the period of observation, quantification and the gravimetric characterization of the sectors, namely overcrowding, exemplified by the fact that the Reception focused mainly on emergency care, reflected in the generation of postpartum absorbents most commonly found in the PPP and Lilac sectors.

In the team/staff item, according to the administrative manager, the training contemplates HCWM, under the responsibility of the nursing manager, who is also responsible for HCWM.

The statement from the manager indicates what that everything that refers to HCW is under the responsibility of the nursing manager. This indicates there is no effective participation of the other sectors in HCWM.

According to the administrative manager, there was only one training session focused on the standardization of the Healthcare Waste Management Plan. This training was inadequate, since this document should be prepared according to the specific characteristics of the institution in which it will be implemented. In addition, there should be continuous training of employees, especially those who are responsible for the training of others.

*3.5.2 Interview with the nursing manager*

Table 3 shows the items and excerpts from the comments that provided evidence about the HCWM, facilitating triangulation of the results obtained in the other stages.

With regard to the number of beds occupied, the respondent was not certain about the right number.

As for hospital size, the respondent, despite not being able to specify the level of overload, tried to exemplify what had been observed in the institution and mentioned the maintenance of four more children in the Intensive Care Unit.

When questioned about the average number of outpatients and the number of outpatients attended per day, she again showed imprecision. Lack of information about the number of beds occupied and number of patients attended directly influences HCWM, since there is a positive correlation between the number of patients and HCW generation.

In the item staff/health professionals, the head nurse pointed out that all health professionals are immunized for hepatitis and tetanus. However, she did not know how many accidents with sharp objects had occurred in recent months. The low number of sharp object injuries in the hospital may be a reflection of the qualification process cited by the nurse or underreporting by staff, since the presence of syringes with bare needles among non-hazardous waste was noted during the gravimetric characterization.

Regarding the regulations on the Waste Management Plan, despite the nurse's response regarding training, she is not the person who manages the training of employees of the outsourced company, since they are trained by that company's nurse. Regarding this training, COL 1 gave the following descriptions:

*"The hospital rarely trains employees, and there is high employee turnover."*

*"It does not have specific training for waste; the company follows what the hospital does."*

*"I did not get training here, the way I work I brought from other places."*

These comments reflect a lack of training in the institution. While stressing the fact that these training sessions were in line with the routine of the health establishment, he emphasized the lack of focus on HCW in this training.

COL 2, when asked about his work, responded as follows:

*"I went in as a collector and they explained to me where the containers were, the need to wash the shelter and the walls. The service they taught me was a general service."*

This indicates there was enumeration of the activities that should be performed and a basic explanation about containers' location and what they should store. However, there was no specific orientation for the collection activity,

**TABLE 2:** Interview with health institution manager.

| Question | Excerpt from the interviewee's response |
|---|---|
| What type of institution and hierarchy level? | *"In practice the hospital can be considered highly complex, because it caters to all the patients who come here."* |
| What is the number of beds occupied? | *"There is an excess of demand, the hospital is overcrowded."* |
| Is there any type of training? | *"You can get more information directly from the nursing manager."* |
| Is there any type of training for HCWM? | *"The only training received was about standardization of the Healthcare Waste Management Plan ... "* |
| What is the composition of HCWM? | *"All the waste part is in charge of the nursing manager ..."* |
| How much financial resource is allocated to HCWM? | *" ... there should be a direction to decrease this percentage due to the scarcity of financial resources."* |

**TABLE 3:** Interview with nursing manager.

| Question | Excerpt from the interviewee's response |
|---|---|
| How many beds are currently occupied? | *"It's not easy to pinpoint, but you can call it overcrowded."* |
| What is the average occupancy rate and number of outpatients attended? | *"Unfortunately there is no current information ... "* |
| How many injuries from sharp objects have been reported in the last 12 months? | *"Usually 2 to 3 cases per year."* |
| You have a specific HCW colour system? | *"It's kind of complicated to remember these colours because every hospital works in its own way"* |
| Do you think current transportation and collection practices provide sufficient safety? | *"When there is food distribution, there is no passage of healthcare waste."* |
| Is there exposure of written instructions and training regarding the HCWM? | *"Yes, but as soon as the meeting is over, everything goes back to how it was before."* |

and training similar to that of a general service assistant was given.

### 3.6 Gravimetric characterization of common waste (D)

Gravimetric characterization of common waste in the Reception and PPP sectors is presented in Figure 5.

With regard to the figure, it is important to highlight that in the Reception area there was prevalence of paper and in PPP there was prevalence of plastics. Regarding paper, there was predominance of asepsis papers and toilet paper, a finding that may be related to the fact that it is an emergency care sector and consequently has a high turnover of pregnant women and a large number of hospital procedures. The waste paper found in the PPP was quite similar to that observed in the Reception, mainly composed of toilet paper and asepsis paper.

In the PPP, among the plastic waste most commonly observed, a contribution of 57% can be highlighted of absorbents used after childbirth. The presence of these absorbents is in accordance with the characteristics of the PPP. However, the main function of these absorbents is to contain bleeding. These wastes may contain pathogens and could be a risk of disease transmission, mainly because some patients are HIV positive and have hepatitis B.

The segregation of this type of waste as common waste shows failure in the training of health professionals, since this type of waste should be segregated and packaged in infectious waste bags.

Bloody postpartum absorbents mixed with common waste (37.5% of the total plastic mass) was also noticed in the Reception sector.

As for plastic waste, we observed in both sectors the presence of potentially recyclable waste such as plastic cups, syringe packs and plastic bags. But we also observed some infectious waste that could contain pathogens such as serum bags and serum equipment. The presence of this waste indicates failure of the HCWM.

Another plastic waste observed was syringes without needles and syringes with capped needles. The presence of needled syringes indicates that employees do not follow the established disposal recommendation regarding sharp objects. Furthermore, the disposal of syringes without needles indicates that the professionals of this sector removed the needles, which is not recommended due to the possibility of accidents.

With regard to occupational risk, the simple segregation of syringes with bare needles as common waste places a considerable occupational risk on the people who handle the HCW.

## 4. MANAGERIAL IMPLICATIONS

This article is important for health institution managers, healthcare waste managers and researchers, by providing information to improve HCW management. The proposed protocol involves qualitative and quantitative tools, providing a useful, quick and practical assessment method that can be used by researchers and health managers to improve HCWM practices. Furthermore, application of the protocol built a database for health institutions about the evolution of HCWM over the years.

Among the main advantages of this protocol in comparison to the other ones, we can mention that it is based on evidence obtained by qualitative tools, such as interviews, questionnaires and observation, and by quantitative tools for characterization, in an attempt to minimize the limitations of each of the tools used.

Consequently, this protocol can provide detailed information about the breakdown of HCW generation by sector and by type, to support efforts to minimize and properly segregate and manage HCW, in compliance with internal rules and general legislation and regulation. In addition, through the comments of health professionals involved in HCWM, it elicits information about their knowledge and practices, providing support for training programs.

## 5. CONCLUSIONS

In this paper, we propose and report the application of an integrated protocol to provide qualitative and quantitative information about HCWM, using literature review as a guide. Furthermore, the protocol can provide some basic methods for managers to implement HCWM in health establishments that do not have a formal procedure and to improve practices in institutions that already have a HCWM program.

A case study was presented to demonstrate the principal results obtained using this protocol. In general, the

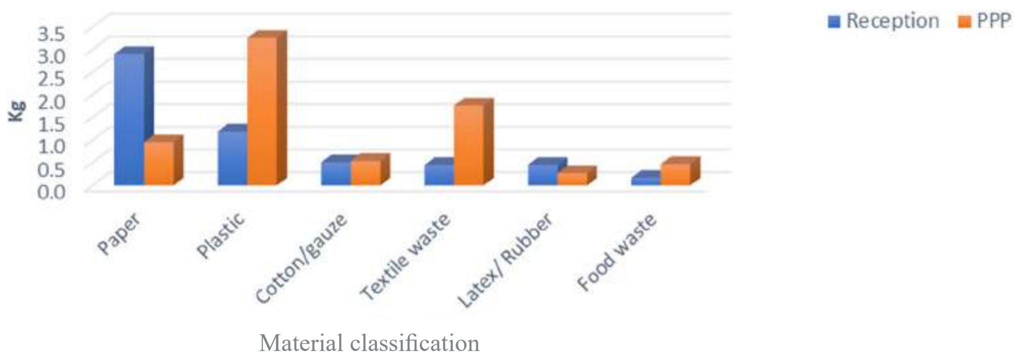


**FIGURE 5:** Gravimetric characterization.

application of the protocol was satisfactory, since even with the adaptation of the protocol to the hospital's reality, all the proposed steps were carried out and the expected results in each of the steps were obtained.

Although the proposed protocol was developed to be applied in any health unit, future studies will be carried out in health units with different characteristics. We hope this protocol will be applied in both developing and developed countries and improvements will be made by other authors and health professionals.

Other studies can focus on the information obtained by the protocol and create a forecast model for the generation and composition of HCW in order to provide support to improved management.

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# ARE GENDER PERSPECTIVES INCLUDED IN EDUCATION FOR SUSTAINABLE CONSUMPTION AND WASTE EDUCATION PROGRAMS? A SYSTEMATIC LITERATURE REVIEW

Letícia Sarmento dos Muchangos [1,2,]* and Philip Vaughter [1]

[1] UNU, Institute for the Advanced Study of Sustainability (UNU-IAS), Shibuya, Tokyo, Japan
[2] Keio University, Graduate School of Media and Governance, Fujisawa, Kanagawa, Japan

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**ABSTRACT**
Education for sustainable consumption (ESC) has a role in the provision of knowledge, values, and skills to enable individuals and social groups to become actors of change towards more sustainable lifestyles. Also, it must consider the interests, needs, and perspective of critical stakeholders, empower them and enable their full participation in public debates about sustainability. Though unsustainable consumption is substantially shaped by gender-based inequalities, ESC itself tends to be gender-neutral. This study provides a systematic literature review of research on ESC, with a focus on waste education, and an assessment of the inclusion of gender perspectives in those publications. A structured tool Proknown-C was applied and 46 articles were retrieved and analyzed – 28 generally related to ESC and 18 to waste education in particular. The articles were published intermittently between 1990 and 2017, with case studies on 32 countries. From the aggregate number of articles, 22 addressed gender in some form, however, in its majority, it was limited to superficial mentions and to the presentation of the study subjects in a sex-disaggregated manner, with fewer exploring gender and its relationships with ESC and waste education. This study offers a contribution to the scholarly and practical debate on ESC and waste education programs, and the inclusion of gender perspectives, as well as, kindle in-depth research on the subjects. It also aims to provide points for better understanding the role of gender perspectives in promoting sustainable consumption practices and lifestyles.

## 1. INTRODUCTION

Unsustainable consumption and production is a tremendous environmental problem and is, therefore, a top priority for the global environmental agenda (UNEP, 2016). In 1992, resulting from the Rio de Janeiro UN Conference on Environment and Development, Sustainable Consumption and Production (SCP) was acknowledged as an all-embracing theme linking environmental and development challenges (UNEP, 2010). More recently, in the 2030 Agenda for Sustainable Development, SCP gained a central role for the achievement of sustainable development, reflected in its identification as a stand-alone Sustainable Development Goal (SDG 12) and also as a cross-cutting issue in many of the 17 goals and 169 targets adopted in the agenda (Ekins and Lemaire, 2012; Statistics Sweden, 2016). SCP relates to the use of products and services to respond to basic needs and provide a better quality of life, with minimal usage of natural resources, toxic materials, and emissions of by-products throughout the life cycle of those products or services, ultimately protecting the needs of future generations (UNEP, 2012).

Gender inequality issues are determinant and fundamentally shape unsustainable consumption and production practices (Sexsmith, 2012). This is particularly true in the consumption field, though consumer choices are also influenced by income level, social status and biases, however, gender is still a significant driver (OECD, 2008; Schultz and Stieß, 2009). Women tend to have less resource-intensive (and therefore more sustainable) lifestyles and display more pro-environmental attitudes, whether rich or poor, while men's consumption patterns and ecological footprint are usually greater than women's (Bulut et al., 2017; Chant, 2006; OECD, 2008). In addition, the gendering of non-sex-differentiated products and services is a standard marketing tool used to assign gender characteristics and increase demand and consumption for given products (UNEP, 2016). The Global Gender and Environment Outlook (2016) for instance, provide informative examples

* Corresponding author:
Letícia Sarmento dos Muchangos
email: leticia.muchangos@unu.edu


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of how femininity and masculinity norms steer meat, car and personal cosmetics' consumption. On the other hand, gender-based inequalities and social exclusion are key factors undermining people's ability to anticipate and prepare for major disasters, as well as; it shapes their susceptibility and exposure to the impacts of climate change (Le Masson, Norton, and Wilkinson, 2012). Women and girls' figure as more socially, economically and environmentally vulnerable to the effects of unsustainable consumption and production practices, such as extreme weather events and disasters. For example, studies have been reporting that climate change-related disasters and catastrophes, prompt gender-based impacts, with women and girls suffering from injuries, and diseases more than the men, as well as, being more prone to suffer abuse and sexual violence in post-disaster situations (OECD, 2008; Schultz and Stieß, 2009; Stevens, 2010).

To address the current global unsustainable consumption challenges, raising sustainability awareness in individuals is imperative. As such, education is indispensable (Avan, Aydinli, Bakar, and Alboga, 2011; McKeown, Hopkins, Rizi, and Chrystalbridge, 2006). As Stanszus et al. (2017) stated, "The nexus of education, consumption and sustainable development has been at the top of the agenda since the very inception of the political process towards sustainable development at the Rio Conference in 1992." Accordingly, Education for Sustainable Consumption (ESC) has the specific role of provisioning knowledge, values, and skills to enable individuals and social groups to become actors of change towards more sustainable consumption lifestyles. Consumers' rights are therefore integrated into ESC, in order to empower them and to enable their full participation in both public debate and the economy in an informed, confident and ethical way (Choi and Didham, 2009; Stanszus et al., 2017; UNEP, 2017).

Nevertheless, ESC is commonly gender-neutral and ignores gender-based inequalities, even though those same inequalities determine and aggravate unsustainable consumption practices (Gough, 2016; OECD, 2008; Sexsmith, 2012; Stevens, 2010). Despite the scholarly recognition of a gender influence in ESC, a state-of-the-art analysis is still lacking. As far as the authors understand, there are no systematic review studies that address ESD in general nor ESC and gender specifically.

This paper, therefore, presents an examination of the research developments around ESC, focusing on the subject of waste education and the inclusion of gender perspectives in waste education. Waste is a component of SCP and invariably a relevant theme of ESC (Thoresen, 2010; UNEP, 2010). Waste management is present in three targets of SDG 12 (United Nations, 2016):

- Target 12.3 By 2030, halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses;
- Target 12.4 By 2020, achieve the environmentally sound management of chemicals and all wastes throughout their life cycle, in accordance with agreed international frameworks, and significantly reduce their release to air, water and soil in order to minimize their adverse impacts on human health and the environment;
- Target 12.5 By 2030, substantially reduce waste generation through prevention, reduction, recycling, and reuse.

The specific objectives of this study are to: (1) comprehend what is the current state of the research literature on ESC and waste education; (2) identify instances were publications include gender perspectives; and, (3) analyze the extent and specific content of these gender perspectives' inclusion.

## 2. MATERIALS AND METHODS

The purpose of a systematic literature review is to present a balanced and impartial summary of previous relevant and quality research to inform the researchers' decisions (Crombie and Davies, 2009). Compared with traditional literature review approaches, systematic review offers a structured review format, which is more transparent and replicable (Schulze, Nehler, Ottosson, and Thollander, 2016). Furthermore, a systematic review should encompass a comprehensive and systematic search for studies; an explicit and reproducible strategy for screening and including the studies, as well as, for data extraction; a detailed report and analysis of the results; and, a presentation of potential future pathways for research and practice (Caiado, de Freitas Dias, Mattos, Quelhas, and Leal Filho, 2017). To build up knowledge for the completion of this systematic review, an adaptation of the tool Knowledge Development Process intervention instrument - constructivist (ProKnow-C) was applied. Proknow-C was developed and presented in 2008, by the Laboratory of Multi-criteria Methodologies in Decision Support, from the Federal University of Santa Catarina in Brazil. The primary goal of such a tool is to aid in the process of selection and analysis of bibliographic material in a given subject, through a scientific process (Ensslin, Dutra, Ensslin, Chaves, and Dezem, 2015; Rosa, Petri, Matos, Ensslin, and Ferreira, 2015). Proknow-C follows a constructivist process that results in the formation of an in-depth understanding of the surveyed subject, based on the researcher's interests and limitations (Rosa et al., 2015). It aims to mainly address the lack of structure in the process of identification, selection, and analysis of bibliographic references with scientific recognition, in what relates to the connection between the research objectives and the alignment and relevance of the retrieved content (Ensslin et al., 2015). ProKnow-C application includes the following steps: (1) definition of the bibliographic portfolio (BP); (2) bibliometric analysis of the portfolio; (3) systemic analysis; and (4) definition of the research question and its objectives (Caiado et al., 2017; Ensslin et al., 2015; Viegas et al., 2016).

- For the BP definition (step 1), initially, four broadly recognized scientific electronic database were selected - ERIC, Scopus, Science Direct, and ISI Web of Science (WoS), with the search procedure combining the search terms in the titles, abstracts, and keywords. Two search strings were followed: the term 'education for sustainable consumption' corresponded to the ESC theme search, and the terms, 'waste education,' 'waste awareness,' 'waste teaching, and 'waste learning,' corresponded to the overall waste education theme search.

- Continuing with the BP definition, inclusion and exclusion criteria were applied to the results retrieved from the previous activity. In defining the BP, the intended subject for knowledge development is decided by the researcher (Ensslin et al., 2015). In this case, five criteria were selected: the type of publication; research discipline; relevance to the study, language, and availability. For the type of publication, only peer-reviewed academic journal articles were considered. This choice was based on the notion that those commonly retain a higher level of quality and are more readily available (Garza-Reyes, 2015; Schulze et al., 2016). For the research discipline, within the ESC string, no criterion was applied, while for waste education, articles related to the Waste Management discipline were included. On the relevance to the study, it was limited to studies addressing education programs (focusing on solid waste education for the waste education string), and excluded, for example, articles only reporting on education awareness levels, and articles with content outside of the scope, such as reports on waste characterization and generation estimations. The final two language, and availability, considered the inclusion of articles in English and Portuguese and articles where the full text was available online, with no time-period limitation. As a result, a total of 1996 articles were identified and initially stored in the reference management software, Endnote.
- Next, an automatic screening of duplicate articles in EndNote, followed by a manual screening of the titles and abstracts to check the alignment with the subject of interest resulted, in the exclusion of additional articles. The screenings yielded a total of 62 articles, which were downloaded for full-text analysis, resulting in the final number of 46 articles being used for the steps 2 and 3 of the Proknow-C tool respectively, and for the gender inclusion assessment.

A summary of the steps performed to select the papers of the BP is presented in the Table 1.

To assess the inclusion of gender perspectives in the BP, a subsequent word search query within the 46 articles was completed, comprising the words, 'gender,' 'sex', 'woman,' 'women,' 'man,' 'men', 'male', 'female', 'girl(s)' and 'boy(s).' The bibliometric analysis and the gender perspective assessment were performed through qualitative data analysis, a task supported by the use of software QSR NVivo (QSR International Pty Ltd, 2017). The list of articles included in the literature review is displayed in Table 2.

## 3. RESULTS AND DISCUSSION

### 3.1 Bibliometric analysis

*3.1.1 Journal of publication*

Generally, the majority of the articles were published in Procedia – Social and Behavioural Sciences (9 articles), the International Journal of Environmental and Science Education (6 articles) and Waste Management (5 articles), with all articles found within the waste education string. The remainder English language journals and the two Portuguese language journals within the waste education string - Revista Eletrônica de Administração and Cadernos EBAPE.BR, each presented one article (Figure 1). For the ESC string, five articles were published in Procedia – Social and Behavioural Sciences, while three articles were in the International Journal of Environmental and Science Education.

*3.1.2 Distribution across time and geographical focus*

The articles were found to be fragmented in time and location. The publication period spanned from 1990 to 2017, with 2016 presenting the highest number of publications (9 articles), while no publication was found for eight consecutive years in the 1990s (1991-1998), nor for seven consecutive years in the early 2000s (2000-2006), and none in 2010 (Figure 2). Most publications occurred after 2010, with all articles in 2012 addressing ESC, with and all articles in 2014 doing the same, followed by four publications on ESC in both 2016 and 2017. Individually for waste education, the earliest publication was in 2007, and most articles were published on this topic in 2013 and 2016, four and five articles, respectively. In almost three decades, more than 80% of all articles were published in the period between 2011 and 2017. The steady increase of research studies on this years can likely be linked to the outcomes of the UN Decade of Education for Sustainable Development (2005-2014), declared in 2002, followed by the launch of the UNESCO's Global Action Programme (GAP) in 2014, as well as from the global adoption of the SDGs (UNESCO, 2007, 2014; United Nations, 2016).

