VOLUME 06 / June 2019

detritus

Multidisciplinary Journal for Waste Resources & Residues

Editor in Chief:
RAFFAELLO COSSU

an official journal of:
iwwg
international waste working group

CISA

ISSN 2611-4135 / ISBN 9788862650601
DETRITUS - Multidisciplinary Journal for Waste Resources & Residues

Legal head office: Cisa Publisher - Eurowaste Srl, Via Beato Pellegrino 23, 35137 Padova - Italy / www.cisapublisher.com

Graphics and layout: Elena Cossu, Anna Artuso - Studio Arcoplan, Padova / studio@arcoplan.it
Printed by Cleup, Padova, Italy
Front page photo credits: 'Sources of livelihood' courtesy of Nimai Chandra Ghosh, India

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Papers should be submitted online at https://mc04.manuscriptcentral.com/detritusjournal
Instructions to authors may be found at https://detritusjournal.com/guide-for-authors/
For any enquiries and information please contact the Editorial Office at editorialoffice@detritusjournal.com

Registered at the Court of Padova on March 13, 2018 with No. 2457

www.detritusjournal.com

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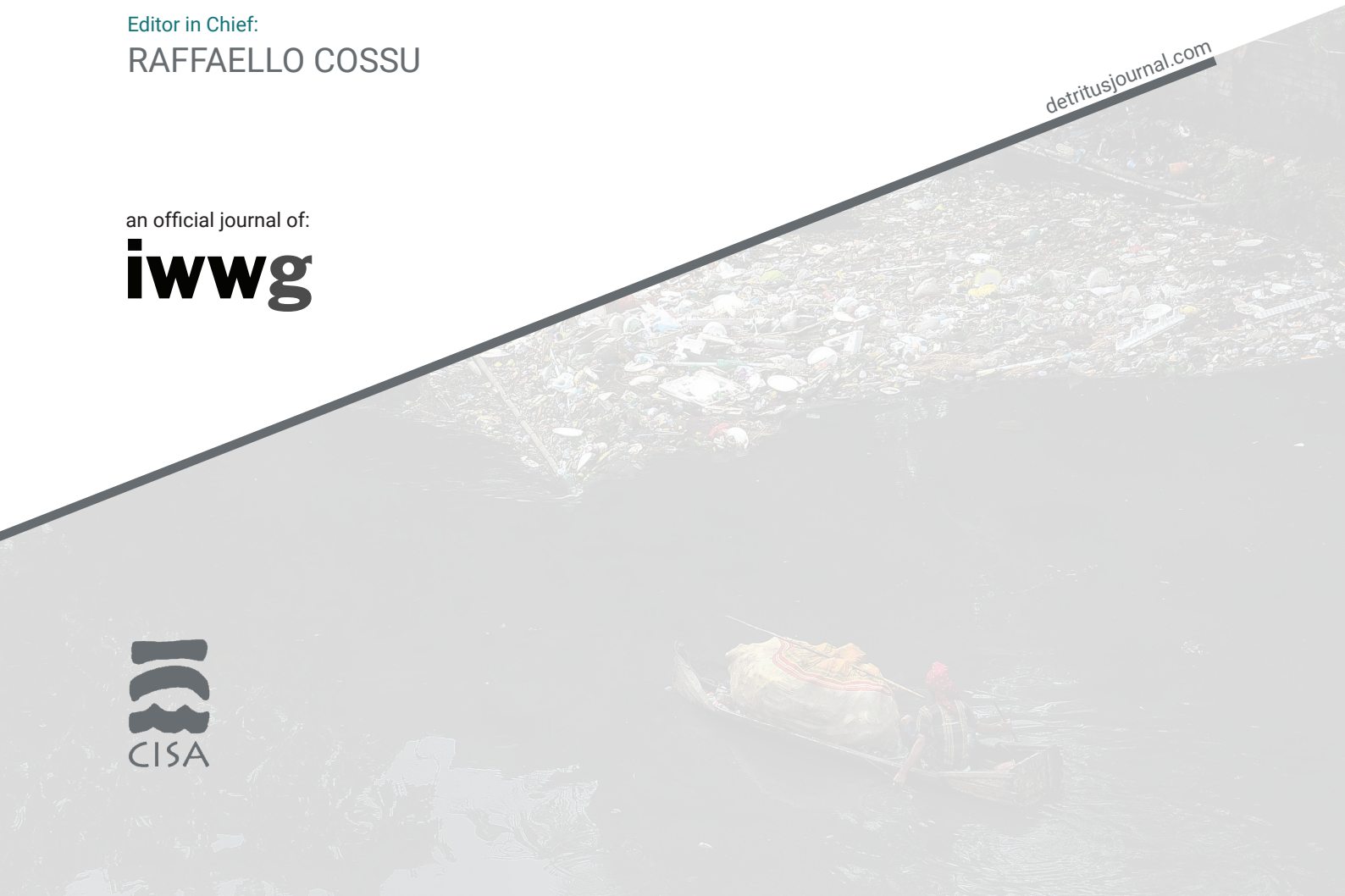