The distribution across continents of study locations, suggests a global interest in the subject, mainly from higher-income economies, such as the United States of America, United Kingdom, Germany, Italy, and Australia. Middle-income economies such as Indonesia, Malaysia, and Brazil, had two articles each, all focusing on waste education, except Turkey, which had one article on ESC and another on waste education. The remaining locations

**TABLE 1:** Literature search process.

| | Education for sustainable consumption | Waste Education |
|---|---|---|
| Databases | Eric; ScienceDirect, Web of Science; Scopus | |
| Abstract, Title, Keywords | education for sustainable consumption | waste education, waste awareness, waste teaching, waste learning |
| | 9,157 | 20,375 |
| Criteria for inclusion/exclusion | 976 | 1020 |
| Exclusion of duplicates and Title and Abstract review | 32 | 30 |
| Full-text analysis | 28 | 18 |
| Final results | 46 | |

**TABLE 2:** Articles analyzed in the review.

| Authors | Keywords | ESC Theme specific / Waste type |
|---|---|---|
| (Brumby et al., 2011) | N/A | Alcohol consumption |
| (Çelikler and Harman, 2015) | SCAMPER technique; Solid waste; Awareness; Science student | Solid waste |
| (Chalfoun, 2014) | Campus greening; energy efficiency; energy simulation; hands-on education | Energy conservation and efficient use |
| (Chandra, 2014) | Environmental education for sustainability; Indigenous perspectives; Sustainable development; Traditional ecological knowledge; Western science | Traditional knowledge |
| (Crawford, Luke and Van Pelt, 2015) | Curriculum; inquiry; environmental sustainability; plastic pollution; global citizenship; STEM | Packaging |
| (Danilane and Marzano, 2014) | Consumer education; consumer education; social learning theories; ducat; education for sustainable development; social learning theories | Consumer education |
| (Davis, 2008) | N/A | Waste management |
| (Finlayson et al., 2017) | Undergraduate research; Pedagogy; Case study; Food waste; Agriculture | Food waste |
| (González-Gaudiano, 1990) | N/A | Development of environmental education for sustainable consumption |
| (Gough and Scott, 1999) | N/A | Tourism |
| (Hadiyanti, 2016) | A group approach; Empowerment; Waste recycling | Household waste |
| (Hadjichambis et al., 2015) | Decision-making; Environmental education program; Environmental representations; Sustainable consumption | Consumer education |
| (Hawas and Al-Habaibeh, 2017) | Building energy efficiency; education; innovation; public engagement; simulation; thermal performance | Energy |
| (Indrianti, 2016) | Household waste; Waste bank; Sustainable; Education; Quran education park | Household waste |
| (Kamaruddin, Pawson and Kingham, 2013) | Social learning; Sustainable waste management; NGO | Municipal solid waste |
| (Kanchanabhandhu and Woraphong, 2016) | Model; solid waste management; Multilateral cooperation; Semi-urban community | Solid waste |
| (Kearns et al., 2013) | learning cities; learning festivals; lifelong learning | Learning festivals |
| (Kossieris, Kozanis, et al., 2014) | Analytics; End-user; ICT; Smart metering; Urban water management; Water-energy nexus; Web Implementations | Water |
| (Kossieris, Panayiotakis, et al., 2014) | E-learning; Moodle platform; Online Implementation; Online education; Smart metering; Water demand management; Water efficiency | Water |
| (Kumar, Somrongthong and Shaikh, 2015) | Health care workers; Waste management; Infectious waste; KAP; Quasi-experimental study | Intensive Healthcare waste |
| (Nowak et al., 2009) | Community-based eco-pedagogy; Eco-centered Early Childhood Education; Parent involvement Introduction | Consumer education |
| (Leger and Pruneau, 2012) | Climate change mitigation; environmental behavior; environmental competences; environmental education; family systems | Climate change |
| (Maddox et al., 2011) | Waste management; Recycling; Environmental education; Sustainability; Intergenerational influence | Household waste |
| (Mostowfi, Mamaghani and Khorramar, 2016) | Board game; Fun toolkit; Children psychology; Educational game; Environmental education; Recycle game | Household waste |
| (Nagata et al., 2014) | Customer involvement; Platform for consumer and suppliers; Rebuild relationship between consumers and retaile | Supply chain: Retailert-Consumer |
| (Nowak et al., 2009) | N/A | Product life cycle and traceability |
| (Oliver, 2016) | Carbon footprint; Case study; Environmental behavior; Role-play; Transformative learning | Carbon footprint |

| Authors | Keywords | ESC Theme specific / Waste type |
|---|---|---|
| (Oto, Cobanoglu and Geray, 2012) | Environmental bioethics; environmental education; environmental sustainability; sustainable airport | Environmental bioethics |
| (Painter, Thondhlana and Kua, 2016) | Food waste; Universities; Interventions; Food waste prevention; South Africa | Food waste |
| (Pankina et al., 2016) | Consumer culture; design; ecological design; ecological culture; ethics of design | Consumer education |
| (Patterson et al., 2009) | N/A | Conservation of natural resources |
| (Pérez-Belis, Bovea and Simó, 2015) | Waste electrical/electronic equipment; WEEE; Environmental education; Toy Survey | Waste electrical and eletronic equipment |
| (Polanec, Aberšek and Glodež, 2013) | Awareness; Educational concept; Adult education; Microtraining; Waste management | Solid waste |
| (Radulescu and Radulescu, 2011) | Consumer; Education; Healthy environment; Right to a healthy environment | Marketing |
| (Redman, 2013) | Sustainability education; Transformative change; Pro-environmental actions; Sustainability competencies; environmentally responsible consumption | Household waste |
| (Ribas et al., 2017) | Social Responsibility Environmental Management; Environmental Marketing | Solid waste |
| (Roeder, Scheibleger and Stark, 2016) | Self-perception theory; social labelling; sustainability communication; teaching method | Sustainable manufacturing |
| (Ruini et al., 2016) | Carbon footprint; ecological footprint; mediterranean diet; sustainability | Food consumption |
| (Schreinemachers et al., 2017) | Education; Food behavior; Impact evaluation; Malnutrition; Nutrition-sensitive agriculture; Randomized controlled trial; Vegetables | Food and sustainable agriculture |
| (Stanszus et al., 2017) | curriculum development; education for sustainable consumption; ethics; intervention design; mindfulness; mindfulness-based stress reduction; sustainable consumption; values | Mindfullness |
| (Traversa et al., 2017) | Children; food safety; nutrition | Food |
| (Vieira and Echeverria, 2007) | Public administration; Environmental education; Sustainability; Urban development | Municipal solid waste |
| (Wahba, 2012) | Environmental Advertising; Green Advertising; Greenwashing Advertising; Think Sustainability | Advertising design |
| (Yang, Chien and Liu, 2012) | Foreign Countries; Learning Motivation; Learning S; Taiwan | Energy conservation |
| (Zain et al., 2013) | Sustainable; Entrepreneurship; Education; Innovation; 3R | Municipal solid waste |
| (Zarate, Slotnick and Ramos, 2008) | N/A | Household waste |

identified had one article each, with two articles not being location-specific. Moreover, despite the broad geographic distribution, only one article focused on cross-country research - a case study comparing Canada and the Solomon Islands (Figure 3).

## 3.2 Content analysis

Figure 4 presents a summary of the main findings of the content analysis, described in the following sub-sections.

### *3.2.1 Study methodologies*

The study methodologies applied were also analyzed according to the application of quantitative, qualitative or mixed research methods. Commonly, for quantitative methods, the emphasis is on techniques such as descriptive statistics, regression models and parametric analysis. On the other hand, techniques that characterize qualitative methods include literature review, media analysis, case studies, interviews and focus groups, historical and ethnographic analysis, theory, and discourse analysis (Aikens, McKenzie, and Vaughter, 2016; Caiado et al., 2017; Palomo, Figueroa-Domecq, and Laguna, 2017; Schulze et al., 2016).

None of the retrieved articles used only quantitative methods. A total of 31 articles (68%), presented qualitative methodologies and 15 articles (32%) had a combination of quantitative and qualitative and methods – a mixed methodology.

The majority of articles reported on education initiatives - programs, workshops, frameworks, web tools, and field experiments (36 articles). These included either all phases of the initiative (problem description/background, design, implementation, and evaluation of results), a combination of some of the phases, or a description of a single phase. For instance, in six articles the focus was on the design process of programs, frameworks, or a lesson plan (Hadiyanti, 2016; Kossieris, Panayiotakis, et al., 2014; Leger and Pruneau, 2012; Nowak, Hale, Lindholm, and Strausser,

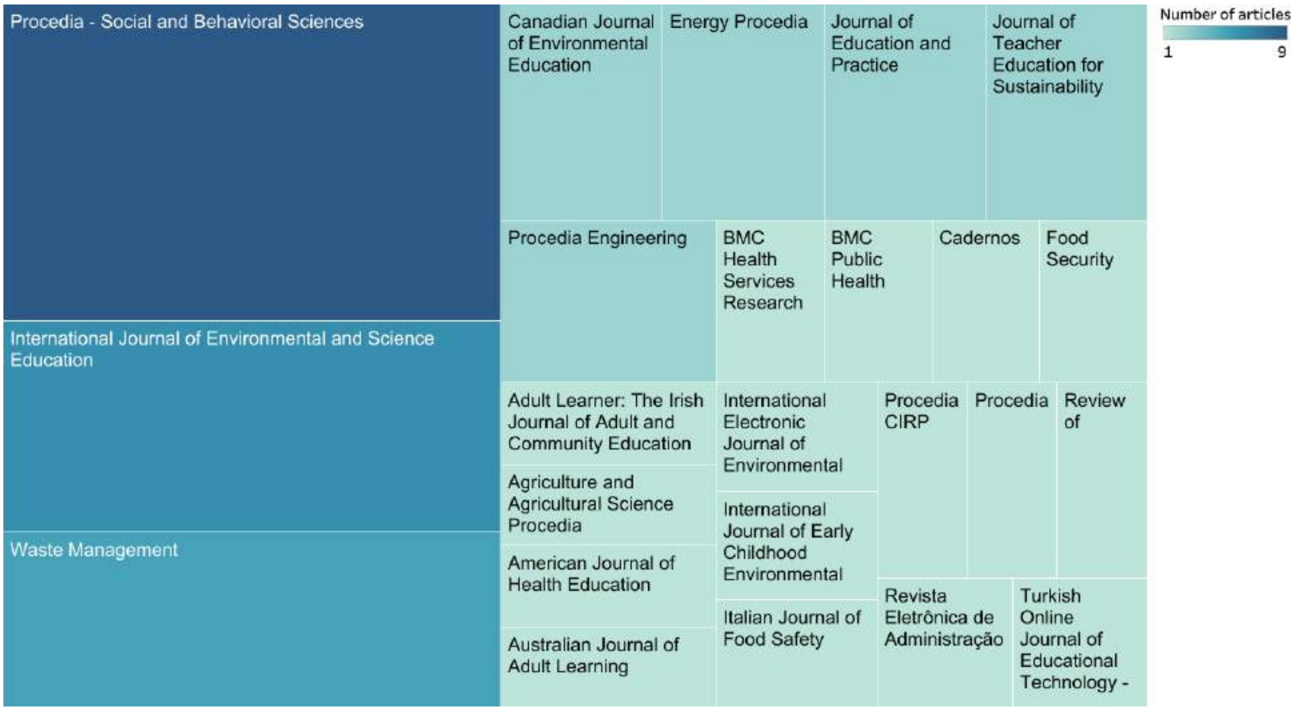


**FIGURE 1:** Number of articles per journal.

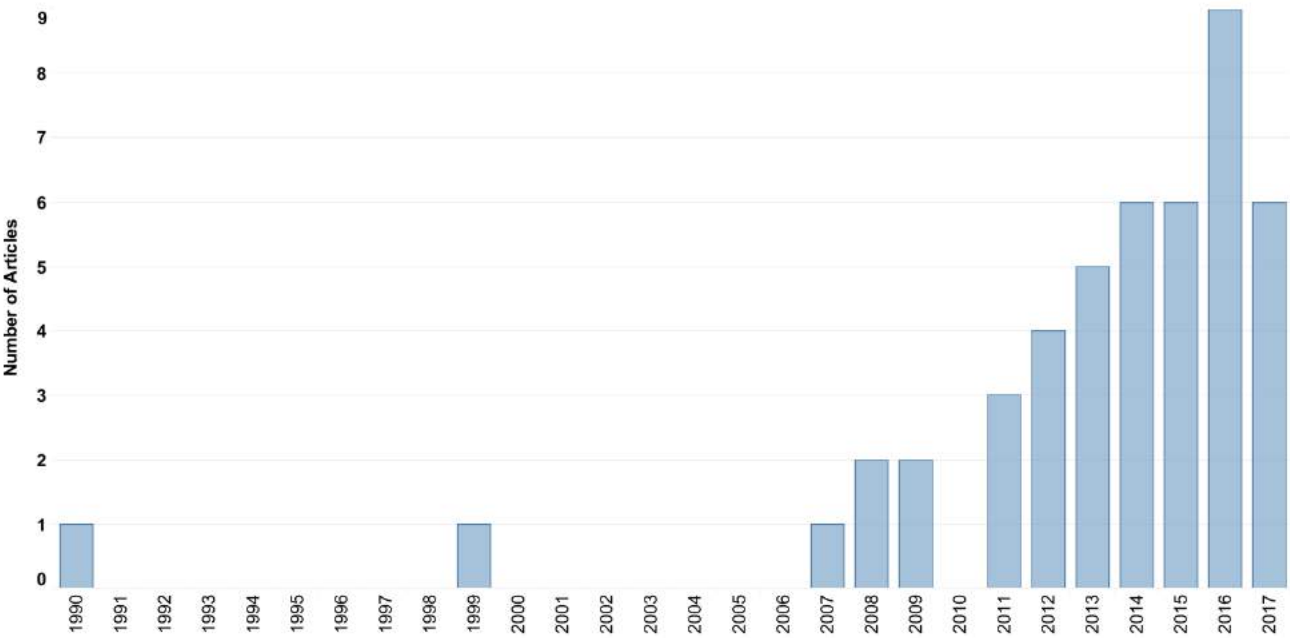


**FIGURE 2:** Articles published from 1990 to 2017.

2009; Polanec, Aberšek, and Glodež, 2013; Ribas, Vicente, Altaf, and Troccoli, 2017). There were also literature review studies, at times resulting in curriculum development and methodology proposals (Chandra, 2014; Davis, 2008; Oto, Cobanoglu, and Geray, 2012; Stanszus et al., 2017; Wahba, 2012). The remaining articles include a policy framework analysis, problem evaluations, the description of the implementation of a theoretical framework, and cultural and historical analysis (González-Gaudiano, 1990; Painter, Thondhlana, and Kua, 2016; Pankina, Khrustalyova, Egarmin, and Shekhova, 2016; Ruini et al., 2016; Vieira and Echeverria, 2007).

*3.2.2 Type of education and thematic focus*

The classification for types of education was based on the OECD definition of forms of learning: formal, non-formal and informal learning (Organisation for Economic Co-operation and Development (OECD), 2005):

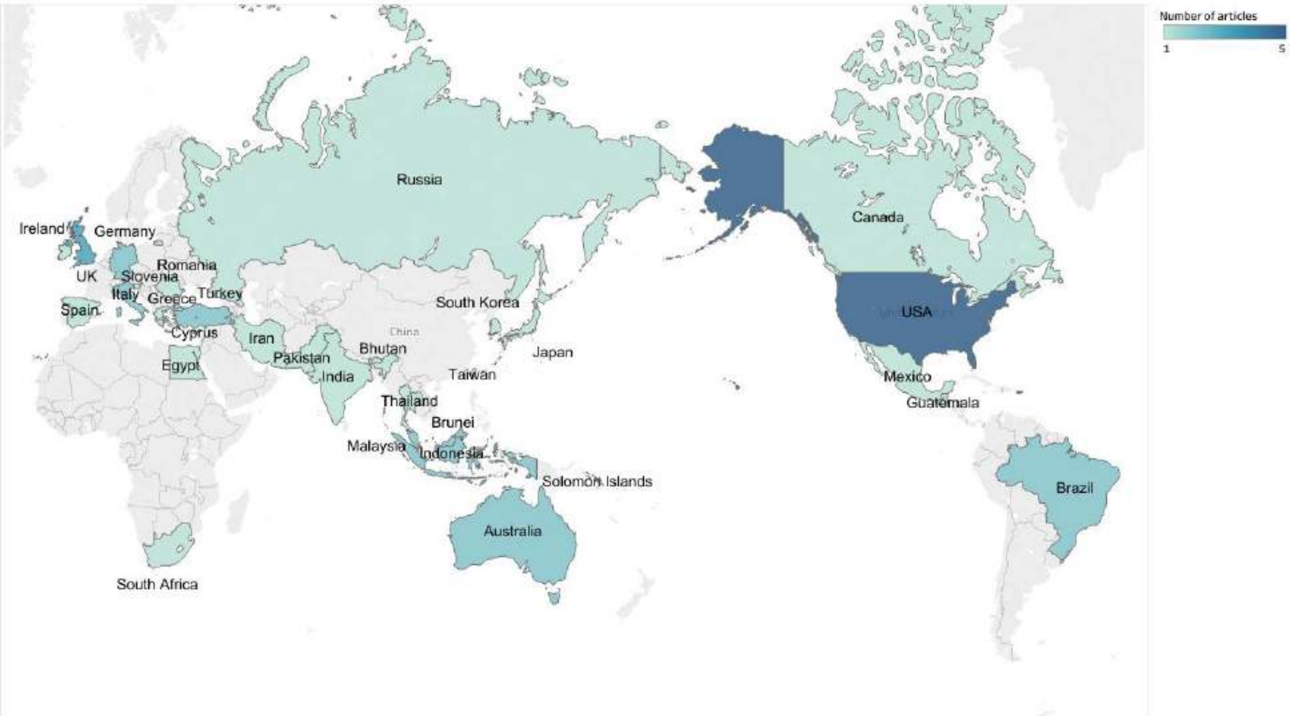


**FIGURE 3:** Geographical focus of the studies.

- Formal learning is usually related to an educational institution or an enterprise that offers formal education and training programs, guided by a formal curriculum; it leads to a recognized completion certificate or degree;
- Non-formal learning refers to organized instruction programs that occur in educational or labor settings, guided or not by a formal curriculum, but does not lead to a formal qualification;
- Informal learning, which is the result of daily work-related, family or leisure activities, occurs outside of organized education or training provision, hence, not leading to certification; it is often referred to as 'experience' or 'unintentional learning', happening throughout life without the learner necessarily being aware of the knowledge or skill gained.

Concerning the thematic focus, all articles were grouped according to the ten themes of ESC, as introduced by Thoresen (2010): Life quality; Lifestyles; Resources; Economics; Consumption and the environment; Consumer rights and responsibilities; Information management; Health and safety; Change management and Global awareness. Also, articles specific to waste education, which is part of ESC's Consumption and the environment theme, were further examined according to the waste type targeted.

An equal number of articles focused on formal and non-formal education – 15 articles each; six articles focused on the combination of non-formal and informal; three articles focused on the combination of formal and informal; two articles on the combination of formal and non-formal education, two articles on the combination of all types of education; and three articles did not specify the education type's focus.

The formal education articles spanned from primary to tertiary levels with one study inclusive of faculty members as target subjects. These articles covered several ESC topics such as sustainable tourism, product lifecycle, energy conservation and efficiency, consumer education, packaging, carbon footprint and food (Chalfoun, 2014; Crawford, Luke, and Van Pelt, 2015; Danilane and Marzano, 2014; S. Gough and Scott, 1999; Hawas and Al-Habaibeh, 2017; Nowak et al., 2009; Oliver, 2016; Pankina et al., 2016; Traversa et al., 2017). Specifically in the waste education field, two articles addressed food waste, while the remaining articles addressed household waste, municipal solid waste, and solid waste in general (Çelikler and Harman, 2015; Davis, 2008; Finlayson, Gregory, Ludtke, Meoli, and Ryan, 2017; Maddox, Doran, Williams, and Kus, 2011; Painter et al., 2016; Redman, 2013; Zain et al., 2013).

As for the articles on non-formal education, these addressed different societal segments, including the local community in household waste management and natural resource conservation (Hadiyanti, 2016; Indrianti, 2016; Patterson, Lindén, Edward, Wilhelmsson, and Löfgren, 2009); the public and private sectors in environmental bioethics and food consumption (Oto et al., 2012; Ribas et al., 2017; Ruini et al., 2016); health professionals dealing with alcohol consumption issues and healthcare waste management (Brumby et al., 2011; Kumar, Somrongthong, and Shaikh, 2015); household water end-users (Kossieris, Kozanis, et al., 2014; Kossieris, Panayiotakis, et al., 2014); consumers and retailers (Nagata, Azuma, Oda, Fujiwara, and Hanya, 2014); the general public on solid and municipal solid waste management (Kamaruddin, Pawson, and Kingham, 2013; Polanec et al., 2013); and, studies directed to youth from middle school age (Hadjichambis, Paraskeva-Hadjichambi, Ioannou, Georgiou, and Manoli, 2015), to university age (Yang, Chien, and Liu, 2012).

Concerning the combination of non-formal and informal education, the majority dealt with ESC topics in general, such as awareness raising for teens on sustainable manufactured products (Roeder, Scheibleger, and Stark, 2016), climate change issues at the community level (Roeder et al., 2016), consumer literacy for parents in South Korea (Lee, Jo, and Lim, 2015), and the discussion on the relevance of learning festivals for urban citizens, with Ireland as case study (Kearns, Lane, Neylon, and Osborne, 2013). On the other hand, articles on waste education specifically were targeting communities for household waste (Zarate, Slotnick, and Ramos, 2008), and solid waste management (Kanchanabhandhu and Woraphong, 2016).

The combination of formal and informal education and the formal and non-formal education produced three and two articles, respectively. The former combination included topics on household waste management for youth in the community (Redman, 2013), and waste electrical and electronic equipment management for children and parents (Pérez-Belis, Bovea, and Simó, 2015), in addition to, an article on the relevance of traditional knowledge inclusion in the Western environmental education (Chandra, 2014). For formal and non-formal education, articles took up the topic of mindfulness in ESC, with university students and employees as target subjects (Stanszus et al., 2017), as well as household waste management for elementary and middle school students (Mostowfi, Mamaghani, and Khorramar, 2016).

All forms of education combined were present in two articles, one on ESC for food and sustainable agriculture for children, parents and local the community (Schreinemachers et al., 2017), and one on waste management to evaluate a local education program (Vieira and Echeverria, 2007).

Lastly, in three of the articles reviewed, it was not explicitly stated the education type focus, with one focusing on information management and consumer rights and responsibilities, (Radulescu and Radulescu, 2011), another focusing on information management targeting Egyptian environmental designers (Wahba, 2012), and the other reporting on the status of ESC in Mexico (González-Gaudiano, 1990). Figures 4 to 6, present the main findings of the content analysis for the 46 articles.

## 3.3 The inclusion of Gender perspectives

As the result of the word search of the terms presented in the Materials and Methods section, the inclusion of gender perspectives could be seen in a total of 22 out of 46 articles, 12 corresponding to ESC and ten corresponding to waste education.

### *3.3.1 The landscape of the gender perspectives' inclusion*

The inclusion of gender in the set of 22 articles, can be observed in 2009, 2011-2014 with two articles per year, followed by a spike between 2015-2016, with four and seven articles respectively, and a retraction in 2017 with only two articles. As for the publications, the International Journal of Environmental and Science Education and Procedia – Social and Behavioural Sciences had five and four articles each, followed by two publications each from the Waste Management Journal and the Journal of Education and Practice. The remaining journals presented one article each.

Geographically, the study areas were diverse, with articles distributed almost evenly between the Global South and North (See Figure 8). Generally, the education programs described in 19 articles which an audience was identified were directed to four large target groups – Children (including parents); Community (including the general public and household users); Professionals (from the health sector and environmental designers); and Youth. Most of the subjects referred to the ESC themes, Consumption and the Environment, Lifestyles, Resources, and Information management.

### *3.3.2 Gender focus and Language*

In five of the articles, gender was generally mentioned to contextualize, describe and give examples of aspects that were relevant to the study the authors were reporting on (Chandra, 2014; Kamaruddin et al., 2013; Pérez-Belis et

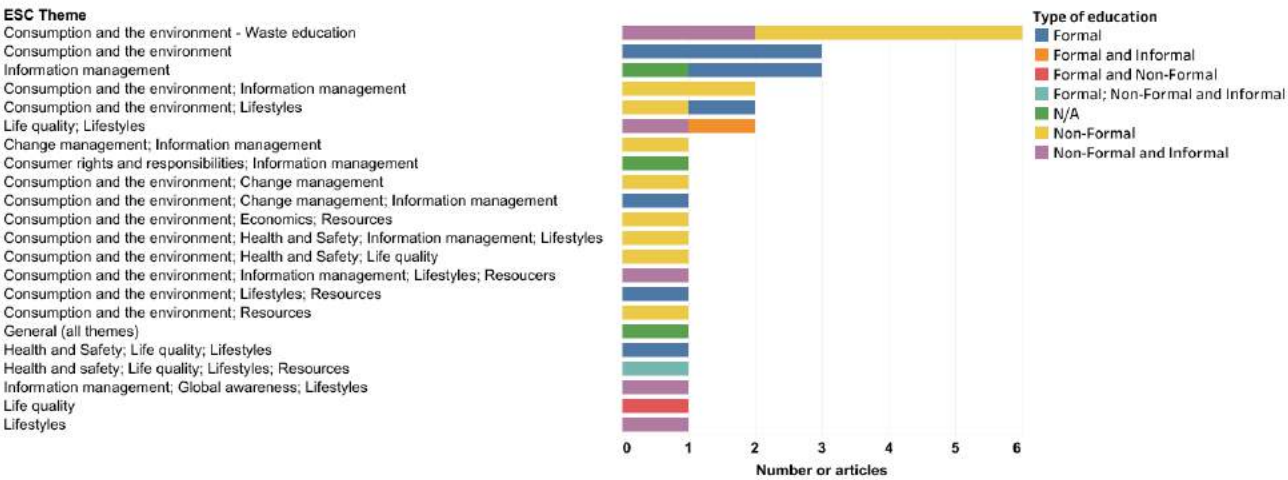


**FIGURE 4:** ESC themes and type of education.

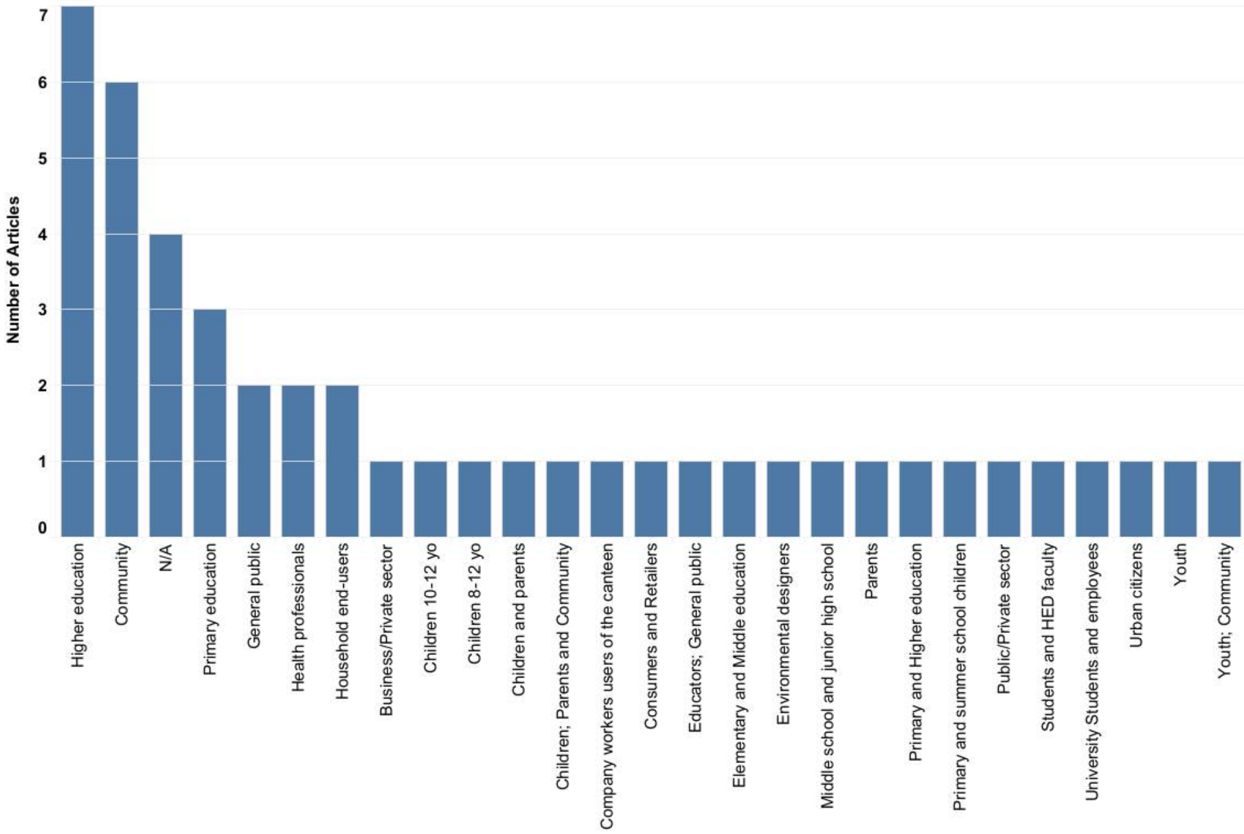


**FIGURE 5:** Target groups.

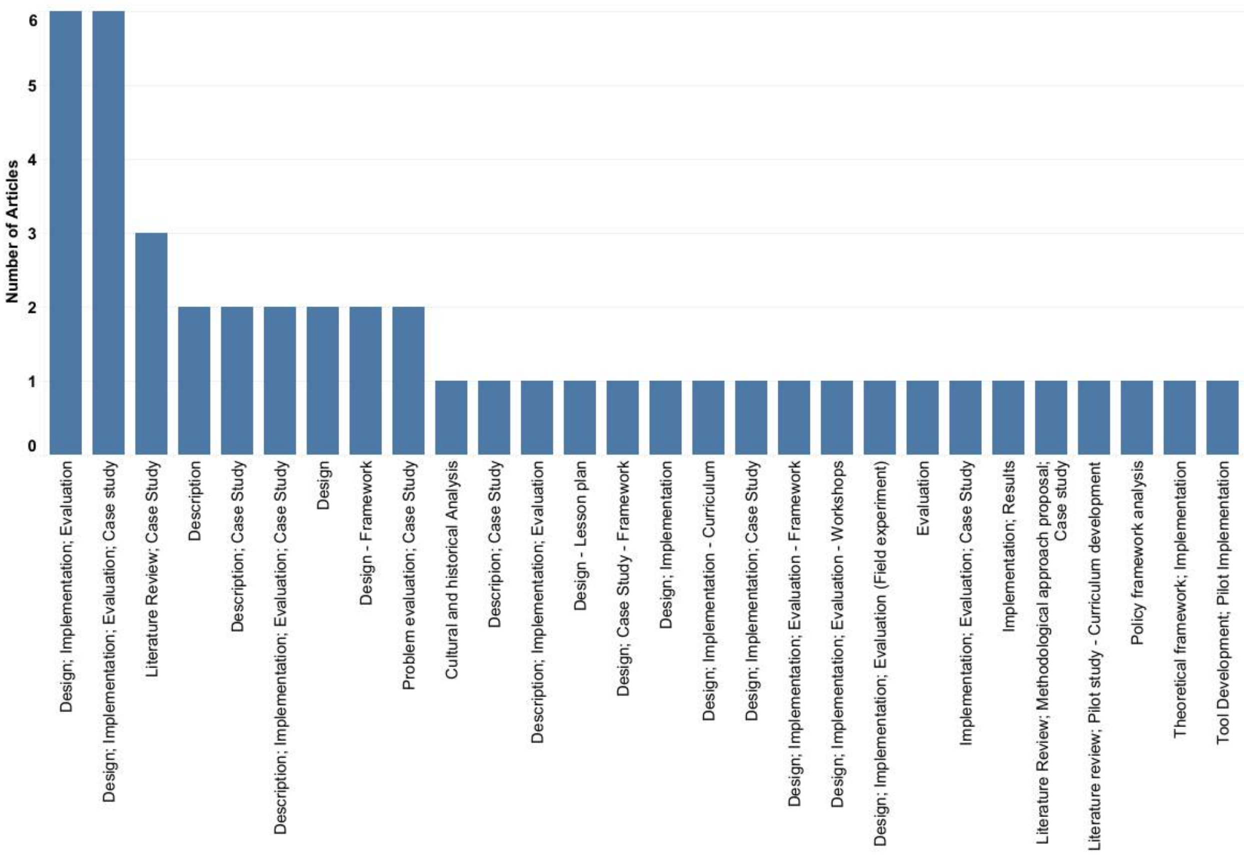


**FIGURE 6:** Research techniques.


**FIGURE 7:** Articles including gender perspectives: Target groups and ESC sub-themes.

al., 2015; Roeder et al., 2016; Wahba, 2012). In an additional article, the authors also mentioned the suitability of the education program they were proposing, to "both boys and girls" (Mostowfi et al., 2016). In another set of articles, gender was solely included to describe the composition of the studied groups, qualitatively and quantitatively. That was the case in Yang et al. (2012), mentioning the student participants, testing their proposed digital game system to aid in energy conservation education; for a gender differentiation of the study subjects as well as their parents' education level in Çelikler and Harman (2015); and, in Kossieris, Panayiotakis, et al. (2014), on "sex", being one of the several socio-demographic characteristics included in the semi-structured interviews conducted.

On the usage of language, four articles displayed ambiguity issues where the authors did not clarify if they were referring to one or both sexes. For instance, expressions such as "man-made," "man and the environment" and "known to man," were used (Hadjichambis et al., 2015; Nowak et al., 2009; Pankina et al., 2016; Radulescu and Radulescu, 2011). On the contrary, the distinction of (farm) men and women is seen throughout Brumby et al.'s (2011) work, to describe a training program for health professionals dealing with alcohol misuse in farm families. Precise language use, considering aspects related to gender neutrality versus specificity, is pertinent to avoid injustices and discrimination issues. Particularly in the field of education, in 1987, a call was made to UNESCO for the avoidance of gender-specific language, as a result of the growing awareness of the power of language. The agreed premise is that "If words and expressions that imply that women are inferior to men are constantly used, that assumption of inferiority tends to become part of our mindset" (UNESCO, 1999). For instance, it has been widely reported that research and practice in education (particularly in environmental education), has been centered in the male experience or has seen them as universal subjects without differentiating between male and female, even if "man" is not a universal and dominant agent on the environment (Gough, 2003, 2013; UNESCO, 1999).

### *3.3.3 Female and male stereotypes*

In one of the articles, the author briefly discussed that through comparison of the subjects' reactions, female participants exhibited a higher emotional response to the education program compared with their male counterpart. The author raised the possibility that this difference, was related to the notion that "women react more emotionally to environmental problems, which coupled with an internal locus of control (or enhanced effectiveness knowledge) will likely to lead to acting pro-environmentally" (Redman, 2013).The rhetoric described in the article, raises an issue that has been brought up by other researchers, which points to the fact that more concerned pro-environmental discourses and attitude towards green consumer behavior, are more gender-available to girls, than to boys (Autio, Heiskanen, and Heinonen, 2009; Blenkinsop, Piersol, and Sitka-Sage, 2017). Predominantly in Western culture, women have been assigned specific attributes related to nurturing skills and tolerance, and while these traits are likely to be developed as a result of the cultural assignment, it then creates a self-fulfilling prophesy cycle, in which the onus of upkeeping sustainability, and caring and protecting the environment is more significant for women (Autio et al., 2009; Jolley and Shields, 2009; March, Smyth, and Mukhopadhyay, 1999).

### *3.3.4 Gender influence and the role of women*

The distinctive role of gender was discussed in seven articles.

While Redman (2013) considered how social norms were likely to dictate the gender differences regarding the subjects of the study's willingness to adopt a lower meat diet, Pankina et al. (2016), included gender as an indicator for the consumer market of everyday products.

More extensively, Kanchanabhandhu and Woraphong (2016) and Kumar et al. (2015), included gender-specific analysis of the level of knowledge, awareness, and participation, before and after the education program's implementation, for a model of solid waste management and for a training model for healthcare waste, respectively. Apart from describing the design and implementation processes, the authors also relied on statistical tools to calculate the different indicators of knowledge, awareness and practice changes.

A similar emphasis on statistics to analyze the difference in the impact of the program for boys and girls was reported in a school gardening intervention for sustainable food consumption and agriculture practices in Bhutan (Schreinemachers et al., 2017). Additionally, an investigation of food waste reduction in a South African university campus, together with data collection, sampling, and analysis in a sex-disaggregated medium, was completed. As an outcome, the authors argued that focus on gender should be a central requirement in setting up the interventions to reduce food waste generation, as well as, the necessity for it to be the focus for future research (Painter et al., 2016).

Lastly, a single article fully addressed the significance of assessing gender aspects characterizing the education program's targets subjects and used those to inform the strategies and resource allocation, along with the design and implementation of the program. Patterson et al. (2009), reported on an adult environmental education program targeting fisherwomen as the result of a preliminary analysis. The analysis elucidated that in the target area: women's self-help groups were in place working on the community's sustainable use of financial resources; a low level of awareness on the importance of coral reefs was prevalent among fisher people, particularly the fisherwomen; and, women were in a better position to influence male fishers, and children in the household. The importance of conducting a needs assessment including the existing gender profile, before the program development, has been recognized as a success factor in the development of appropriate interventions, particularly in sustainability-related issues (Aguilar, 1999; European Institute for Gender Equality, 2016; March et al., 1999).

Additionally, the acknowledgment of women as particularly influential in shaping sustainable consumption and waste management initiatives was stated in a couple of articles. Hadiyanti (2016), mentioned the presence of homemakers and working women, part of a community group engaged in a recycling project; while Indrianti (2016) discussed the role of a women's group in initiating and maintaining a waste bank project (Figure 8).

## 4. CONCLUSIONS AND FUTURE RESEARCH OPPORTUNITIES

This study provides the first effort to comprehensively review the literature on Education for Sustainable Consumption (ESC), with a focus on Waste Education and the inclusion of gender perspectives, to assess the most up to date research in the field.

The application of the tool ProKnow-C granted the identification and bibliometric analysis of 46 academic articles, which were distributed globally and intermittently, from 1990 to 2017. The central portion was published in 2016, and the most frequent publisher of such articles being the Procedia – Social and Behavioural Sciences journal. Also, all forms of education practices – formal, non-formal, and informal, were described, targeting school children, youth, urban communities, public administrations, and the private sector, and several research methods were applied.

The limited number of articles retrieved indicates on the one hand that the topics of ESC and waste education are under-exploited in the scientific research community, and on the other hand, that the search criteria selected might have limited the number of resulting publications. To address the former, more studies need to be undertaken, particularly on ESC themes other than the dominant Con-

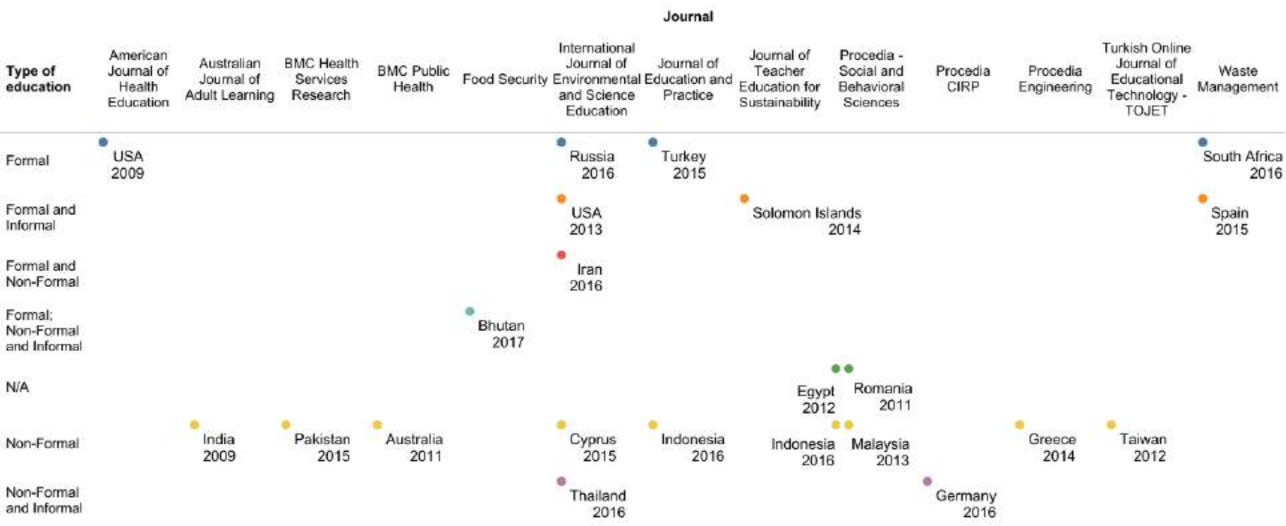


**FIGURE 8:** Summary of articles with gender perspectives' inclusion.

sumption and the environment and Lifestyles; the latter issue can be tackled by modifying the search criteria to include, for example, articles that are not fully accessible online (limiting the study to abstract review), and by including grey literature.

On the review of gender perspectives, 22 out of 46 articles presented some form of inclusion of gender aspects. However, for the most part, gender was superficially addressed. For example, gender was often only mentioned in passing or solely to present sex-disaggregated data to describe the studied groups. Other identified issues, worthy of further research include the usage of gender-blind language and the inclusion of discourse that can potentially contribute to reinforcing gender stereotypes.

While there were cases of recognition and comprehensive consideration of gender into the research, these were limited, ultimately reflecting a need in the field of environmental education's research and practice, particularly in ESC and waste education, to expand on the topic and present ways towards the inclusion of gender perspectives. Accordingly, a follow-up study addressing this research and practice gap will explore the links between gender, waste, and education, and propose a framework for gender mainstreaming in waste education programs . The goal being, to harness the potential of gender inclusion for the effectiveness of the education programs, and for its contribution to gender equality.

## ACKNOWLEDGMENTS

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# USE OF GEOGRAPHICAL INFORMATION SYSTEM FOR THE EVALUATION OF SOLID WASTE MANAGEMENT PRACTICE IN KHULNA CITY

Smita Golder and Muhammed Alamgir *

*Department of Civil Engineering, Khulna University of Engineering and Technology, Khulna-9203, Bangladesh*

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**ABSTRACT**

Generation and characteristics of municipal solid waste (MSW) in Khulna city has been analyzed along with the associated environmental impacts and existing MSW practices with the help of geographical information system (GIS). The status of the existing municipal solid waste management (MSWM) such as generation, collection, on-site storage, transportation and open dumping has been identified. The daily generation of MSW is estimated about 520 tonnes of which food and vegetable wastes are the main components (79% on average). GIS is used to locate and to analysis the existing waste collecting bins and containers. This study shows that GIS can be treated as a decision support tool for ensuring and practicing efficient collection system of MSW.

## 1. INTRODUCTION

Urban solid waste management is considered as one of the most instantaneous and momentous environmental problems, faced by developing Asian Countries. Municipal Solid Waste Management (MSWM) is one kind of responsibility combining various activities such as collection, transportation, disposal, processing and treatment of solid waste. As the world is approaching to the urban future and economic development, managing solid waste crisis is undoubtedly one of the key challenges of the 21$^{th}$ century. The increasing volume of waste along with the rapid economic headway and globalization is creating a serious difficulty to ecosystems and human health. The problem of municipal solid waste management has acquired an alarming dimension in the developing countries during the last few decades. Compared to high income countries, the urban residents of developing countries produce less per capita solid waste, but the capacity of developing countries to collect, process disposal or reuse it in a cost effective way is limitted (Visvanathan and Trankler, 2004).

High-tech evolution, the rapid economic headway, globalization, overpopulation, affluence, lack of proper garbage collection have accelerated the dynamics of the urbanization process in advancing countries. Waste management questions are approaching to the forefront of the global habitat schedule at an enlarging amount. Municipal solid waste consists of domestic waste generated by urban residents (households) with addition of commercial wastes but typically excludes industrial hazardous waste and domestic sewage sludge (James, 1997). The waste generated by households in urban areas constitutes a significant component of the municipal solid waste generated and therefore it has a direct bearing on the design of municipal waste management systems (Benitez, 2008). Compared to high income countries, the urban residents of developing countries produce less per-capita solid waste, but the capacity of the developing countries to collect, process, dispose or reuse the solid waste in a cost-effective way is limited (Visvanathan and Trankler, 2004).

Khulna, the third largest metropolitan city of Bangladesh, generates a massive quantity of waste every day from different sources. Khulna city normally generates 500 tonnes per day of solid waste every day. Of the total, only 270 TO 300 tonnes are collected and dumped into the open dumping ground at Rajbando landfills of Batiaghata upazila. The rest that lies uncollected on Khulna city streets every day due to poor monitoring and logistic supports and negligence of the field-level corporation workers and officials, exposes the residents to greater health risks and environmental hazards. The remaining uncollected wastes are dumped into drains, open spaces, road-sides and water-bodies, not only blocking the city's drainage links but also creating an unhealthy and stinking environment. Due to dumping into open areas, airborne and waterborne

* Corresponding author:
Muhammed Alamgir
email: alamgir63dr@yahoo.com

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diseases such as like diarrhea, dysentery, jaundice and skin diseases can be spread out and the fitness condition of people may be at risk.

Recently, there has been an increase in research that uses Geographic Information System (GIS) application as a tool for MSW management estimation and planning. MSW management practices require collection of decisive information which is for taking corrective measures as well as for proper planning to ensure sustainability (Ramachandra and Saira, 2003). GIS is a computer system capable of holding and using data describing places on the earth's surface. GIS is a good decision support tool for SWM planning. To analysis the present SWM practice and to identify the served and un-served area from the coverage GIS map in KCC are the specific objectives of the study.

## 2. LITERATURE REVIEW

Literature on solid waste management is extensive in scope and comparison for both developed countries as well as for developing countries. To solve the inherent problems regarding solid waste management using GIS, particularly for developing countries, several specific researches have been done.

According to (Kathiravale, Muhd Yunus, 2008), it is reported that developed countries generally generate more waste than developing countries. The generation of waste varies considerably between countries based on the culture, public awareness and management (Hazra and Goel, 2009; Wagner and Arnold, 2008). According to Sujauddin et al. (2008) the generation of waste is influenced by family size, their education level and the monthly income. Gender, peer influence, land size, location of household and membership of environmental organization explain household waste utilization and separation behavior (Ekere, 2009). It has been reported that collection, transfer and transport practices are affected by improper bin collection systems, poor route planning, lack of information about collection schedule (Hazra and Goel, 2009), insufficient infrastructure (Moghadam et al., 2009), poor roads and number of vehicles for waste collection (Henry, 2006). Tadesse et al. (2008) analyzed the factors that influence household waste disposal decision making. Results showed that the supply of waste facilities significantly affects waste disposal choice. Inadequate supply of waste containers and longer distance to these containers increase the probability of waste dumping in open areas and roadsides relative to the use of communal containers. Insufficient financial resources limiting the safe disposal of waste in well equipped and engineered landfills and absence of legislation are mentioned by (Pokhrel and Viraraghavan, 2005).

During the last few decades, the problems associated with municipal solid waste (MSW) management have acquired an alarming dimension in Bangladesh. High population growth rate and increase of economic activities in the urban areas of developing countries combined with the lack of training in modern solid waste management practices complicate the efforts to improve the solid waste management services (Ahsan et al., 2014). In Bangladesh, the solid waste management has so far been ignored and is one of the least studied environmental issues. Recently the concerned stakeholders have begun to consider this sector to be an essential component to protect human health and nature. The urban population in Bangladesh has increased at a very steep rate of about 6% per year and concentrated mostly in six major cities, namely, Dhaka, Chittagong, Khulna, Rajshahi, Barisal, and Sylhet. Current estimations showed that about 13% of total population and 55 to 60% of total urban population are living in these cities (Alamgir, 2005).