Detritus – Multidisciplinary Journal for Waste Resources and Residues – is aimed at extending the "waste" concept by opening up the field to other waste-related disciplines (e.g. earth science, applied microbiology, environmental science, architecture, art, law, etc.) welcoming strategic, review and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

SUSTAINABLE LOW-COST WASTE MANAGEMENT: LEARNING FROM AIRLINES

Waste management around the world is characterised by a very wide range of levels of technology and service efficiencies. Clearly, socio-economic conditions (such as financial resources, technical education, infrastructures, etc.) are the main issues at the basis of these differences, manifested not only between industrialised countries and developing countries (DCs) but also within the same administrative areas, as is the case of the European Community (World Bank, 2018; Eurostat, 2019).

However, many other factors contribute towards these differences, including:

- Population density. This has a marked effect on waste quantities and, consequently, collection programs and volumes required for treatment and disposal of the waste. Indeed, the latter represents the main driving force for incineration as a prevailing waste management option in countries such as Japan, Singapore, Switzerland and many others, (Cossu, 2009).
- Waste quality. All decisions, any criteria, and recycling programmes in waste management are heavily based on this factor. Aspects such as presence of hazardous substances, their concentrations, purity of waste fractions might originate different solutions.
- Market for recycled waste fractions. This factor is closely linked to the industrial and socio-economic organization of the specific geographic area and to the local demand for products and services.
- Specific local situations (climate, topography, infrastructure, land planning, culture, etc.).
- Regulations. These may be of a varying nature (recommendation, address, prescription, etc.) and are capable of creating marked differences between one country and another.

This picture is further negatively complicated by the transfer of inappropriate technologies from one country to another. Traditionally (and still persisting today!), this issue was confined to developing countries where the implementation of advanced technologies designed in (and for) industrialised countries may prove inappropriate for various reasons (complexity, maintenance, lack of professional education and skilled technicians, operational costs, infrastructures, etc.), as widely highlighted in the literature (i.a. Grossule and Lavagnolo, 2018). However, improper use of technologies is also encountered in industrialised countries. In this case, the main factors impeding the use of specific technologies include an inadequate maturity of the technology, a non-homogenous waste quality and operational costs (energy and staff), in addition to a series of regulatory and bureaucratic issues. As an example, management problems experienced at several pyrolysis and gasification plants operated in Europe, including the lack of an adequate commissioning phase and survey of local conditions, are widely acknowledged.

Moreover, the transfer of inappropriate technologies may contribute towards creating so-called “Cathedrals in the desert”, i.e. oversized facilities which are disconnected from the local reality, uneconomical, useless and frequently totally abandoned.

In numerical terms, more than 50% of global MSW production is still dumped or poorly landfilled, while the rest is treated using a series of different technologies (sanitary landfilling, recycling, anaerobic and/or aerobic stabilization, etc.) (World Bank, 2018), some of which may prove to be considerably complex and expensive. It was Laila Iskandar, working with the poor Zabbaleen recycling communities in Cairo, who famously said that, “waste management is far too important to be left to engineers; they build facilities which look like 4-star hotels”.

Despite this inhomogeneous scenario in global waste treatment, the aims and objectives of a modern waste management strategy tend to align and coincide throughout all corners of the world.