Khulna, the third largest metropolitan city of Bangladesh, is in the southern part of the country with its location on the axis of Jessore-Mongla port, the second largest seaport of the country. The study area Khulna City Corporation area 45.65 square km, located in between 24°45' and 24°54' north latitudes and in between 89°28' and 89°35' east longitudes. Khulna City Corporation area (KCC) has a population of about 1.4 million with 31 Wards. It's elevation is 7 feet above the MSL. Acording to Khulna City Corporation (KCC), Annual Report of Khulna, 2016, Khulna city generates approximately 520 ton/day of solid waste but waste carried by KCC to the final disposal is about 300 ton/day. So about 220 ton wastes are dumped illegally per day in Khulna city here and there. The current location of landfill is about 7-8 km away from the KCC area. (Rafizul, Alamgir, Howlader, Kraft and Haedrich, 2009), reported that the city authority generally manages the MSW however; recently, some NGOs, CBOs and private organization are working for door-to-door collection with city authority's initiatives. Door-to-door collection systems were introduced recently for MSW collection from generation sources, mainly from households, and major portion of wastes are disposed to the nearest secondary disposal site (Alamgir, Ahsan, McDonald, Upreti and Rafizul, 2005). (Haque, 2005), reported that the location of disposal (secondary) sites of KCC represents the unconsciousness about the environmental and public health hazards which arises from disposing of waste in improper location. A suitable site must have environmental safety criteria's. Criteria for site selection includes natural physical characteristics as well as socioeconomic, ecological, and land use factors (Debasish Adhikary and Shahidul Islam, 2015). (Riyad, Zohur-Uz-Zaman and Farid Hossain, 2015), has also described the present scenario of inconvenient bin placements in Khulna city. It is reported that KCC is responsible for collecting and removing the deposited solid waste in dustbins/containers surrounded by the whole city area and finally transporting it at the ultimate disposal sites. Residents are responsible for bringing their generated solid waste to KCC's primary collection points where dustbins/containers are situated. (Islam Rafizul, Risvi Kizer and Ashiqur Rahman, 2013), have proposed a GIS based optimized route plan for collection and disposal of municipal solid waste (MSW) from SDS to ultimate disposal site (UDS) in Khulna city. Many suggestions were considered in this spatial planning proposal while working in GIS, they are (i) identification of exact location of MSW bins with GPS demarcating on the base map; (ii) maintaining a record of SDSs; (iii) a map showing the road network in different areas; (iv) a map showing the distances between the bins; (v) location of the SDSs; (vi) record of available vehicles and equipment for MSW management; (vii) allocating a unique number to all the SDSs so it can be easily and quickly located in case of any complaint registered or planning and maintenance; (viii) maintaining a record about the type of SDSs and (ix) record of the responsibilities and assignment of work, equipments, vehicles etc. of the MSW maintenance and also the logistics information about the the transportation involved in the system. Figure 1 shows the map of Khulna City Corporation area.

GIS is a computer support tool used for capturing, storing, querying, analyzing and displaying spatial data from real world for a particular set of purposes. This technique is used to locate and analyse the existing waste collecting bins and containers and to generate optimal route for collecting solid wastes. GIS is a tool that not only reduces time and cost of site selection, but also provides a digital data bank for future monitoring program of site. (Shohel, et al., 2013), also showcased application of GIS in solid waste management for Khulna city. In his study, the criteria for suitable waste bin location includes natural physical characteristics and mental characteristics for user as well as socioeconomic, ecological, land use factors, user travel time, concern about public institute, easily access and publicly concern about environment road width.

### 2.1 Solid waste management in KCC

KCC is the formal public sector organization of the government that is responsible for SWM in KCC area. The SWMS by the KCC is shown in Figure 2.

Map of the Khulna city corporation area

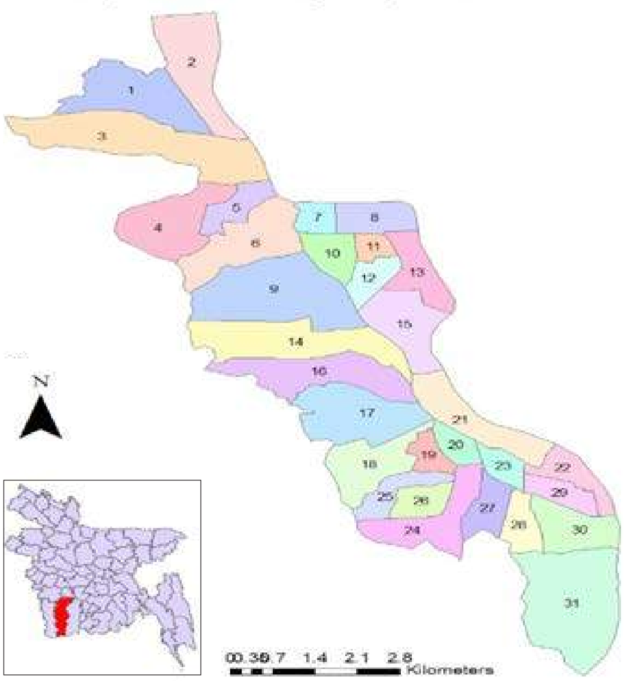


**FIGURE 1:** Khulna City Corporation.

## 3. METHODOLOGY

### 3.1 Investigation of primary and secondary disposal site

The questions of field survey for gathering the required information about the existing situation of primary point and SDS located in KCC and answers from city dwellers and other stakeholders were recorded. A straight espial and audit have been prosecuted among the native people and related organizations with a questionnaire survey. Study on the local people about their perception and feedback to SWM system organized by KCC and the way other people respond to it, is observed also for perceiving the management behaviour of solid waste. The detailed map had the information about road network, major building, religious buildings, cinema halls, land mark, and water streams. Information about waste bins location as well as open dumps were collected by general survey and entered in the GIS database.

### 3.2 Model analysis

Model is developed on GIS for selecting suitable location of waste bin in study area. Developing model has been used some considerations, there are:

Firstly, all contexts are included to the study area like as population, density, household waste generation, socio-economic condition and existing solid waste management consideration of study area.

Secondly, equal waste bin size and capacity like as concrete bin capacity is 1000 Kg and haul container capacity is 2500 Kg.

Thirdly, per capita waste generation rate at 0.5 kg/day are considered.

Fourthly, inhabitants are open choice to disposal waste service household to waste bin.

Methodology followed is shown in the Figure 3.

## 4. RESULTS AND DISCUSSION

### 4.1 Current Situation of SWM from Questionnaire Survey

The recent survey shows that about 500 tonnes wastes are produced daily in Khulna city where of the total, only 250 to 270 tons are collected and dumped into the open dumping ground at Rajbando landfills of Batiaghata upazila. The rest does not even reach to the dumping ground owing lack of logistic support and manpower. The remaining about 180 tons of uncollected wastes being dumped into drains, open spaces, road-sides and water-bodies, not only block the city's drainage links but also create an unhealthy and stinking environment. In absence of dustbins, residents of Khulna city force to dump wastes into drains, canals and road-sides. In the Khulna city, the main producers of solid waste are residences, whole and retail sale market places including shopping places, streets, hotels and restaurants, hospitals and private clinics, educational institutions, cinemas, bus, railway and launch/steamer ghats, slaughter houses etc. In the collection system, normally NGOs and CBOs conduct the door to door collection. Now a day's city corporation authority also conducts door to door collection system in a small area.

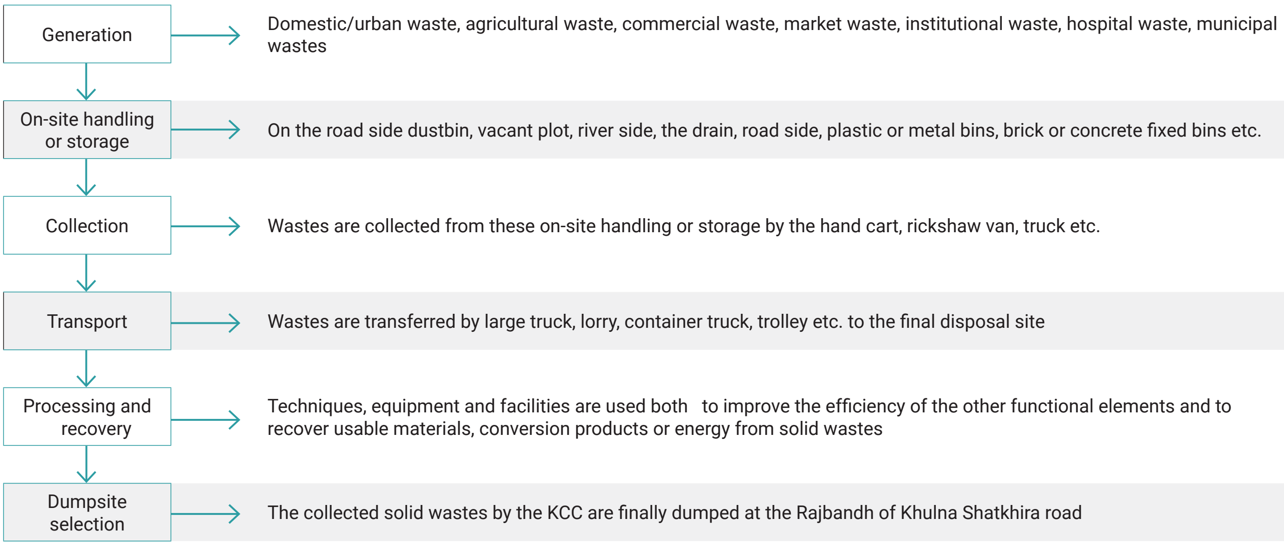


**FIGURE 2:** Solid waste management system in KCC.

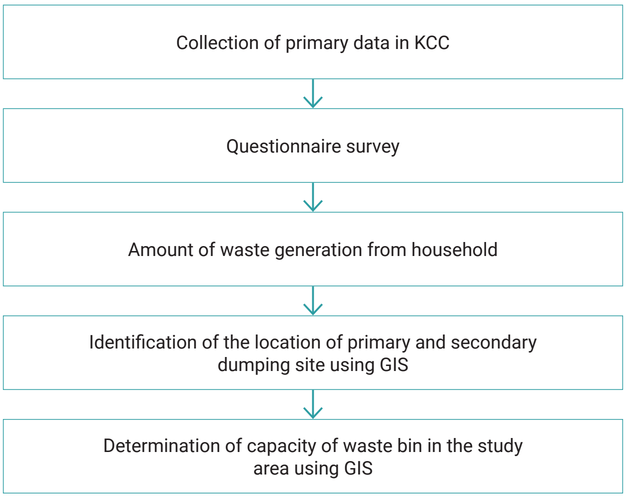


**FIGURE 3:** Flow chart of methodology.

To investigate the present scenario of MSW management in Khulna city, questionnaire survey among 200 responsible people has been conducted. It was noticed that, KCC collects around 55%, non-government organization and community based organization (NGO and CBO) collects 10% of MSW from the household and the unmanaged waste remains 35% here and there into drains, canals and road-sides of Khulna city (Figure 4).

KCC is responsible for collection and removal of deposited solid waste in dustbins/containers surrounded the whole city area and finally transport it at the ultimate disposal sites. Residents are responsible for bringing their generated solid waste to KCC's primary collection points where dustbins/containers are situated. Recently, NGO's have introduced door-to-door collection of solid waste in different area of Khulna City Corporation for the management of solid waste in cooperation with the city authority and respective ward Commissioner. From the questionnaire survey, it is reported that door-to-door collection system is most preferable waste collection system as household themselves especially women do not have any permanent disposal of waste near their location. People are sometimes unconscious to throw the waste to the streets rather than using dustbin. For better review, a bar diagram (Figure 5) is provided here.

An analysis of the household participation is shown in Figure 6. On the average 70% of the households in the study area are participating in the collection system. Out of the remaining non-participating 30%, the majority, i.e., 21% lives in areas that are of rural characteristics. The households of this rural criteria discard their waste in an open excavation within their own premises. The actual non-participation is 9% of which 6% are still not motivated, another 3% do not pay. The survey was also carried out about the collection prototype of MSW from SDS among 200 responsible persons and stakeholders, it was noticed that KCC collects MSW from SDS 168 (84%) once per day, 28 (14%) twice per day and 4 (2%) more and shown in Figure 7.

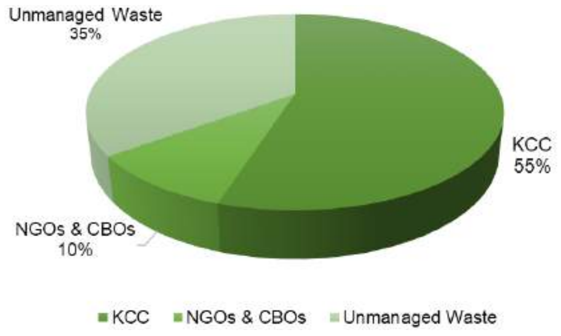


**FIGURE 4:** Waste management capacity.

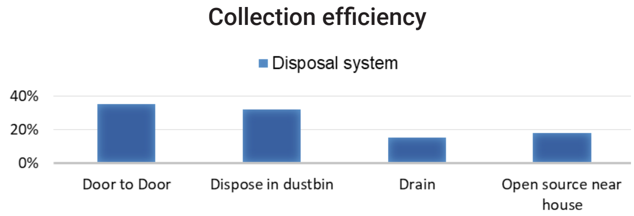


**FIGURE 5:** Distribution of waste disposal.

Figure 8 shows source storage aspects of MSW as revealed from people comments. The survey results reveal that (Figure 8a) that 62% do the source storage of the generated wastes in a dustbin, 18% polythene bag, 15% in open space. Most of the people practice mixed storage at source (Figure 8b). People are not happy (Figure 9) with present performance of city authority to manage the MSW problems. Figure 10 shows that most of the people prefer daytime, 10:00am to 2:00pm for the disposal of wastes from source, while a considerable portion (18%) do not have any choice of fixed time. It is clear that the present SWM in Khulna city is not well sufficient and should be improved immediately. So, an initiative for integrated urban SWM is essential to minimize waste generation with supporting reuse and recycling options in Khulna city.

### 4.2 Analysis of service coverage of existing waste bin and container

At present, there are approximately 180 waste bins and 35 secondary disposal sites (SDS) located in KCC area. This study is based on collected field data, questionnaire survey and literature review to identify the present status of solid waste management in Khulna City Corporation. From field survey of the study area, it was found that the preferable distance of primary waste bin for the households ranges approximately 500 m, which is not covering the full area. The capacity of each waste bins ranges between 500 kg to 1000 kg and from estimating the study area, per capita waste generation rate is 0.5 kg per day. Among 35 SDS points, there are 14 open and 21 haul containers. Model is developed on GIS for identification and selection of waste bin in the study area. GIS as a tool which is used in the analysis of the existing situation of waste bin management and then the selection of some suitable locations and also in the requirement of number of the waste bins. The existing waste bins are located in GIS model by pointing the exact location of waste bins in geo-image of Khulna City Corporation by Google map. This situation assisted to improve the service efficiency. Determinations of required number of waste containers were done based on the population of each ward and the capacity of each container. The optimum location of waste bins and containers had been suggested with present containers and bins. Self-judgment was applied to choose the essential locations. KCC solid waste management facility is not enough to cover the study area fully on time. There are 21 different size KCC containers in the study area retaining about 65 tonnes capacity. There are 14 open SDS in Khulna city in where people and KCC workers dispose their waste and it's capacity is 28 tonnes. The KCC workers collect wastes at day-time from residential areas but the KCC trucks collected the wastes at night or at early in the morning. While city authority has some limited numbers of non-motorized Rickshaw vans and Hand trolley, those are mainly used for the collection of MSW from Community bins located at roadside, home side, near market, and transfer to SDS. Besides this drain sludge's are also collected by these vans. The existing waste bin and containers can hold about 54.6% generated waste of the study area. Rests of the waste (45.4%) are dumped illegally here and there. Table 1 shows the total waste collection from the waste bins and containers in Khulna city.

Following, Figure 11 shows location of primary points

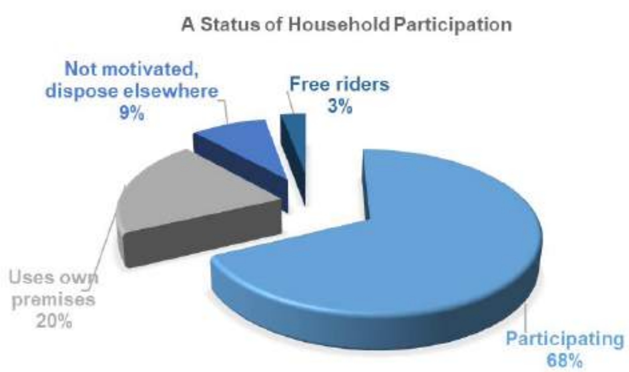


**FIGURE 6:** Status of household participation.


**FIGURE 7:** Prototype for MSW from SDS.

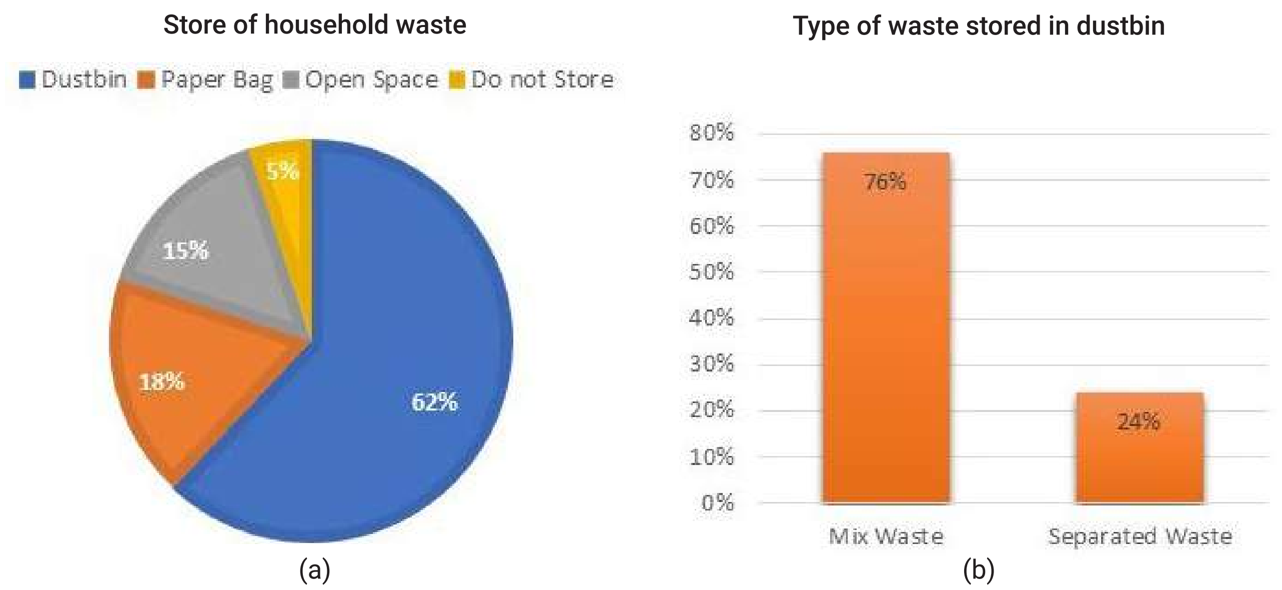


**FIGURE 8:** Source storage aspects of MSW as revealed from people comments.

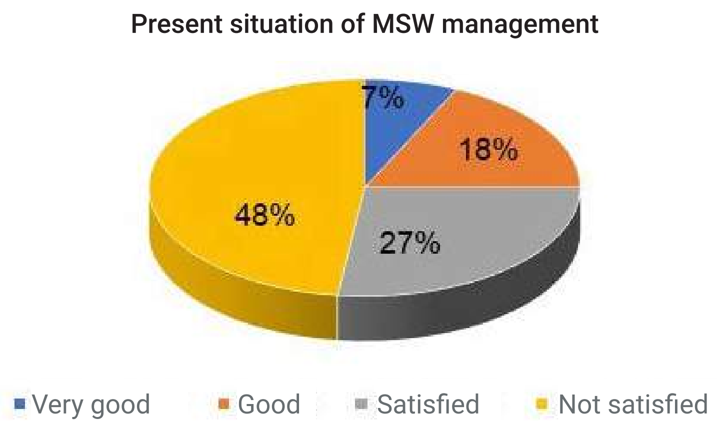


**FIGURE 9:** Opinions on the performance of city authority about MSW management.

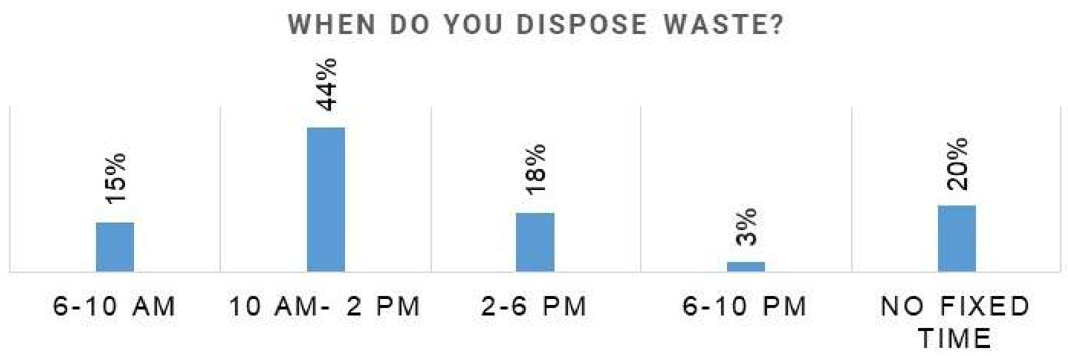


**FIGURE 10:** Disposal of waste

in KCC. At present, there are 180 bins in KCC. Figure 12 shows the service area of each dustbin. Here the area of influence or service area of a single dustbin considers 500 meter radius (highest) form its center. The capacity of each waste bins ranges between 500kg to 100 kg and generation rate is considered 0.5 kg per capita per day. People in ward no. 7, 10, 11, 12, 19, 20, 25, 26 have the full services with the existing dustbins. At ward-3 and ward-31, there are no such bins or containers for solid waste disposal. Population density of that site in KCC area is lower than the other parts in KCC area. Community based SW collection vans collect wastes once or twice in a week from that site. Ward no. 7, 10, 11, 12, 19, 20, 25, 26 of the study area are fully residential area and waste collection practice is per-

**TABLE 1:** Capacity of existing waste bins and containers in Khulna city.

| Types | Capacity (tonnes) | Total Number | Total Waste Collection (tonnes) |
|---|---|---|---|
| KCC containers | 2.5 | 16 | 40 |
| | 5.0 | 5 | 25 |
| Open SDS | 2.0 | 14 | 28 |
| Waste Bins (brick and concrete) | 1 | 180 | 180 |
| | | | **Total = 273 tonne (54.60%)** |

formed regularly by KCC authority. With some exceptions of illegal dumping, ward 2, 4, 14, 16, 22 and 23 have quite good waste management practices. In some part of the ward-1 and ward-9, mainly in slum areas, unprivileged peoples had a lack of proper sanitation and solid waste dumping facilities and only wastes are collected at night from the adjacent containers. So, these sites need proper SWM observation.

Table 2 shows service category dustbin in KCC, which has three category: only served area, over served are and un-served area. Only served area means the area which is properly served by one KCC containers. Overserved area means more than 2 dustbin facility. And the area which is not served by any dustbin facilities is known as unserved area. As for example, ward no 01 has three category but ward no 10 has two category- only served area and over served area. Table 3 and Table 4 shows the estimation only served, over served and un-served area and population. Figure 13 shows distribution of overserved dustbin service in KCC. Figure 14 represents the SDS location and service area in KCC. Though KCC managed solid wastes on ward basis, it is quite impossible to reach 100% efficiency in Municipal Solid Waste Management (MSWM). But adequate SWM can improve the SWM efficiency to a substantial extent. Distribution of waste disposal, prototype collection for MSW from SDS, household participation, initiated by local community and supported by KCC was observed. But in some areas under KCC, it is noticed that the SWM practice was not sustained thoroughly which results medley waste disposal and causes environmental pollution.

From the questionnaire survey, it is found that solid wastes are generated more from March to August month of the year. In rainy season, the scenario was quite worse with the flooding the wastes to the roads with rain water and block the seepage part and drainage systems for flowing the dirty water. Comparatively KCC waste collection authority helps to minimize the SW problem, but the management of waste collection from household by the local SW collecting bodies are responsible for the illegal waste disposal. As the residential area was highly over-populat-

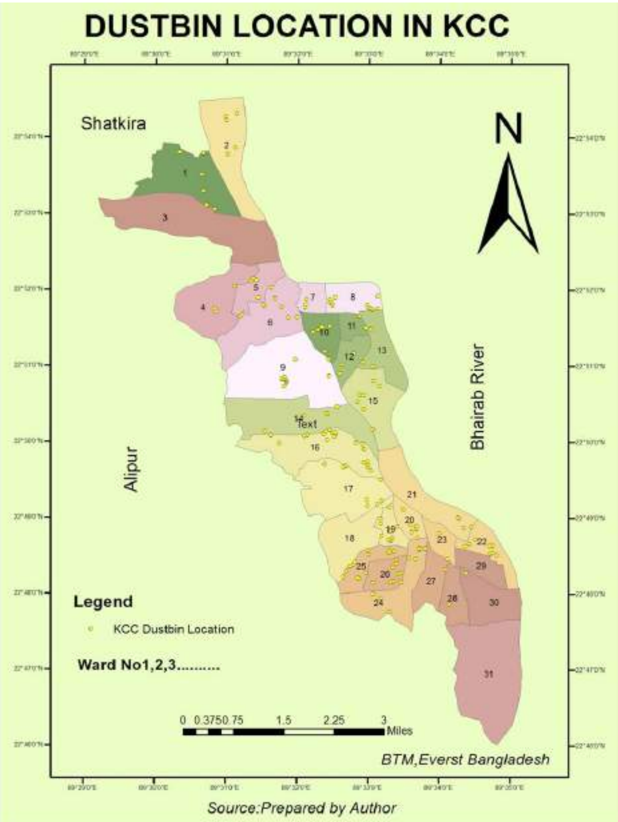


**FIGURE 11:** Location of primary points in KCC.

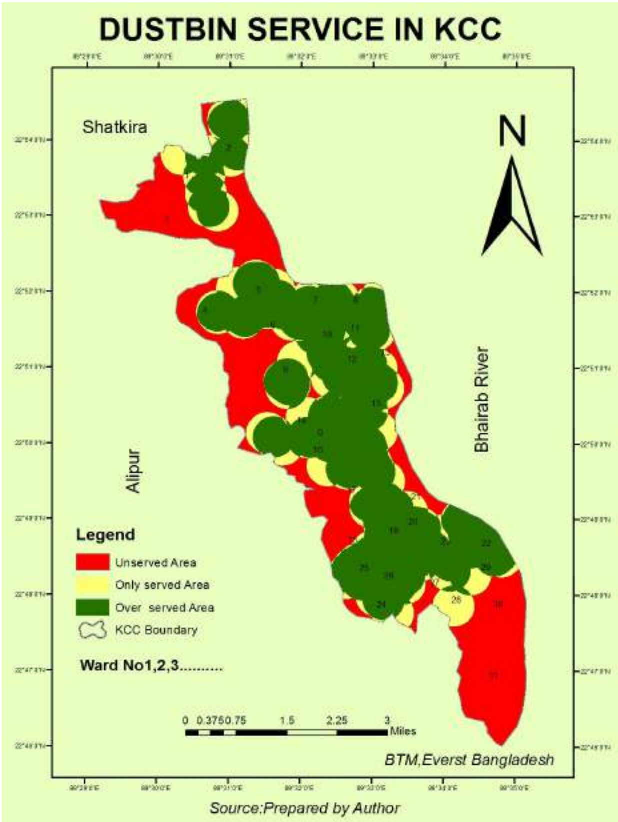


**FIGURE 12:** Dustbin Service in KCC.

**TABLE 2:** Service category dustbin in KCC.

| Ward No | Only served Area (No) | Over Served area (No) | Unserved Area(no) | Grand Total |
|---|---|---|---|---|
| 1 | 1 | 1 | 1 | 3 |
| 10 | 1 | 1 | | 2 |
| 11 | 1 | 1 | | 2 |
| 12 | 1 | 1 | | 2 |
| 13 | 1 | 1 | 1 | 3 |
| 14 | 1 | 1 | 1 | 3 |
| 15 | 1 | 1 | 1 | 3 |
| 16 | 1 | 1 | 1 | 3 |
| 17 | 1 | 1 | 1 | 3 |
| 18 | 1 | 1 | 1 | 3 |
| 19 | | 1 | | 1 |
| 2 | 1 | 1 | 1 | 3 |
| 20 | | 1 | | 1 |
| 21 | 1 | 1 | 1 | 3 |
| 22 | 1 | 1 | 1 | 3 |
| 23 | 1 | 1 | | 2 |
| 24 | 1 | 1 | 1 | 3 |
| 25 | 1 | 1 | | 2 |
| 26 | | 1 | | 1 |
| 27 | 1 | 1 | 1 | 3 |
| 28 | 1 | 1 | 1 | 3 |
| 29 | 1 | 1 | 1 | 3 |
| 3 | 1 | 1 | 1 | 3 |
| 30 | 1 | 1 | 1 | 3 |
| 31 | 1 | | 1 | 2 |
| 4 | 1 | 1 | 1 | 3 |
| 5 | 1 | 1 | 1 | 3 |
| 6 | 1 | 1 | 1 | 3 |
| 7 | 1 | 1 | | 2 |
| 8 | 1 | 1 | 1 | 3 |
| 9 | 1 | 1 | 1 | 3 |
| **Grand Total** | **28** | **30** | **22** | **80** |

**TABLE 3:** Served, over server and unserved area (Sq. km).

| Ward No | Only served Area | Over Served area | Unserved Area | Grand Total |
|---|---|---|---|---|
| 1 | 2.227856 | 2.227856 | 2.227856 | 6.683568 |
| 10 | 0.809781 | 0.809781 | | 1.619562 |
| 11 | 0.364905 | 0.364905 | | 0.72981 |
| 12 | 0.659046 | 0.659046 | | 1.318092 |
| 13 | 1.119565 | 1.119565 | 1.119565 | 3.358695 |
| 14 | 2.691663 | 2.691663 | 2.691663 | 8.074989 |
| 15 | 1.659229 | 1.659229 | 1.659229 | 4.977687 |
| 16 | 2.253474 | 2.253474 | 2.253474 | 6.760422 |
| 17 | 2.298736 | 2.298736 | 2.298736 | 6.896208 |
| 18 | 1.617838 | 1.617838 | 1.617838 | 4.853514 |
| 19 | | 0.492191 | | 0.492191 |
| 2 | 2.179358 | 2.179358 | 2.179358 | 6.538074 |
| 20 | | 0.499665 | | 0.499665 |
| 21 | 1.725098 | 1.725098 | 1.725098 | 5.175294 |
| 22 | 0.825691 | 0.825691 | 0.825691 | 2.477073 |

| Ward No | Only served Area | Over Served area | Unserved Area | Grand Total |
|---|---|---|---|---|
| 23 | 0.510105 | 0.510105 | | 1.02021 |
| 24 | 1.678222 | 1.678222 | 1.678222 | 5.034666 |
| 25 | 0.762184 | 0.762184 | | 1.524368 |
| 26 | | 0.66492 | | 0.66492 |
| 27 | 0.811904 | 0.811904 | 0.811904 | 2.435712 |
| 28 | 0.735918 | 0.735918 | 0.735918 | 2.207754 |
| 29 | 0.659339 | 0.659339 | 0.659339 | 1.978017 |
| 3 | 3.657733 | 3.657733 | 3.657733 | 10.973199 |
| 30 | 1.320933 | 1.320933 | 1.320933 | 3.962799 |
| 31 | 3.902639 | | 3.902639 | 7.805278 |
| 4 | 2.034172 | 2.034172 | 2.034172 | 6.102516 |
| 5 | 0.775361 | 0.775361 | 0.775361 | 2.326083 |
| 6 | 2.159462 | 2.159462 | 2.159462 | 6.478386 |
| 7 | 0.471743 | 0.471743 | | 0.943486 |
| 8 | 0.943937 | 0.943937 | 0.943937 | 2.831811 |
| 9 | 3.540127 | 3.540127 | 3.540127 | 10.620381 |
| **Grand Total** | **44.396019** | **42.150156** | **40.818255** | **127.36443** |

**TABLE 4:** Served, over served and unserved population (as 2020, BBS).

| Ward No | Only served Pop. | Over Served Pop. | Unserved Pop | Grand Total |
|---|---|---|---|---|
| 1 | 31370 | 31370 | 31370 | 94110 |
| 10 | 28600 | 28600 | | 57200 |
| 11 | 29960 | 29960 | | 59920 |
| 12 | 80370 | 80370 | | 160740 |
| 13 | 30830 | 30830 | 30830 | 92490 |
| 14 | 40840 | 40840 | 40840 | 122520 |
| 15 | 39730 | 39730 | 39730 | 119190 |
| 16 | 55420 | 55420 | 55420 | 166260 |
| 17 | 46880 | 46880 | 46880 | 140640 |
| 18 | 25890 | 25890 | 25890 | 77670 |
| 19 | | 40650 | | 40650 |
| 2 | 29060 | 29060 | 29060 | 87180 |
| 20 | | 34810 | | 34810 |
| 21 | 38590 | 38590 | 38590 | 115770 |
| 22 | 33410 | 33410 | 33410 | 100230 |
| 23 | 28310 | 28310 | | 56620 |
| 24 | 66350 | 66350 | 66350 | 199050 |
| 25 | 41860 | 41860 | | 83720 |
| 26 | | 27930 | | 27930 |
| 27 | 48630 | 48630 | 48630 | 145890 |
| 28 | 34600 | 34600 | 34600 | 103800 |
| 29 | 31550 | 31550 | 31550 | 94650 |
| 3 | 35550 | 35550 | 35550 | 106650 |
| 30 | 55330 | 55330 | 55330 | 165990 |
| 31 | 50350 | | 50350 | 100700 |
| 4 | 22080 | 22080 | 22080 | 66240 |
| 5 | 23650 | 23650 | 23650 | 70950 |
| 6 | 32430 | 32430 | 32430 | 97290 |
| 7 | 22870 | 22870 | | 45740 |
| 8 | 28640 | 28640 | 28640 | 85920 |
| 9 | 53460 | 53460 | 53460 | 160380 |
| **Grand Total** | **1086610** | **1139650** | **854640** | **3080900** |

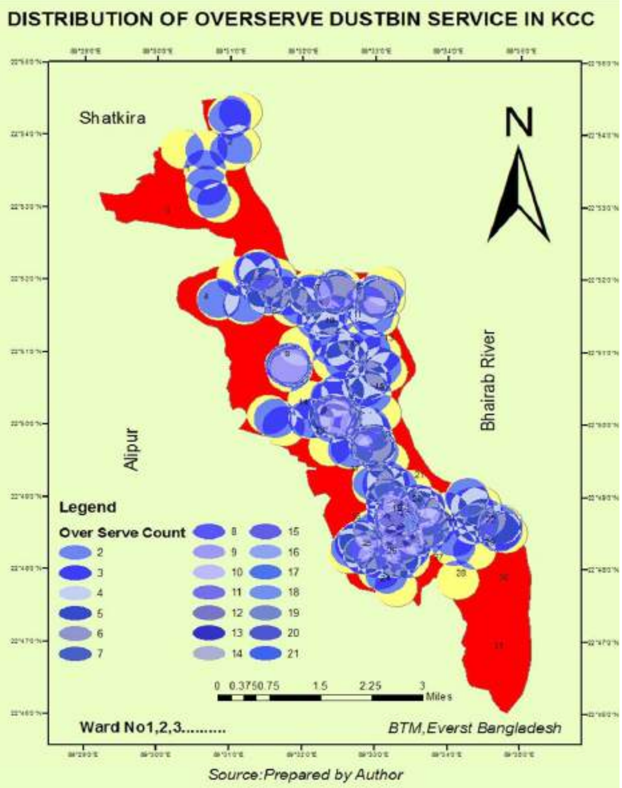


**FIGURE 13:** Distribution of overserved dustbin service in KCC.

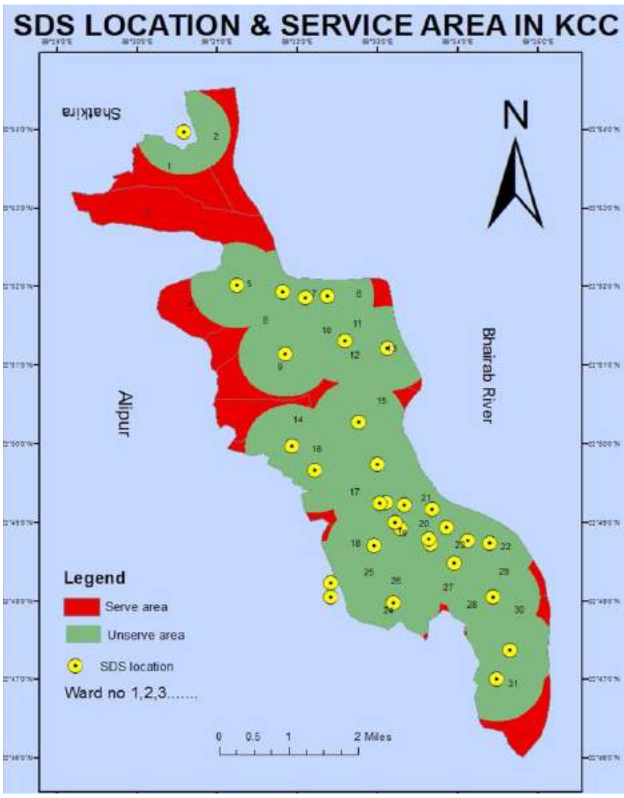


**FIGURE 14:** SDS location and service area in KCC.

ed, unconcerned people threw their wastes beside the playground, along the roadside and to the open-spaces as well which induce odor and health problems. Waste accumulation around the waste containers reflected traffic problem. The pedestrians can also get affected by odor problem induced by SW around the containers beside the roads.

## 5. CONCLUSIONS

After the analysis of the present situation of KCC area, it is found that about 55% of generated SW had been collected by KCC with the existing bins and containers. So, problem arises when 45% waste are illegally dumping which can cause various environmental problems and diseases. The location of dustbin and container is identified in the map by using GIS for better review. By analyzing the capacity of bins, the total served area is found 86.57 sq. km and total unserved area is about 40.82 sq. km. Also people's participation and concern are the most dominant supports to clean the surroundings.

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# IS THERE A FUTURE FOR THE INFORMAL RECYCLING SECTOR IN URBAN CHINA?

Benjamin Steuer [1,*], Roland Ramusch [2] and Stefan Salhofer [3]

[1] *Institute of Sinology, University of Vienna, Spitalgasse 2, 1090, Vienna, Austria*
[2] *European Bank for Reconstruction and Development | EBRD, London, United Kingdom*
[3] *Institute of Waste Management, University of Natural Resources and Life Sciences, Muthgasse 107, 1190, Vienna, Austria*

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**ABSTRACT**

This article constitutes a comprehensive overview that summarises first-hand research findings obtained by the authors through scientific engagement with the informal recycling sector (IRS) in mainland China over the past six years. During this research period, especially between 2013 and 2016, we found several indications that the working environment of informal stakeholders in urban China was worsening. Among these challenges the IRS faces, two are especially noteworthy – a decline in profits and increasing regulatory pressure from the government. In our analysis we primarily focus on the segment of the IRS that deals with collection and pre-processing (sorting, separating, cleaning, bulking, refurbishment, material extraction) of recyclables and Waste Electrical and Electronic Equipment (WEEE) in urban China. Our results indicate that despite their strong prevalence in this domain, informal actors are increasingly subjected to falling profit margins and regulatory pressure. However our results also shows that the IRS has in the past responded to both challenges rather effectively and thus may be able to maintain its presence in urban Chinese Waste Management (WM).

## 1. INTRODUCTION

From an economic perspective, the development of the People's Republic of China (PRC) is an extraordinary case: Between 1978 and 2011 the country achieved average GDP growth rates of around 10% (Taube, 2014) and thereby became the second largest economy in the world. However, this catching-up process has, especially in the last 20 years, generated high environmental costs and externalities that seriously challenge the country's environmental sustainability (He et al., 2012; Su et al., 2013). One of the problematic environmental consequences of this growth is the unprecedented increase in Municipal Solid Waste (MSW) in urban China. Between 1980 and 1998, annual MSW generation has nearly quadrupled from 31.3 million tons in the base year to 113 million tons in 1998 (Wang and Nie, 2001). In 2004, urban MSW quantities took another leap to 190 million tons (Chen et al., 2010) and in 2010 surpassed the benchmark of 200 million tons (Yang et al., 2013). This drastic development is more properly reflected in annual MSW growth rates: As there is no comprehensive data on waste quantities generated at the household level. The government and various research institutions merely present various 'guess-estimates'. These show that the annual MSW generation rate oscillated between 3-10% during the late 1990s up until the early 2000s (Wang and Nie, 2001).

In the last one and a half decades this rate increased to 6.5-10% per year (Dorn et al., 2012; MEP, 2012). A similar tendency is exhibited in the growing generation of a new waste stream, WEEE, which has only in recent years exhibited a slight decrease in growth (see Figure 1).

In parallel to the challenge of growing waste quantities, China's waste management (WM) system features three interesting developments: Firstly, the government has tackled a long existent loophole as it began to establish its environmental legislation since the mid-1990s. Given that there have been few formal institutional structures to cope with environmental aspects before that time, the Chinese central government has exhibited a considerable effort in creating a comprehensive regulatory framework. Environmental protection, Cleaner Production, the Circular Economy (CE) and WM, which were not formally covered before that period (Steuer, 2016) have been gradually institutionalised since 2000. Secondly, the Chinese government has begun to re-build a formal, physical WM infrastructure since the late 1990s and early 2000s. Initially, Chinese municipalities had developed dense, public networks to collect and process MSW from industry and households. However this system

* Corresponding author:
Benjamin Steuer
email: benjamin.steuer@univie.ac.at

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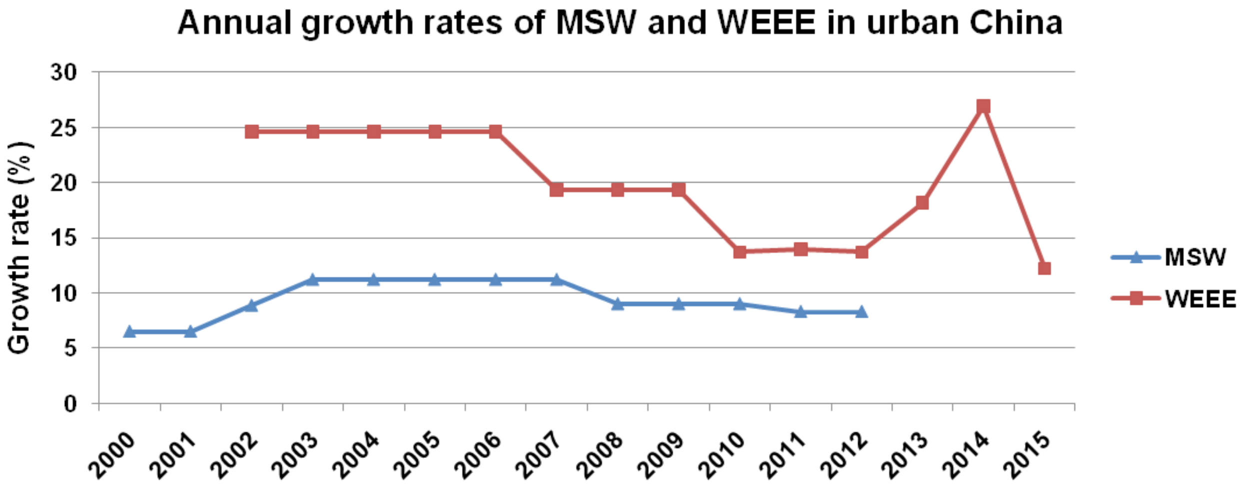


**FIGURE 1:** Annual growth rates for MSW and WEEE in urban China (Steuer et al., 2018).

fell victim to a politically initiated process of dismantling and reduction during the economic reforms of the 1980s: Formal, state-controlled MSW management was simply considered too cost intensive and strained too much of local budgets that focussed on fostering economic growth (Li, 2002; Fei et al., 2016; Steuer et al., 2017; Tong and Tao, 2016). The withdrawal of the formal sector generated an institutional leeway that directly coincided with, and in part, triggered the third major characteristic of in urban Chinese WM. Beginning with the late 1980s, a gradually growing migratory flow of rural labour entered the cities in search of work and also engaged in WM relevant activities. The emerging informal recycling sector (IRS) begun to engage in recyclable waste collection and pre-processing (sorting, separating, cleaning, bulking), as its stakeholders realised that they could commercialise these valuable secondary materials by selling them to manufacturing enterprises (Tong and Tao, 2016; Ensmenger et al., 2005; Steuer et al., 2015). Over time this new segment has established its own rule-based, organisation system and physical infrastructure, which has helped the IRS to achieve a strong informal dominance over the collection and pre-processing of waste recyclables and WEEE in urban China (Linzner and Salhofer, 2014; Steuer, 2016; Steuer et al., 2017).

Despite this story of success in the shadows of formality, recent macro-economic and WM developments in China have led to various difficulties for the IRS: Not only have informal incomes plummeted because of secondary resource price decreases over the last five years, but there are also indications that formal regulatory pressures have increasingly curtailed informal WM operations. Based on the findings of several research projects and field surveys on Chinese WM between 2013 and 2016 this paper deals addresses the question of whether the IRS in Chinese WM will be able to survive amidst this challenging environment. Our major findings are that the segment is extremely flexible and can adapt to changing demands by adjusting its institutional systems: Formal regulatory pressures induced the IRS to adapt their own collection and exchange practices. Secondly, we observe that the sector has achieved a level of maturity and service effectiveness, which not only benefits waste generating households, but also contributes to alleviating municipal budgets, as will be shown for the case of Beijing. Thirdly, recent economic changes indicate a slight recovery of domestic economic demand and thus increasing prices for secondary resources. If continued, this development may in turn reflect on the income levels of the IRS and thus sustain its operation.

## 2. MATERIALS AND METHODS

The present paper summarizes the result of several research projects: individual field investigations by Mr Steuer and a set of preceding publications of the authors (Linzner and Salhofer, 2014; Steuer et al., 2015; Steuer, 2016; Schulz and Steuer, 2017; Steuer et al., 2017; Steuer et al., 2018). The findings presented were primarily collected in the cities of Beijing and Guangzhou and supplemented by investigations in Shenzhen, Shanghai, Wuhan, Guiyu and the wider Guangdong area. During our research period over 250 structured and semi-structured interviews with stakeholders of the Chinese IRS were conducted. The intentions for doing so were twofold: First, the interviews aimed to determine and assess typical WM related particularities of the IRS, such as daily collected volumes (kg/cap/day), monthly income in Chinese Renminbi (RMB/cap/month) and related selling and buying prices of recyclables (RMB/kg or unit). Second, the interview questions also addressed sources and exchange patterns of waste recyclables obtained by the IRS. In this regard, we intended to assess the sources of waste recyclable supply of the IRS, collection and exchange routes and habits of interaction. Preliminary results from earlier interviews have gradually helped to discern informal WM infrastructures, such as Trading Points (TPs) in Beijing (Steuer et al., 2017) or informal second hand markets for electronic devices in Beijing and Guangzhou (Steuer, 2016). These network-like connected markets and TPs have further served as reference points for our observations and interviews, so as to adequately capture the transaction intensity of waste recyclables by the IRS. In regard to waste streams, we primarily focussed on WEEE and recyclables (paper, cardboard, plastics, metals, wood), because it is only these fractions that the IRS is interested in given their economic value. Finally, there is a need to mention the difficulties regarding the obtaining of data on this segment in urban China. Due to the IRS' informal nature available information is relatively sparse. The Chinese government itself does not offer any statistics on informal activities in WM. Moreover the subject itself is treated as politically sensitive since the IRS operates outside of the legal framework, offers only limited options for control by the state and is considered as an unwanted societal phenomenon (Steuer, 2016). Therefore this paper is also supplemented by earlier research on the subject matter, as well as by media reports, which often provide useful supplementary information on matters of income, size and collection quantities of the IRS.