This indeed represents the positive result achieved by an impressive growing globalization and consequent diffusion of culture and science, supported by the Internet, the media, conferences, scientific journals, common publishing targets in academic career, and exchange of scholars and students.

The aims and objectives of a modern waste management strategy can be summarized as follows:

- Industrial production with minimisation of waste generation by contrasting planned obsolescence, avoiding disposable goods and extending producer responsibility;
- Design and production of goods which promote reuse and facilitate recovery and recycling;
- Source segregation and reuse of waste fractions;
- Environmentally-sound waste collection programmes;
- Optimisation of consumption and recovery of energy and material resources from unavoidable waste;
- Sustainable management of recycling residues with

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Detritus / Volume 06 - 2019 / pages 1-3
https://doi.org/10.31025/2611-4135/2019.13818

control of contaminants and hazardous substances (this aspect is frequently underestimated in circular economy strategies);
- Adoption of a combination of technologies to synergise advantages (thermal treatment for combustibles, biological treatment for putrescibles, stabilization of mobile contaminants and sustainable landfill sinking);
- Control of short- and long-term emissions, prevention of diffuse emissions and control of greenhouse gasses (GHGs);
- Minimisation of health risks while paying strong attention to the public opinion and perception;
- Scientific monitoring of ecotoxicological effects arising from WM.

Based on the previously illustrated discrepancy between the inconsistent global WM scenario and the common views in modern WM strategies, in order to progress from the fictitious to reality, the following needs should be addressed:

- Increase in access worldwide to an appropriate waste management system;
- Pursuit of the aims and objectives of a modern waste management system by adopting affordable low cost solutions, with minimal expenditure of energy and material resources.

By successfully fulfilling these needs it may seem as though you are squaring the circle.

However.... airlines have already done something similar!

In the not so distant past, flying was a privilege reserved for the wealthy. The availability however of fast-moving transport solutions represented a common interest for an increasing number of individuals.

It could indeed be argued that an identical discrepancy is encountered in waste management globally!

Of course, nowadays a lot more people can afford to travel by plane at a reasonable cost in safe conditions. The way in which this has been achieved should be an inspiration for the waste management world, merely in terms of analogy. Consequently, indirect disadvantages linked to the fact that transport is the fastest growing source of greenhouse gas emissions in the world, and that airline travel is a major part of this increase, are not considered here.

An overview of the main reasons underlying this success, focusing mainly on European low cost airlines which have successfully developed budget flight models, is given in Table1.

All features are substantially aimed at saving time and cutting costs, while at the same time guaranteeing rigorous safety conditions.

Leaving behind the airline metaphor, the following list of possible options could be taken into account for the purpose of turning solid waste management into a low-cost efficient system:

a) Any decision in WM should be based on a thorough and updated knowledge of waste quality variation in space and time; incredibly, this aspect is often neglected, resulting in inappropriate solutions and related costs;
b) Flexible strategies linked to the local situation (e.g. refraining from conducting source segregation and separate collection of a specific fraction in the absence of an end user at a convenient distance);
c) Recycling programs should not defer to moralistic principles but should rather be based on urban mining concepts (recovery of resources should be reliable, realistic, affordable, with no demagoguery, economically and environmentally convenient);
d) Separate collection should not strive to achieve percentages in terms of amount of collected materials but rather in terms of quality of recycled material (collect less but of a better quality);
e) Organised involvement of the informal sector, associations, NGOs, etc.;
f) Simple technologies of proven efficiency should be preferred;
g) Technologies should be suited to the specific local conditions;
h) The same technology should be implemented throughout a given geographical area or country with the aim of saving on maintenance costs (spare parts supply, staff training, etc.);
i) In some specific situations the acquisition of services provided by experienced enterprises might be preferred over the direct acquisition and operating of facilities;
j) The so-called "Blue solutions" should be applied wherever possible, based on the principle whereby there is no need to spend/invest more to protect the environment, but rather lessons should be learned from the environment and from what nature has already created in order to establish new business and social capital;
k) A holistic approach should be adopted in spreading resources among the different WM steps (collection, transport, treatment, disposal);
l) Integrated approach to WM technologies with no ideological preclusion (shrewd combination of recycling, landfilling and thermal treatment) ;
m) The convenience of material suppliers should be assessed in terms of transportation (zero km, repercussion on the community), use of resources (lower production of CO^2, renewable energy, possibility of constant supply) and economic impact;
n) Economic return should be ensured through synergies with other economic/social activities (informal sector, recycling, reuse, etc);
o) Particular focus should be placed on the recycling of putrescible fractions prior to landfilling. Biological stabilization of residual putrescibles by in situ treatment should be opted for over expensive mechanical-biological off-site pre-treatment;
p) Landfill technologies should aim to drastically reduce the abuse of expensive geosynthetics, by substituting these with equivalent low-cost products (natural materials, suitable residues, etc.) when conveniently available locally;
q) Following traditional biological treatment, there is no need to remove residual COD, mainly made up of humic substances, from the treated MSW leachate. Requirements to comply with discharge standards set below