## 3. RESULTS

### 3.1 Size, collection quantities and income levels of the IRS

With regards to assessing the dimension of the IRS, one of the most challenging tasks concerns the estimation of the sector's size. As noted in the Materials and Methods section, there are no comprehensive assessments for the Chinese case. Moreover, preceding estimations compiled by the authors vary considerably: from below two million in urban areas (Chen et al., 2010), to over 3.4-5.6 million (Linzer and Salhofer, 2014), up to as much as 18 million people (Gu et al., 2016) are documented to be active in informal WM in China. It must be presumed that these variations also originate in certain labour force fluctuations, determined by the recyclable market and respective demand for materials and collection services. Such fluctuations may be as high as 2 million people per year, which was the estimated increase in 2014 that has been presented by the central government (MOC, 2015). Our analysis for Beijing indicates a similar picture: Over the years, the numbers of informal actors have gradually increased, but not in a strict linear fashion (Table 1). Collected data are here again mostly guess-estimated figures presented by preceding research with the exception of the data set for 2013: In this year's research project we extrapolated informal actor sizes in Beijing by using informal waste markets (Trading Points) as a proxy for assessing informal actors' presence in Beijing. The numbers we extrapolated suggested that approximately 0.77% of Haidian's population are engaged in informal waste recovery. A bold extrapolation for entire Beijing thus indicated that the size of the IRS in the Chinese capital comprises approx. 150,000 individuals.

One of the key research tasks over recent years was to ascertain the collection capacities of the IRS regarding recyclables and WEEE. Generally we found that for both fractions of informal stakeholders have performed very effectively (Steuer et al., 2015; Steuer, 2016; Steuer et al., 2017). For Beijing's communal recyclable waste, various estimations contend that the IRS in urban China achieves recovery rates ('Sum of recyclables collected' divided by 'sum of recyclables generated') of around 17-38% (Linzner and Salhofer, 2014) or 20-40% (Ensmenger et al., 2005). In absolute numbers this may amount to as much as 1-2 million tons of recyclables annually in Beijing (Ensmenger et al., 2005; Solidwaste, 2015). In our 2016 research project we went one step further and juxtaposed the quantities of recyclables generated at the household level with those collected by the IRS within one sub-district of urban Beijing. Based on interviews and waste quantity documentation of generators and collectors, it was revealed that the informal sector achieves a recovery rate for recyclables of around 90% (Steuer et al., 2018). However, here it has to be kept in mind that we only surveyed one of 22 sub-districts that are subordinated to the district of Haidian. These findings can therefore not be taken as a representative picture for the entire Beijing area.

For the case of WEEE, an analysis of various preceding studies shows that informal collection recovers around 60-80% of End-of-life (EOL) devices from households in urban China (Table 2), which generate the majority (approx. 70-80%) of WEEE in urban China (Yang et al., 2008; Wang et al., 2011).

The effectiveness of informal collection becomes even more visible if juxtaposed to recovery rates of the formal system (Table 3). Despite an attempt to raise these quotas via a state initialised pilot project (the Old-for-New scheme (OfN)), formal collection still lags behind informal performance.

Here again it should be noted that the IRS plays a significant function in the context of the CE in two ways: firstly, a number of formal recycling companies (n=12) interviewed in 2015 indicated that 90% of WEEE supplied to them, stem from informal channels (Steuer et al., 2015). Secondly, informal collectors primarily aim to sell recovered devices for refurbishment and repair at nearby second-hand markets for electronics (Steuer, 2016). The primary motivation for doing so is that informal stakeholders can reap a bigger profit from selling devices for reuse than from salvaging components or selling it to formal recycling. What needs to be kept in mind is that this practice is not formally forbidden, but it contravenes the state's effort, which is to promote WEEE recycling and material extraction via its wide network of formal recycling yards (Schulz and Steuer, 2017).

Despite these strong signals, interviews conducted by the authors between 2013 and 2016 have indicated that individual collection quantities (kg/cap/year) have gradually declined in recent years. Statements from informal stakeholders have revealed multiple factors that account for this development: (1) a lack of domestic demand for secondary raw materials associated with declining eco-

**TABLE 1:** Size of informal stakeholders in Beijing's WM (1998 – 2016) (Steuer et al., 2018).

| Year | 1998 | 2000 | 2009 | 2010 | 2011 | 2012 | 2013 | 2016 |
|---|---|---|---|---|---|---|---|---|
| Number (in 1,000) | 82 | 100 | 160 | 130 | 130-300 | 140-186 | 150 -170 | 160 |

**TABLE 2:** Informal collection of WEEE from urban households in China (Steuer, 2016).

| **Informally collected WEEE** (% of quantity generated by urban households) | **Area** | **Year of measurement** |
|---|---|---|
| 60.0 | China | 2011 |
| 88.0 | China | 2011 |
| 57.0 | Peking | 2005 |
| 50.0 | Peking | 2005 |
| 30.0 | Peking | 2008 |
| 60.0 | Peking | 2010 |
| 60.0 | Peking | 2011 |
| 30.0 | Peking | 2012 |
| 76.0 | Shanghai | 2013 |
| 55.0 | Xi'an | 2010 |
| 51.0 | Baoding | 2012 |
| 50.0 | Hangzhou | 2013 |
| 37.0 | Taizhou | 2009 |
| 43.0 | Ningbo | 2003 |

**TABLE 3:** Recovery of WEEE by formal stakeholders (Steuer, 2016).

| **WEEE received by** | **Proportion of recovered WEEE (%)** | **Area** | **Year of measurement** |
|---|---|---|---|
| Recovery and take back stations | 14<br>10<br>10<br>13<br>24 | Taizhou<br>Peking<br>Peking<br>Baoding<br>Shanghai | 2009<br>2010<br>2011<br>2012<br>2013 |
| Return to retailers before the OfN (2009-11) | 16<br>4<br>14 | Ningbo<br>Peking<br>Peking | 2003<br>2005<br>2005 |
| Return to retailers during the OfN (2009-11) | 7.8<br>20<br>20<br>4 | Xi'an<br>Peking<br>Peking<br>Taizhou | 2010<br>2009-2011<br>2009-2010<br>2009 |

nomic growth in China; (2) increases in informal stakeholder numbers in urban areas have induced an increase in competition for recyclables and WEEE; (3) stronger efforts of the formal private sector to engage in material collection via online-based collection platforms (Steuer, 2016; Steuer et al., 2017 and 2017b).

The element of income constitutes the most decisive motivation for stakeholders to enter the IRS. Broadly speaking income opportunities for the IRS were comparatively high during the 1990s and 2000s and still maintained an attractive level above local minimum wages in the first-and-a-half decades after 2000 (Li, 2002; Ensmenger et al., 2005; Tong and Tao, 2016; Steuer et al., 2017). What can be gained from WM activities very much depends on the market prices for secondary materials, and these in turn are determined by the macro-economic demand for secondary resources. Moreover, the recent decline in China's GDP growth also affects informal recyclable collection quantities and in turn derived incomes. Our two surveys in Beijing (2013 and 2016) revealed that informally recovered quantities (kg/cap/day) and respective incomes have plummeted rather drastically (see Table 4). To a significant degree this development can be attributed to the decrease in falling recyclable prices. Beyond that it needs to be mentioned that the surveys differed in terms of their spatial coverage: While the results of 2013 stem from interviews within eight sub-districts in Beijing, the 2016 survey was limited to one such sub-district. In a slightly related fashion, we also discerned that recyclable markets or TPs, where incomes for collecting parties (Waste Pickers and Waste Merchants) are generated, exhibit varying price offers for recyclables. Thus the 2016 survey may in fact have only captured those TPs that offer a comparatively weak profit margin for collecting stakeholders, which in turn might only improperly reflect actual profit realities.

Assessing informal incomes generated through collection and selling of WEEE is more difficult for the following reasons: As our interview partners have stated on several occasions there is no strict specialisation between stakeholders that only focus on recyclables or on WEEE. Rather the IRS collects what it can get. Secondly, the markets for informal WEEE exchange differ significantly in terms of prices paid and offered across various cities. Thirdly, dif-

**TABLE 4:** Prices (in Chinese RMB) & recovered quantities of recyclables (metals, paper, plastics and cardboard) and income levels of the IRS in Beijing (Steuer et al., 2018).

| **Category** | **Year** | **Waste Pickers** | **Waste Merchants** | **Middle Men** |
|---|---|---|---|---|
| Number of interviewees | 2013 | 25 | 54 | 19 |
| | 2016 | 31 | 29 | 16 |
| Median collected quantity (kg/cap/day) | 2013 | 16 | 311 | 890 |
| | 2016 | 14 | 80 | 116 |
| Median net income (RMB/ month) | 2013 | 1,200 | 2,500 | 5,250 |
| | 2016 | 650 | 3,000 | 2,000 |
| Price of iron scrap (RMB/ ton) | 2015 | | 1,600 | |
| | 2016 | | 900 | |
| Price of copper scrap (RMB/ ton) | 2015 | | 38,000 | |
| | 2016 | | 32,000 | |
| Price of aluminium scrap (RMB/ ton) | 2015 | | 10,300 | |
| | 2016 | | 7,600 | |

ferent recyclable categories reap different prices and that is especially reflected in preferences of the IRS for foreign brands vis-à-vis domestic ones. This aspect is related to different quantities of valuable components inside the devices, as well as to brand reputations influencing the resale potential of refurbished at second hand markets (Steuer et al., 2015; Steuer, 2016; Schulz and Steuer, 2017).

### 3.2 Institutional systems of the IRS

The first structural aspect that characterises the IRS in China as well as in the global domain (Wilson et al., 2006) is the high degree of internal organisation. Based on observations in different Chinese cities we identified three different groups that engage in different activities in informal recyclable collection (see Table 5): firstly, the group of Waste Pickers (WPs), who roam through the streets by foot and primarily screen waste bins in public and residential areas for recyclables. Given their limited collection capacity this group recovers smaller quantities than Waste Merchants (WMs), who focus on doorstep collection directly from households and who, due to the use of tricycles, can transfer collected recyclables over relatively wide distances. The third informal group are Middle Men (MM), who buy recyclables from the two collector groups and use trucks to transfer these materials to their private depots. There, MM pre-process the fractions (storing, cleaning, separation, and sorting, refurbishing, material extraction) and then sell these to manufacturers or recyclers (Steuer et al., 2017). Trading Points (TPs) as infrastructural nodes for exchange play a decisive role and will thus be discussed separately in the following paragraphs.

The collection service efficiency of the IRS is strongly dependent on how its stakeholders engage with the suppliers of recyclables, i.e. primarily households. In this regard, those persons in charge of recyclable and WEEE collection adopt a pro-active stance to connect and establish relationships with households. One practice is that WMs roam through residential quarters and offer their services through verbal announcements. Another often practiced way is that WMs set up cardboard signs in front of residential compounds, on which their mobile phone number and the recyclables or WEEE devices they collect are stated (Figure 2).

Moreover, many collectors also make use of business cards to connect to their customers. In this regard, interviewed WMs stated that good communication, i.e. ensuring the flow of information, is most decisive for increasing working efficiency: Arrangements via phone calls provide stakeholders with the flexibility to align collection time and routes with actual household demand (Li, 2002; Steuer et al., 2015). The core strategy behind this practice is that WMs pursue the development of a steady customer base and therefore mutual reliability and trust have to be fostered. For households the pivotal reason to cooperate with the IRS can be attributed to a particular value concept that Chinese households apply to recyclables and WEEE. In contrast to western societies, Chinese residents perceive discarded recyclables and EOL devices not as rubbish, but as a valuable resource that if sold can contribute to overall household income (Zhang and Wen, 2014). This stance explains why urban Chinese households exhibit a preference for those types of WM services that offer a pecuniary reward or compensation in exchange for waste recyclables. As a result Chinese households have expressed a strong preference for informal collection and relatively little interest for formal collection, which cannot or only in a reduced form offer monetary compensation for waste recyclables (Steuer et al., 2018).

At the level of physically manifested systems, the IRS has produced a highly innovative system, which is the aforementioned TPs (Steuer et al., 2017). These infrastructural nodes are of a mostly small and scattered nature, resembling small markets, which are composed of small booths and trucks with mobile ground scales. At these highly mobile points that are open for at least half a day recyclable- and WEEE-collecting stakeholders (residents, WPs, MM) convene and exchange materials for money. MM operate as buyers, who further process waste recyclables before transferring them to industry or recycling (Figure 3). The system itself combines elements of innovation as well as imitation: TPs are innovative in the sense that they are a bottom-up solution devised by informal stakeholders so as to overcome long transport distances. As we extrapolated in our 2013 study, Beijing's sub-district Haidian features a TP-to-resident proportion of around 0.87 TPs per 10,000 inhabitants. This particular spatial distribution of TPs thus facilitates collection and transfer activities of the involved stakeholders. At the same time however the TP system bears a strong resemblance (imitation) to the prior existing formal Supply and Marketing Cooperatives, which were established in Maoist China and which featured similar functions and density as the TPs. Thus the IRS appears to have re-innovated a former formal, effective system.

As for the aspect of WEEE collection and refurbishment the IRS has further developed an upgraded version of the TP system. At its centre are second-hand markets for refurbished and repaired electronic devices. These operate

**TABLE 5:** Informal stakeholder characteristics in urban Beijing MSW (Steuer et al., 2018).

| Stakeholder | Waste Pickers | Waste Merchants | Middle Men |
|---|---|---|---|
| Means of transport | By foot | Tricycle | Truck |
| Source of recyclables | Public bins, bins in residential quarters | Households (doorstep collection) | Small enterprises, households, Trading Points |
| Waste management activity | Collection, selling at Trading Points | Collection, selling at Trading Points | Buying at Trading Points, storage, sorting cleaning, bailing material extraction selling to industry or recycling |

**FIGURE 2:** Cardboard signs of Waste Merchants in Beijing, Haidian district (© Steuer, Ramusch and Salhofer, 2016).

under the premise that the collected devices still possess a certain reuse value and are thus not transferred to recycling by informal collectors. Here again, the decisive motivation for informal stakeholders to sell End-of-Life devices to second-hand markets for refurbishing and reuse is that the profits generated are higher than what is offered by formal or informal recyclers. The legal position of these markets is situated somewhere between the formal and the informal realm: In many markets investigated in Beijing and Guangzhou, sellers have, in most cases, obtained licenses for their operations. On the other hand, however, the refurbished devices often lack legal warranties and proper labelling (Steuer, 2016). The functionality of the markets itself is again very similar to the TPs (see Figure 4): households and businesses as generators of obsolete devices either directly exchange EOL electronics at markets for monetary compensation or they sell their obsolete devices to informal collectors, who then further transfer these to the markets. Another, third transfer channel, comes in the form of repair services. These are generally small shops that sell devices obtained from consumers to markets or in some cases they provide repair services directly to second-hand markets (see Figure 5). Finally, markets serve as suppliers of refurbished devices to households/ consumers, who are attracted by comparatively cheap second-hand electronics

As we alluded to above, the main purpose of second-hand markets is the generation of high profits for the IRS. Profit margins themselves are highly dependent on the device age and thus possible demand as well as the location of the markets itself: for example, in 2015, profit rates of 100% appeared to be quite common for 25-inch TVs that were not older than 5 years. These profits however are also subject to the location of markets within the respective city: preliminary observations by Steuer and Schulz have indicated that the more second-hand markets are located at the outskirts of city centres, the lower the profit margin will turn out (Steuer, 2016).

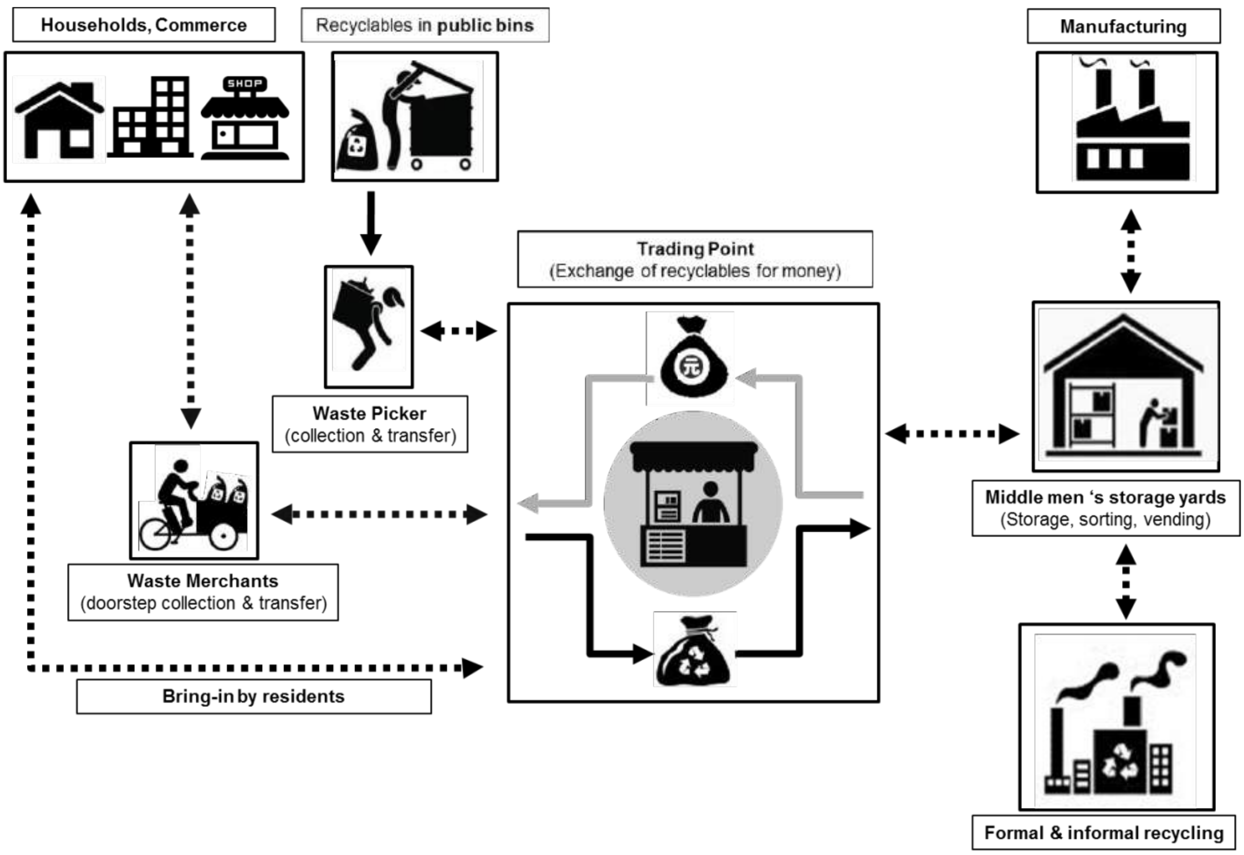


**FIGURE 3:** The connective function of Trading Points [Grey arrows indicate monetary flows, black arrows refer to recyclable flows and two-directional arrows indicate the exchange of money for recyclables] (Steuer et al., 2017).

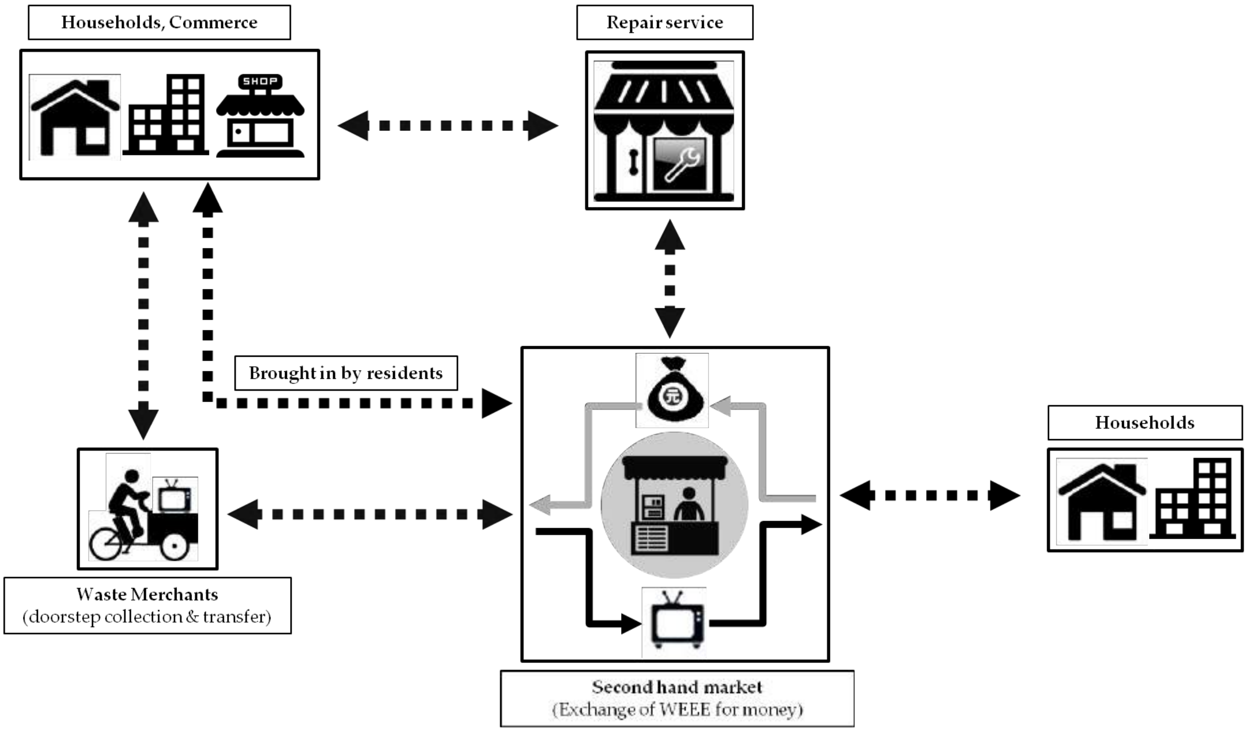


**FIGURE 4:** WEEE transfers for reuse via second-hand markets (Grey arrows indicate monetary flows, black ones represent WEEE flows and two-directional arrows indicate the exchange of money for WEEE) (Steuer, 2016).

### 3.3 Challenge 1: declining profits for recovered recyclables and WEEE

Despite the evidence of the IRS' effective performance there are still some challenges confronting informal stakeholders in WM. The first problem relates to the decline in profits for recovered recyclables and WEEE. Via several interviews conducted in 2016, informal stakeholders have repeatedly stated that market demand for recyclables has significantly decreased since 2014 (Steuer et al., 2018). These perceptions are mirrored by China's online recyclable price indices: between 2015 and 2016, the values of secondary copper, aluminium and iron decreased by 16%, 26% and 44% respectively (CRRA, 2017). Such deterioration of prices, according to the interviewees, has affected all recyclable fractions except for paper and cardboard, which in turn has led to some negligence in the informal collection of the less valuable recyclables. More importantly for the IRS however is the influence of these declines on


**FIGURE 5:** Guangzhou's Dashatou second-hand market with (a) trading area and (b) integrated repair service structures (© Yvan Schulz).

the incomes of stakeholders (see Table 4). This situation has been further aggravated by the fact that competition between stakeholders has increased due to a continuously growing influx of newcomers in the sector. In this regard, the China Ministry of Commerce has estimated that there were about 2 million new persons entering the IRS in 2014 alone (MOC, 2015). Both factors have led to lower overall incomes given lower collection quantities (kg/cap/day) resulting from stronger competition and lower recyclable prices. Nevertheless, both factors may have no effect on overall recyclable and WEEE collection dominance of the IRS as higher stakeholder numbers may still recover the daily generated recyclable quantities.

The challenge of declining profits also extends to the informal WEEE refurbishment and trading. Here however, the impetus comes from a slightly different angle. On the one hand, households as primary generators of WEEE in urban China have increasingly realised the potential economic value of these EOL devices. By implication, this makes it harder for the IRS to achieve a good profit from this first instance of transaction. Moreover, decreases in profits originate in changes at the second step of transaction: the markets themselves, at least those observed in Beijing and Guangzhou, have undergone a gradual process of government-induced formalisation. Measures implemented over recent years aimed to formalise the processes of refurbishment and resale, i.e. via the issuance of obligatory service guarantees offered to customers, billing and invoice requirements as well as increasing documentation requirements. Interviewed traders have stated that such measures have indeed affected their profits to the point that some saw leaving this business as the final option (Steuer, 2016). In Guangdong province's Guiyu town, second hand markets for electronic components were even regulated to such a degree that trading stalls were empty during our visits. The reason for such, local traders stated, was that salvaged component prices were too high. This in turn was a result of various fees imposed on traders by the local government, which were then translated into higher component prices. In response to the question as to why the traders had entered the market, they explained that the local government had confronted them with the choice to either to integrate and resettle within the industrial park's market, or to close down their operations (Steuer, 2016).

### 3.4 Challenge 2: formal regulatory measures against the IRS

The strong and lasting dominance of informal actors in urban WM has quite instinctively lead municipal governments to respond. After having withdrawn from WM in the early 1980s and thus leaving the field open to informal engagement (Li, 2002; Fei et al., 2016; Tong and Tao, 2016), the state returned in the mid-1990s by establishing a formal institutional structure to tackle the challenge of urban waste. Despite the fact that the centrally issued, major legislative pieces such as the law on solid waste left the aspect of collection relatively undefined, municipal governments addressed collection and also attempted to manage respective activities by the IRS. What is shown in these rather general legislative pieces of municipal governance is that local governments have predominately adopted a prohibitive stance against the IRS (Figure 6). Among the measures, prohibitive ones primarily aim to expel informal WM activities from public places, while integrative approaches include organisation under, and cooperation with, official authorities.

The implication of these measures is manifested in daily practises of local governance. In regard to the prohibitive measures, the aim is to set up alternative, formal structures. In regards to this aspect, informal interviewees indicated that TPs are more often subject to control and monitoring, which often forces stakeholders to suspend their

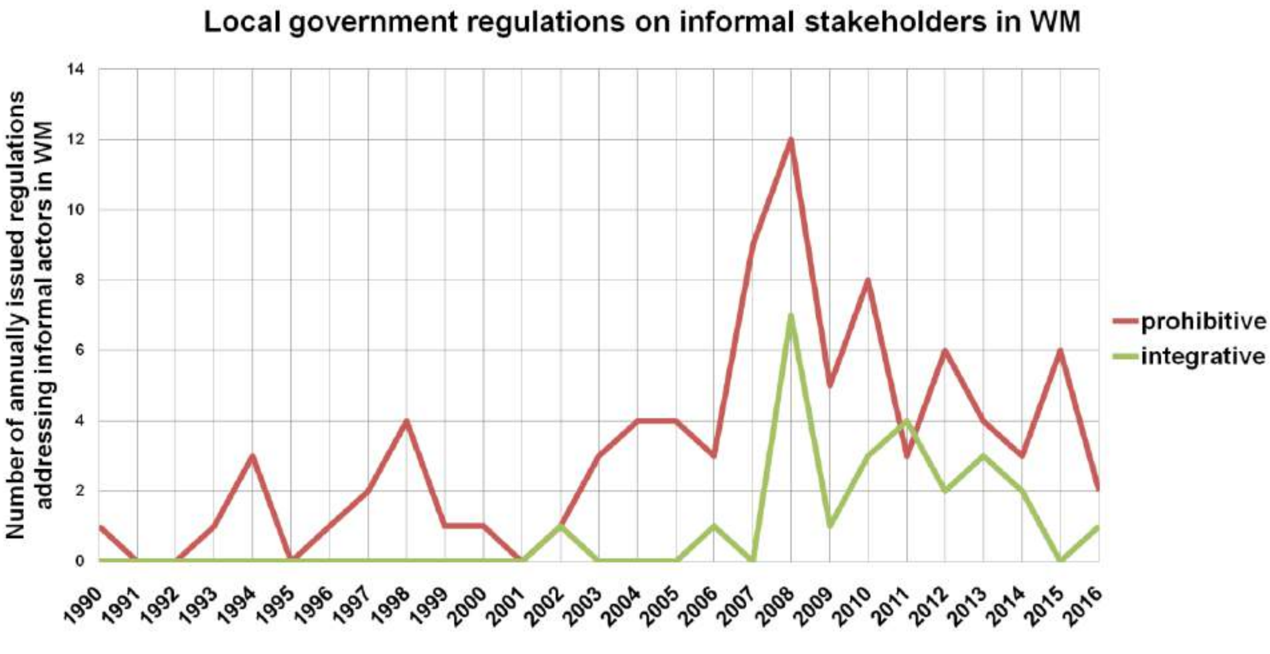


**FIGURE 6:** Local regulations on the informal sector in urban China (Steuer et al., 2018).

operations. Second hand markets are, as mentioned above, increasingly confronted with the necessity to formalise. At the next higher level, the so-called 'waste villages' that are situated in the outskirts of cities such as Beijing's Dongxiaokou, are increasingly forced to relocate. All of these measures affect the operation of the IRS. Moreover, local governments have since the 2000s implemented pilot programs that aim to set up a parallel formal waste collection system. Formal collection stations have been established within residential quarters so as to directly channel waste recyclables into the formal system. In Beijing alone the number of these booths has been gradually growing and as of 2014 stands at 4,400 units (Zhang and Wen, 2014). Quite clearly the overarching objective of this step is to establish the link between generators and the formal WM system, while the IRS should simultaneously be cut off from access to waste recyclables and WEEE. In regards to the integrative approaches adopted by local governments, the general idea is to incorporate the IRS within the formal framework e.g. via the issuance of licenses and a top-down organisation via the municipal government. Such efforts were however met with rejection by the IRS, which perceived this approach as highly negative due to the high cost burdens that officials demanded for the licensing process (China.com, 2007; Chen at al., 2010; Zhou, 2010).

## 4. DISCUSSION

We have presented an overview on the status-quo of the IRS in China together with two significant challenges that it faces. The first of these two, i.e. the aspect of prices, is relatively simple as it is primarily a question of domestic demand for secondary resources. Given the decline in prices in 2014-2015, informal stakeholders, especially MM, have resorted to stockpiling recovered recyclables and waiting until prices recover (Cnenergy, 2016). There is a good reason for adopting this 'wait-and-see' strategy. Firstly, the IRS, especially in Beijing, has in its past already been confronted with a similar situation. In 2006 and 2008 the sector was faced with a decline in secondary raw material price declines. In both instances, the IRS managed to survive and adapt to the changing environment. In both instances, lower incomes were accepted by those who remained in business, while others simply abandoned the sector (Chinadigitaltimes, 2016; Sina Finance, 2006). Secondly, recent macro-economic developments hint at a recovery of domestic demand among manufacturing and export oriented industries (SCMP, 2016a and 2016b), which apparently induced a rise in secondary raw material prices in late 2017 (CRRA, 2017). Thus, the challenge of declining profits in the IRS might be regarded as a typical example of a temporarily limited business cycle. This in turn leads to the assumption that the existence of the IRS in waste recyclable collection might be safeguarded in the near future. What is more difficult to assess are the future profit challenges associated with informal WEEE management. Whereas profits from informally collected WEEE that goes to recycling is equally subject to secondary raw material demand, the story is slightly different for EOL devices that are sent to second hand markets because the informal value chain is dependent on consumer preference constellations. While refurbished electronics were especially appreciated by urban consumers in the 1990s and 2000s, the gradual growth of the middle class in urban areas has changed this trend (Steuer, 2016). These consumers are now increasingly looking for high quality products, which leads to second hand goods being increasingly sold to consumers with lower income. It is quite likely that this will also lead to a decline in profits for second-hand, refurbished devices. In summary, we assume that there is a higher potential for these parts of the IRS that engage in waste recyclable collection than for those that deal with WEEE collection and refurbishment.

The second major challenge relates to the increasing rigidity of formal regulations. Initially, the IRS in recyclable and WEEE collection could only emerge because of the decline of formal WM structures. The response of the central government to issue a set of regulatory measures has been ambitious. For example, the collection of recyclables and WEEE has been absolutely ignored in key laws and regulations (Steuer et al., 2015; Steuer et al., 2018; Schulz and Steuer, 2017). In a slightly more ambitious fashion, there have been a few, tentative ad hoc measures to formalise second hand markets for repaired electronic devices. Despite these measures the IRS is yet able to continue its operation in this field (Steuer, 2016). Moreover, those regulations at the local level that directly address the IRS in collection primarily employ a prohibitive approach to confront the segment. Given the historical development in urban Chinese WM, this strategy might however be prone to fail as it ignores the lessons from the dynamic relationship between the formal and informal sector. In various urban areas the attempts to crack down on the IRS have only resulted in evasive measures of its stakeholders, who respond by resettling or transferring their businesses to nearby areas or simply become more vigilant towards formal controls (Steuer, 2016; Tong and Tao, 2016; Goldstein, 2017). In an attempt to replace informal collection the Beijing municipal government, among other major cities, initiated a pilot project in 2000, which aimed at establishing recyclable and WEEE collection booths within residential quarters. This was done to cut the IRS out of recyclable collection from households. However there were some flaws in the implementation of this otherwise cunning concept: firstly these depots offer less money to households for their recyclables than the IRS does. Secondly, the operators of these booths are mostly former informal stakeholders, who took this low paid position (1,500RMB/ month) (FON, 2013) that ordinary Beijing residents would refuse to take (Wang et al., 2008; Tong and Tao, 2016). Given that these operators still have good contacts to the IRS, they tend to resell these now formally collected recyclables back to the informal sector. Therefore the historical development (see Figure 7) between the activities of the formal (left section in the picture) and the informal system (right section in the picture) can be seen as a dynamic, reciprocal relationship in which the IRS has continuously contravened and reacted to formal regulations and measures over time. This is not to say that formal regulations are entirely ineffective. Rather the IRS has developed a strong flexibility to cope with and adapt to formal measures.

What needs to be discussed in this regard is that despite all the problems informal collection might cause for the formal system, it simultaneously creates an immense benefit for municipal WM budgets. Again, the case of Beijing offers a suitable example to explain this matter: in 1998, Beijing allocated 750 million RMB out of its budget to managing MSW. At the same time the IRS achieved a profit of around 1.1 billion RMB (Ensmenger et al., 2005). In regards to how much the performance of the IRS alleviates official WM expenditures data for the years 2010 and 2014 are particularly instructive: in 2010, Chinese experts estimated that practices of the IRS helped to save 400 million RMB in annual official expenditures on WM (Solidwaste, 2010). This may be explained by two factors: firstly, in Chinese WM, the most cost intensive item is personnel costs, which in some municipalities range around 55-70% of total WM expenditures (Ren and Hu, 2014). Given that the IRS dominates the labour-intensive collection operations, any abolishment of the sector would not only transfer this task to formal services and thus increase formal overall expenditures. The formal sector would also have to establish the as of now lacking processing and transfer structures for securing the potential gains from waste recyclable recovery.

The contribution of the IRS further increased in 2014: in this year Beijing's annual waste treatment costs amounted to 2 billion RMB (in 2014), while the services of the IRS simultaneously alleviated the municipal budget by approximately 1.8 billion RMB (Crrainfo, 2014). By implication, dispersing the IRS and substituting it's services would in turn force the municipal government to double its WM expenditures (see Table 6). If this cost-benefit element is juxtaposed to the larger perspective of the formal-informal dynamic in waste collection, one may understand why the Chinese government has not yet fully prohibited the IRS: it simply appears that the government intends to make use of this segment as long as possible to indirectly benefit from its contributions, while it simultaneously attempts to build up a formal system via pilot programs so as to substitute the informal sector on the long term.

## 5. CONCLUSIONS

In conclusion, the IRS in urban China is facing a variety of challenges, which have arguably worsened the working conditions of its stakeholders. These factors stem from domestic market volatility as well as from increasing regulatory efforts by the Chinese government. Both challenges do in fact affect the IRS but may not threaten its existence on the long term. Especially with regards to the second challenge it appears quite startling that despite many years of attempted curtailment the Chinese government still holds on to prohibitive measures to control the sector. This not only runs contrary to the Chinese domestic experience and the growing numbers of the IRS over the years. It also appears to neglect the experiences of other low- to middle-income countries like the Philippines or Brazil, where informal stakeholders have been integrated into the formal WM system (Steuer, 2016). Given the flexibility and proactive response of the IRS to formal measures, this prohibitive regulatory stance might be prone to fail. Therefore it might in fact be the first challenge that

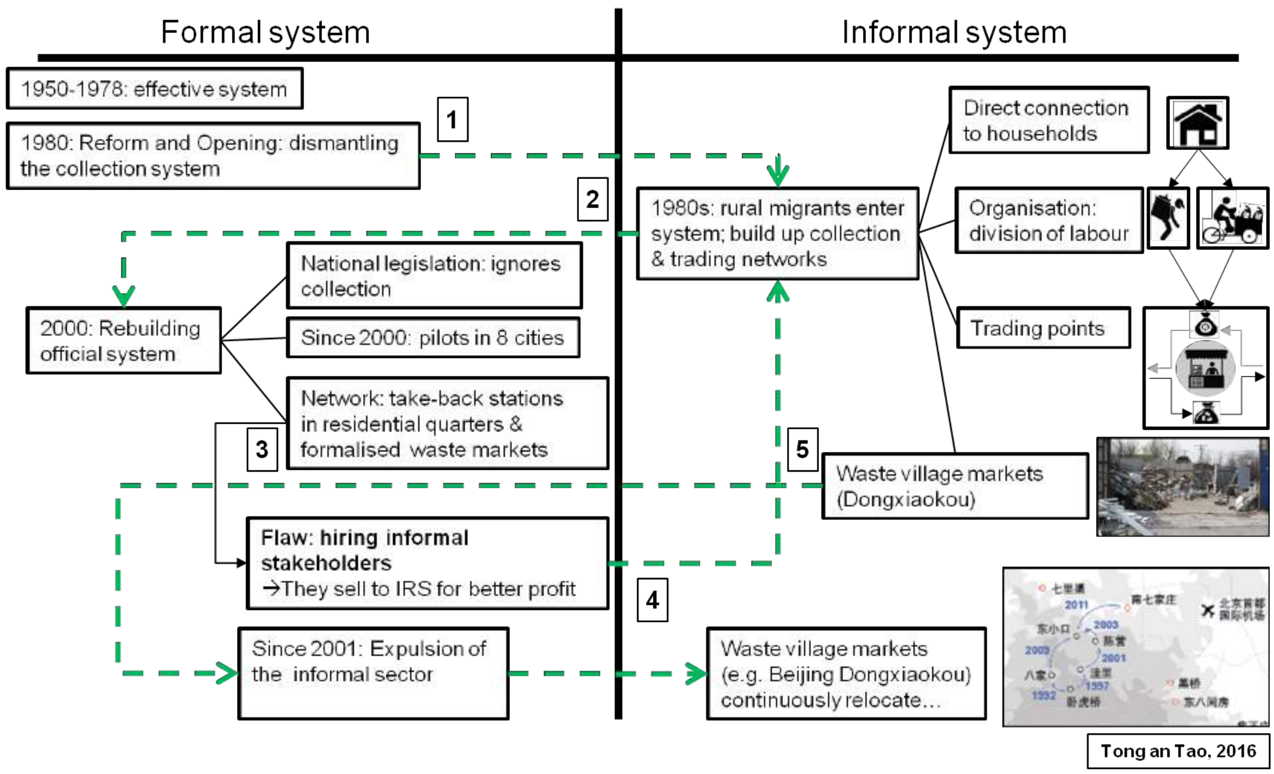


**FIGURE 7:** The interdependent evolution of the formal and informal systems in Beijing's WM: Responsive actions of each domain are indicated by green arrows connecting the action of one side with the response of the opposite side (Steuer et al., 2018).

**TABLE 6:** Municipal budget expenditures on WM and respective cost savings of the IRS in Beijing.

| Year | Expenditures on WM (in million RMB) | Performance of the IRS: Cost savings for municipal budget (in million RMB) | Additional WM expenditures without IRS (in million RMB) |
|---|---|---|---|
| 2010 | n.a. | 400 | n.a. + 400 |
| 2014 | 2,000 | 1,800 | 3,800 |

can exert the biggest pressure on the segment. Should these stakeholders face continuously decreasing income opportunities over longer periods, then the sector will eventually begin to dissipate. Based on our observations, we assume that this development will not occur. Nevertheless, anecdotal evidence has indicated that incomes in this field have slowly, but gradually gone down over the last two decades.

## CONFLICT OF INTEREST

The authors hereby state that there is no conflict of interest in regard to the publication of this paper.

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# INFO FROM THE WORLD

## POWER DYNAMICS AND CONFLICT OF INTERESTS IN THE WASTE SECTOR: THE CASE OF NAIROBI, KENYA

Waste management raises concerns on environmental degradation and public health issues in the city of Nairobi, Kenya where a big share of the urban population still suffers from lack of adequate waste collection services. Governmental and international efforts have focused on implementing waste operations in middle- and high-income areas that fall short of providing informal settlements with equally efficient facilities (Figure 1). This results in increased rates of urban inequalities as well as places waste issues in the broader framework of environmental injustice and marginalization of the poor.

In response to the current gaps, the emergence of spontaneous, informal clean-up activities sheds light on the willingness of communities to be more involved in waste management activities. To that extent, consideration should be given to environmental education and awareness programs to provide the general public with skills and training in the waste sector. It is also important to include them in decision-making processes by supporting participatory planning and co-design of innovative waste management strategies.

There are multiple causes preventing the conduction of efficient Solid Waste Management (SWM) services. Among those, spatial and management concerns are raising about the increasing number of unauthorized dumping sites as opposed to the Dandora Municipal Dumping Site which has been declared "full" for years and keeps on receiving waste material to be disposed of (Kimani 2007).

**FIGURE 1:** Waste collection in informal areas and slums by CBOs. Credits by the author (2016).

Not only does lack of spatial control exacerbate potential risks to human health and the environment, but it also triggers conflicts of interest among the various stakeholders driving illegal businesses and waste trafficking (Njoroge et al., 2014). Illegal waste activities refer to any waste movement that lays outside the regulatory framework Tompson and Chainey, 2011). Both increased levels of urban poverty and lack of regular income foster scavenging operations to sell recovered materials.

Currently, the collection rate is 33 percent, which means that 2,690 tons of waste material (Figure 2) remains uncollected every day. About 100 to 150 tons of waste are reused and recycled daily, equivalent to 3.7 percent of the total waste generated. Those services account for waste fees paid directly to the City Council of Nairobi (CCN) where residents are charged on the basis of the service supplied. However, municipal collection services only cover medium- and high-income areas while cutting out low-income areas and slums which are where more than a half of Nairobi's population lives (Njoroge et al., 2014).

## RECOMMENDATIONS

**Easy access to SWM**. Waste services should be extended to a broader share of Nairobi's population by supplying households in both high-income and slum areas and reducing the fees for the municipal service. Collection activities should be scheduled on a weekly basis and be coupled with regular clean-up of streets and public spaces as well as monitoring of collection and disposal areas.

**Nudge interventions.** These interventions address the installation of collection sites in open and common spaces where people can deliver garbage. Collection points will be easily recognizable and associated with informational signs. They will also host workshop activities to educate local communities on the importance of recovery and recycling.

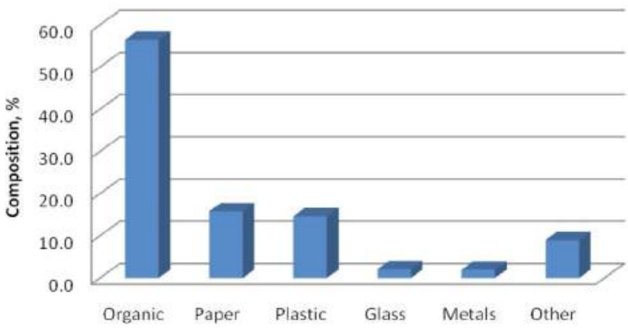


**FIGURE 2:** Physical composition of MSW in Nairobi (Njoroge et al., 2014).

Managing editor:
Elena Cristina Rada
email: elena.rada@uninsubria.it


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**Incentive-based system.** This will provide financial rewards for those who segregate materials and bring garbage to collection areas by attaching monetary values to waste resources and giving new life to recovered materials. With hopes of promoting pro-environmental and more responsible behaviors, we aim to change injunctive norms by promoting good waste practices.

**Hybrid governance in the waste sector.** There is a need to coordinate public and private efforts in the waste sector in order to avoid conflict among resources and illegal competition for inadequate and insufficient service provision. To that end, we advocate better management of the power structures playing in the waste sector by supporting collaboration and shared responsibilities among CCN, local and international NGOs, private companies, and local communities.

**Public awareness and informational interventions.** Beyond action, awareness on waste pollution as a major cause of environmental degradation and disease outbreak is needed. Education on the ease of recycling also plays a crucial role. This can be achieved by installing containers for recyclables, informational signs on trashcan use, how to recognize different materials, facts about the downsides of illegally disposed waste, and examples of successful cases in similar contexts.

**Participatory planning and co-design.** Local communities should be more empowered in the environmental clean-up process by co-designing strategies and taking part in decision-making along with municipal actors and private waste companies. This would allow for the creation of a more responsible, cleaner and environmentally-friendly system where roles and benefits are equally shared among Nairobi's population thereby promoting procedural justice.

Carol Maione [1,*] and Negar Aliakbarshirazi [2]
[1] *School for Environment and Sustainability, University of Michigan, Arbor, USA*
[2] *Department of Architecture and Urban Studies, Politecnico di Milano, Italy*
* email: cmaione@umich.edu

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# RESEARCH TO INDUSTRY AND INDUSTRY TO RESEARCH

## E-BIOPOND® - COUPLING MICROBIAL ELECTROCHEMICAL TECHNOLOGIES TO RACEWAY PONDS TO RECOVER ADDED VALUE FROM BIO-WASTE LECHATES

The increasing interest for energy-positive and carbon-negative waste treatment processes has led to investigate hybrid biological treatment, using photosynthetic microorganisms (PM) cultivation. In these systems, an equilibrium between heterotrophic aerobes and PM is maintained: the aerobic oxidation of organic load is sustained by photosynthetic dissolved oxygen (DO) and nutrients uptake by PM simultaneously leads to remove soluble minerals. Depending on the type of lechate, large amounts of K, N, and P are usually present, together with trace elements (B, Cu, Zn, Mo, Fe, Co and Mn), required for the growth of PM. In these systems, mechanical aeration to stimulate organic carbon oxidation could be avoided (Molinuevo-Salces et al., 2010). Besides, the obtained PM-rich sludge can be a source of added-value hydrolyzates (e.g. bio-fertilizers, bio-stimulants and plant growth promoters) (Acién et al., 2016) or molecules - e.g. natural dyes, anti-oxidants, bio-polymers, carotenoids etc. (Ledda et al., 2016).

One major concern of such approach (especially for organic-rich leachates) is the fragile equilibrium between PM and heterotrophs and the low quality of the obtained PM biomass (Acién et al., 2016). Especially for the treatment of organic-rich streams, such as animal slurries, agro-food industry wastewater and digestate from anaerobic digestion, the presence of easily bio-available organic carbon at high concentrations (soluble-COD > 1 g/L) favor heterotrophic bacteria growth over PM, easily driving the system to anaerobic conditions, where PM are generally inhibited (Olguín, 2012). Suspended recalcitrant organic compounds, also, tend to increase water turbidity and limit light penetration into the culture. Contamination by pathogenic bacteria present in the leachates can also limit PM growth and contribute in lowering process efficiency. Finally, excess of ammonium and other inorganic compounds can have direct inhibiting effects on PM growth rate (Ledda et al., 2016).

Here, we propose the e-BioPond® concept, based on integrating microbial electrochemical technologies to regular raceways-like ponds. This concept was developed at the e-BioCenter – University of Milan, Italy and an explorative experiment was recently published (Colombo et al., 2017). The e-BioPond® uses the electrochemical gradients generated by special microbial electrochemical pipes (MEP), to recover nutrients and oxidize organic carbon by anaerobic electroactive microbes (see Figure 1). The electrochemical system allows maximizing organic carbon oxidation to $CO_2/HCO_3^-$ at the anode, while reducing the oxygen produced by photosynthesis in the separated cathodic chamber. Simultaneously, inorganic carbon and dissolved nutrients are allowed to diffuse through the separator to the cathodic chamber, where PM utilize them as growth medium. The MEP physically separates (by porous materials) PM from the heterotrophic microflora, while guaranteeing strong electron acceptors to anaerobic oxidizers trough the electrochemical system. This would allow obtaining high-quality PM biomass and higher process stability, as compared to state-of-the-art technology.

The system, firstly tested at labscale with a culture of Arthrospira Maxima. The system was fed in semi-continuous mode by pre-decanted swine slurries (3-5 g sCOD/L) to the anodic chamber. The hydraulic retention time in the anodic chamber was around 5 days, to obtain high COD removal rates (95%, 0.65 gsCOD $L^{-1}$ $d^{-1}$). Considering the overal volume of the raceway pond at demonstration scale, these figures correspond to around 7.1 $g_{sCOD}$ $m^{-3}$ $d^{-1}$.

At the cathode, photosynthetic DO (15-18 ppm) sustained MEP operation (electrical power densities 4 $W/m^2$) with considerably high coulombic efficiencies (30-50%). This improves mineralization rates and liberates mineral

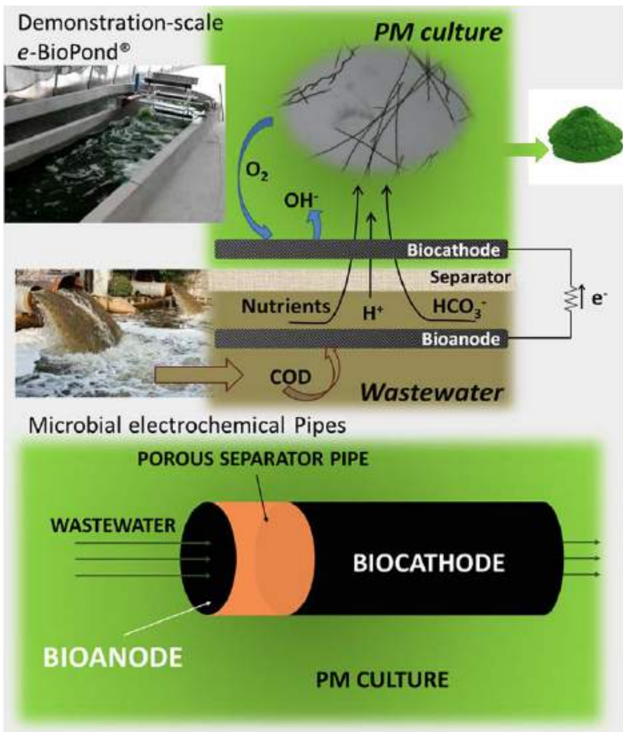


**FIGURE 1:** e-Biopond demonstration scale facility.

Managing editor:
Alberto Pivato
email: alberto.pivato@unipd.it

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forms of carbon and nutrients in the anodic solution. Ammonium are oxydized by different mechanisms (either anodic or microaerophilic at the interface oft he porous separator). Bicarbonate, nitrate and nitrite, as well as ionic forms of other nutrients tend to diffuse to the cathodic chamber through the porous separator. Recovery yields from anodic to cathodic compartments were measured in the range 30-70%, depending on the element. Spirulina growth was regular and at a rate of around 40 mgTSS $L^{-1}d^{-1}$ (Colombo et al., 2017), similarly to control cultures fed with standard culture medium. Another effect of the electrochemical system is the increase of pH in the cathodic compartment, due to incomplete $O_2$ reduction reaction towards an accumulation of hydroxyl ions. In the case of alkalophilic PM cultures (e.g. Spirulina), this is a particular advantage for maintaining harch conditions for pathogenic microbes and in general to keep the monoculture stable.

From June 2018, the e-BioPond® system is being tested in a prototype at a demonstration-scale (raceway pond area of 26 $m^2$ and volume of 3-5 $m^3$), in collaboration with the companies Algaria srl and FCTechnics (EU-H2020 Neptune accelerator project). This plant is located in a greenhouse in proximity of a biogas-production plant and the co-generation unit provides heat to the e-BioPond unit. The raceway pond is maintained at around 28°C by underfloor heating exchangers. Table 1 describes the main process parameters used in the first tests.

**TABLE 1:** Main characteristics and performances of the e-BioPond® system at demonstration scale.

| | Scale | Demonstration |
|---|---|---|
| Photobioreactor volume | L | 3000 - 5000 |
| HRT in anodic compartment | d | 5 |
| Organic concentration | gsCOD/L | 3 |
| Organic loading rate | g sCOD/$m^3$pr/d | 7.5 |
| Volumetric loading rate | L /$m^3$pr /d | 2.5 |
| Removal rate | g sCOD/$m^3$pr/d | 7.1 |
| sCOD removal efficiency | | 95% |
| Influent volumein batch | L/d | 28.8 |

From the obtained biomass, Algaria srl is now extracting phycocyanin, a protein backbone to which linear tetrapyrrole chromophores are covalently bound to cys-teine residues via thioether bonds. Phycocianin has characteristic blue color, absorbing orange and red light (around 620 nm) and is used for its commercial value as natural colorant in nutraceutical, cosmetic, and pharmaceutical industries. Spirulina biomass obtained by the e-BioPond® concept is also under study, for its use as protein-rich feed supplement for fisheries, poultry and pigs.

Further info and contacts can be found at the website: https://sites.unimi.it/e-biocenter.

Andrea Schievano [1,*], Bruno Rizzi [1], Andrea Goglio [1], Giovanni Rusconi Clerici [1], Rosaria Tizzani [1], Matteo Tucci [1], Matteo Broggi [2], Matteo Lucchini [2] and Antonino Idà [2]
[1] *e-BioCenter - ESP, University of Milano (Italy)*
[2] *Algaria srl, Milano (Italy)*
* email: andrea.schievano@unimi.it

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# PORTRAITS

**Professor Em.Dr. ALFONS BUEKENS**
born in Aalst (1942), Flanders, Belgium

Prof. Dr. Buekens was one of the first who researched about the environmental impact of dioxins and furans the and did a great documentation of this scientific area. Next to that his speciality was research on the recycling and treatment of chemical and hazardous wastes.

On the european level he released impulse to modern waste management.

## Studies and training

- Secondary school: Royal Athenaeum Aalst, 1947-59 (greatest distinction)
- State University of Ghent: Degree in Chemical Engineering 1959-64 (great distinction; M.Sc. (1964) and Ph.D (1967) at Ghent University (RUG);
- Applied Sciences, Ph. D. on "The thermal cracking of Hydrocarbons", 1964-67 (greatest distinction)
- Intensive course German (1962-64)
- Trainee in the R&D section of Photo Products Gevaert (now Agfa-Gevaert), Mortsel, Belgium. Assignment: Solvent recovery in the Film Sub-layer Division, august 1963
- Politecnico di Milano (Istituto di Chimica Industriale), Milan, Italy. Collaborator of Prof. Pasquon and Prof. Dente. Assignment: computer simulation of free radical mechanisms in the pyrolysis of hydrocarbons, May-June 1971.

He was full professor at the Vrije Universiteit Brussel (VUB) until 2008, since 2002 emeritus.

## Professional Background

- 1964-1966: Preparation of Ph.D. work, with a grant from I.W.O.N.L.
- 1968-1970: Assistant and First Assistant at the University of Ghent
- 1970-1976: Junior Professor V.U.B. ("Docent")
- 1976-2002: Full Professor at the Free University of Brussels (V.U.B.)
- Part-time professor at: The Catholic University of Louvain (1976-85) / The University of Antwerp (1982-97) / The Technical University of Delft – The Netherlands / The Bosphorus (formerly American) University of Istanbul – Turkey - Teaching a NATO course on environment and waste management / The Technical University of Surabaya – Indonesia / The Technical University of Sofia – Bulgaria / Universidad San Simon de Cochabamba - Bolivia / Universität Essen - Deutschland
- External Examiner Eötvös Universiteit/Technical University Budapest
- TEMPUS Teaching: environment and waste management
- October 1, 2002 Emeritus status; with still some assignments (Safety, Human Ecology VUB) and Research work in the Field of Dioxins
- 2001-2002: 5 months assignment as a guest professor
- 2002-2003: 5 months assignment as a guest professor both at Tohoku University, Sendai, Japan
- 1988-1989: Principal Advisor of the Minister of Environment, Housing and Industrial Plant, (1 June 1988 - 11 July 1989)
- He lectured at numerous universities: Ankara (Turkey), Antwerp (B), Cochabamba (Bolivia), TU Delft (NL), Uni Essen-Duisburg (D), Ghent (B), Hasselt (B), Louvain (B), Sofia (Bulgaria), Surabaya (Indonesia), and was in 2002 and 2003 Invited Professor at the Tohoku University of Sendai (Japan). Since 2008 he is Yongqian Professor and at present Pao Yue-Kong Professor at Zhejiang University (Hangzhou, P. R. of China).

## Prices and Awards

- First Students Prize - Royal Association of Flemish Engineers for the M.Sc. Thesis on "The catalytic alkylation of toluene", 1965
- Robert De Keyser Award, donated by Shell Belgium, for the Ph.D Thesis on "The thermal cracking of hydrocarbons", 1967
- Mentoring-award of the Körber Foundation, Hamburg, 1988
- Recycling Award, donated by the Coca-Cola Foundation, for the Research Work on Activated Carbon Production from Wastes, 1989

Managing editor:
Werner Bidlingmaier
email: werner.bidlingmaier@konsolan.de

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- Award from Tokyo Metropolitan Government, 1977
- Award from Mendeleyev Institute, Moscow, 1988
- Award from Kyushu University, Fukuoka, 1999

### Major Research themes

- Thermal conversion of waste, including biomass, plastics, rubber, hazardous waste
- Air Pollution Problems, Dioxins, Products of Incomplete Combustion, arising from Thermal & Metallurgical Units.

### Scientific and Technical activities

- 70 publications in scientific magazines
- 180 papers (e.g. conferences, seminars, ...)
- 20 articles in books and other works
- 600 advising reports
- 130 lectures in more than 30 countries

There are about 80 contributions in the field of dioxins, one of his favourite research areas

### Acts or Acted as an Expert for:

- The World Health Organisation (W.H.O.), in Germany, Turkey, and Malta United Nations Organisation for Industrial Development (UNIDO) in the P.R. of Romania, the P.R. of China
- E.C. (D.G. VIII, XI, XII and XVII) in England, France, Republic of Guinea-Conakry, Sierra Leone
- The Belgian General Agency for Cooperation of Development, Zaire (now Congo), Indonesia
- The Council of the Prosecution during the Dioxin crisis in Belgium - May 1999, with testing at The Japanese academic an industrial world

Led projects and assignments in a lot of countries, e.g. the E.U., USA, Algeria, Australia (Tasmania), Bolivia, Canada, China, Estonia, Guinea (Conakry), Indonesia, Kyrgyzstan, Rumania, Sierra Leone, Turkey, Venezuela, Zaire, and many others.

### Project Coordination

Prof. Buekens successfully coordinated three EU projects:

- Treatment of Fly ash for sound material utilization, Upcycle, EC Project n° ENV4-CT95-0085, (1996-1999)
- Chemical (pyrolytic) and mechanical recycling of plastics and composites, Cycleplast, EC Project n° PL964158, (1997-2000), with as partner Academy of Sciences of the Czech Republic and Hungarian Academy of Sciences
- Minimization of dioxins in the thermal industrial processes : mechanisms, monitoring, abatement, Minidip, EC, Project n° PL970492, (1998-2001)

and participated in Haloclean. He currently acts as Principal Technical Advisor (PTA) on a number of E.U. research projects.

### Member of Scientific Societies

- Royal Association of Flemish Engineers (K.V.I.V.)
- Flemish Chemical Association (V.C.V.)
- Royal Dutch Chemical Association (K.N.C.V.)
- Belgian Petroleum Institute (B.P.I.)
- International Association for Waste Pollution Research (I.A.W.P.R.C.)
- Institution of Incinerator Engineers
- The Flemish Chapter of the Filtration Society

Since 1976 he acted as an Environmental Consultant for the European Union (D.G. VIII, XI, XII, XVII), for UNIDO and WHO, and as an Advisor to T.N.O. (NL), Forschungszentrum Karlsruhe (D), and VITO (B). For 25 years, he advised the major industrial Belgian Bank (NMKN/SNCI) and conducted more than 600 audits of enterprise.

Main activities are in waste management, thermal and catalytic processes, and flue gas cleaning, with emphasis on heavy metals, dioxins, and other semi-volatiles. He coordinated diverse national and international E.U.-research projects (Acronyms Cycleplast, Upcycle, and Minidip) and participated in many others worldwide.

He is author of a book on Waste Management (1975), edited several books, countless Seminar Proceedings (>50), and a large Technical Encyclopedia (10,000 pages) and authored numerous scientific publications in refereed journals and more than 150 presentations at international congresses. He is a member of Editorial Boards for different journals and book series.

He played a role at the foundation of the Flemish Waste Management Authority O.V.A.M., of a hazardous waste enterprise INDAVER, and of the Brussels Environmental Protection Agency B.I.M./I.B.G.E. He was principal ministerial advisor in Brussels for matters regarding Environment, Housing, and Classified Enterprise (1988/89). He was a Member of the Board of the Belgian Consumer Association and of Conseur, grouping more than 1.5 million members in Belgium, Brazil, Italy, Portugal, and Spain.

He used to be licensed expert for conducting Environmental Impact Assessments (Air, Water, Soil) and Safety Studies regarding large accidents (Seveso Directive).

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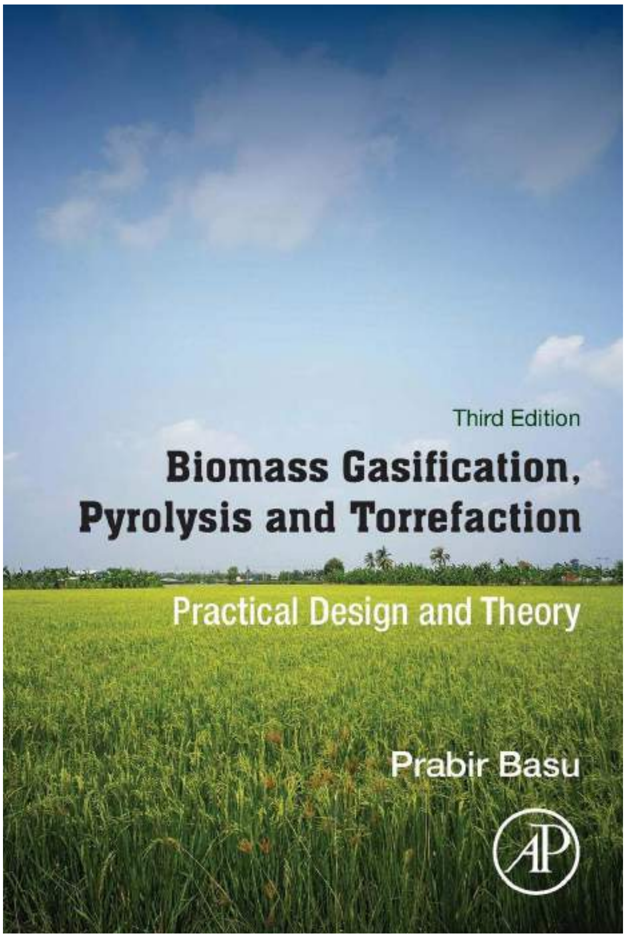
CISA

# BOOKS REVIEW




## BIOMASS GASIFICATION, PYROLYSIS AND TORREFACTION - PRACTICAL DESIGN AND THEORY Third edition

**Edited by Prabir Basu**

The increasing costs of adverse effects caused by accelerated climate change are compelling the entire economy to rely on processes capable of minimizing GHG emissions. In recent decades, the energy-production sector has shifted from a fossil fuel-based production towards processes based on renewable resources. Although this issue has only recently been addressed with any degree of urgency, primary solutions to the problem have been available since long before the industrial revolution. Among these, biomass thermal conversion is indicated as playing a key role in the so-called energy transition, being acknowledged by the scientific community as a carbon-neutral process for energy production.

In particular, torrefaction is aimed at the production of a solid product (known as “torrefied biomass” or generally as biochar), characterized by high energy density and often reused as solid fuel in co-firing processes with coal. Conversely, the main objective of the pyrolytic process is to maximize the production of a liquid product (i.e. Biooil), consisting of a mixture of water and complex hydrocarbons, used for fuel or chemical production. However, both can be considered as steps of the same biomass heating process in an oxygen-restricted environment. The gasification process moreover is defined as an additional step that makes use of a gasifying medium -such as steam, air or oxygen- to obtain gaseous products characterized by exploitable heating value. Further, when the biomass undergoing gasification features a high water content (e.g. algae or raw sewage), the use of supercritical water as conversion medium, in the so-called Hydrothermal Conversion of Biomass, may avoid extra costs incurred through the use of large amounts of energy in the preliminary drying phase.

However, although still relatively low, a growing number of industrial-scale installations are currently implementing this process. Therefore, the major challenge is to transfer the results already available in the pertinent literature to the experts involved with the aim of increasing awareness of the reliability of existing plant-design criteria.

The book “Biomass Gasification, Pyrolysis and Torrefaction” (third edition), edited by Prabir Basu, addresses this issue, revising and expanding on the previous versions by collecting recently updated information focused on design features and operational parameters of biomass conversion reactors and related equipment. The book includes 14 chapters and 3 appendices for a total of 564 pages.

Chapter 1 provides the reader with an introduction to the topic discussed, presenting the state of art with regard to biomass sources, an overview on the thermal conversion processes discussed and an interesting analysis on the environmental sustainability of the technologies mentioned. Chapter 2 proposes a brief focus on issues related to biomass management capable of influencing the economic viability of related full-scale projects throughout their life-cycle, e.g. local availability of biomass and capital costs of full-scale plant. Extensive knowledge of the physical-chemical characterization of available biomass is mandatory in determining reliable reactor design parameters: Chapter 3 lists the relevant biomass characteristics to be collected prior to the design step, including Thermal conductivity, Ignition Temperature, Heating value and Ash content.

Chapters 4 to 9 provide an in-depth delineation of the theoretical thermo-chemical principles, up-to-date technologies, design-criteria and possible yield-optimization of Torrefaction, Pyrolysis, Gasification and Hydrothermal

Managing editor:
Roberto Raga
email: roberto.raga@unipd.it

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Conversion reactors. In particular, Chapter 6 deals with the significant issue of tar residue management by illustrating how to minimize the related detrimental effects on reactor operations.

Chapter 10 represents the novelty of this edition. Authored by Dr. Bishnu Acharya, it provides insights into suitable treatment technologies (e.g. wet scrubbing or filtration) for undesirable products contained in the gas derived from pyrolizers and gasifiers (i.e. condensable gases, particulate matter, ammonia and sulfides).

There is currently widespread interest in partially replacing coal with biomass in existing large thermoelectrical installations with the aim of reducing related GHG emissions, thanks to the carbon-neutral nature of biomass. However, several major incompatibilities have been highlighted in previous or ongoing industrial trials. Chapter 11 discusses the potentialities and issues of the so-called co-firing process in coal combustion plants, involving both raw and torrefied biomass.

Products derived from the thermo-chemical conversion of biomass can also be exploited as chemical feedstocks, thanks to a growing market demand determined by the current shift of manufacturing industries and transportation sectors towards "green-renewable chemicals" and "green-fuels". The theories underpinning the processes involved in converting gasification products and pyrolytic biooils into synthetic fuels and chemicals are introduced in Chapter 12.

Chapter 13 discusses the best available options for auxiliary reactor equipment to be used in the storage, handling and feeding of solid biomass, and to fine tune industrial plant design schemes. Finally, Chapter 14 provides an important contribution by presenting the most suitable analytical techniques to obtain data relating to the chemical-physical features of biomass, as listed in Chapter 3.

To facilitate the readers comprehension three appendixes are provided at the end of the book: Appendix A contains biomass definitions, Appendix B supplies values for constants cited throughout the book together with tables for unit conversion and Appendix C summarizes data tabularly for use in progressing with reactor design.

Briefly, the book provides a detailed but user-friendly guidance of the state of art of biomass thermal conversion processes, namely, torrefaction, pyrolysis and gasification. The chapters guide readers through the book, illustrating the theoretical principles, operational parameters and equipment needed in the preliminary design of reactors. The book is mainly intended for professionals, such as sales engineers, project managers and scientists, to assist them in achieving the knowledge required to address the increasing market demand for full-scale technologies or to focus on developing mandatory improvements.

Giovanni Beggio
*University of Padova, Italy*
email: giovanni.beggio@phd.unipd.it

## ABOUT THE EDITOR

### Prabir Basu

*Dr. Prabir Basu is currently Professor in Mechanical Engineering Department and Head of Circulating Fluidized Bed Laboratory at Dalhousie University, Halifax. Further, he is founding President of Greenfield Research Incorporated and founder of Fluidized Bed Systems Limited, private companies specialized in biomass energy conversion systems and fluidized bed boilers, respectively. Working in the field of energy conversion and environmental studies for 30 years, his current research topics involve projects on the cutting edge of chemical looping gasification, torrefaction and biomass cofiring, amongst others. In addition to more than 200 research papers, he has authored seven monographs in emerging areas of energy and environment, some of which have been translated into Chinese, Italian and Persian.*

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# A PHOTO, A FACT, AN EMOTION


*"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 is one of hundreds of unregulated sites in Latin America."*

## "RECYCLING IN LEON"

Nicaragua

**Timothy Bouldry, United States**

Managing editor:
Elena Cossu
email: studio@arcoplan.it


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This photo was selected to participate in the second edition of Waste to Photo in 2017, the photo contest connected to the Sardinia Symposium, International Waste Management and Landfill Symposium organised by IWWG.

More than 100 photos were received to enter the second edition. The competition, open to all, aimed at recreating a scenario representing the global situation with regard to waste and landfill.

The panel of judges, which included members of the organizing and scientific teams, chose the winning photo based on how well it exemplified the entry requirements.

In addition, the most significant shots were used to set up a photographic exhibition to illustrate the differences, the contradictions, the difficulties and progresses encountered by this complicated issue in a series of contexts throughout the world, ranging from the developing countries to the more industrialized nations.

Mr. Timothy Bouldry won the first Prize of the 2017 Edition.

Elena Cossu
*Studio Arcoplan, Italy*
email: studio@arcoplan.it

## 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.

http://www.timothybouldry.com/

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