TABLE 1: Overview of common features in low cost airline models and in the perspective of a low cost waste management system. LCC= Low cost carrier.

Features	Adopted measures by LCC	Potential measures in WM
No luxury or high cost items	• No video entertainment, thus no TV set and no operation of a central audio or video station	a), b), c), f), g), k), l), o), p), q), r)
Wise spending strategies	• Well-proportioned fleet • Bulk buying of same model of aircraft • No frills on board	a), b), c), d), g), h), k), l), m), o), p), r), s), v), w)
Simplified and standardized technical solutions	Same reliable and well proven aircraft models: • Easy management and maintenance (professional staff are trained on the same vehicle) • More convenient supply and storage of spare-parts • Increased crew flexibility	c), f), j), h), o), q), t), u)
Simplified operation	• Non-reclining seats (cheaper to buy and maintain • No back pockets (less time for cleaning)	d), f), i), h), m), q), r), s), t)
Staff saving	• Young motivated staff • Simplified training scheme (same aircraft model) • Multi-tasking staff	e), h), i), k), n), t), u)
Extra revenue generation	• No free on-board services • Some companies offer lottery tickets • Separate fees for checked-in luggage and extra bags on board • Payment for seat reservation	b), d), e), j), n)
Siting	Small airports: • Low fees • High negotiation power	h), g), k), m), o), s), v), w)
Energy saving	Young aircraft fleets • Baggage weight restrictions	c), d), j), m), q)
Intense use of the facilities	• Aircraft are used almost non-stop with rapid changeover times • Aircrafts return to the home hangar • Overnight maintenance	h), i), o), u), k)
Minimum overhead	• Fuel bought in favourable market periods • Direct online booking only	j), m), n), p)
Time management	• Every effort is made to reduce operation time • On time flights promote the company image	b), h), i), m), o), p), t), u), v), w)
Safety	• High safety records (money saving and good image)	i), j), m), t), u), v), w)

150 mg/L for COD, generally based on Reverse Osmosis, should not necessarily be adopted;

r) Waste management should not be overregulated (as occurs increasingly in numerous industrialised countries) as this may represent an obstacle to a virtuous waste management strategy, in both economic and technical terms;

s) Regulations should be flexible, open to significant innovative scientific development and compatible with specific local situations;

t) Science-driven educational WM programs in schools and universities should be increased, accessible to all, including local administrators;

u) Standardised and simplified operational and maintenance manuals should be provided to all technical staff;

v) An organised reasonable involvement of stakeholders in taking decisions prior to implementation of WM strategies might avoid costly opposition and protests afterwards;

w) Communication tools aimed at contrasting potentially misleading fake news (possibly resulting in unnecessary opposition by the public and related costs) should be developed.

To conclude, low cost strategies do not necessarily imply a reduced performance in protecting the environment and the public health; they should however represent a cost-effective solution intended to extend access of the populations worldwide to sustainable waste management systems.

Squaring the circle? Prepare for take-off!!!

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AT HOME IN AN UNHOMELY WORLD: ON LIVING WITH WASTE

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Article Info:
Received:
14 January 2019
Revised:
10 June 2019
Accepted:
17 June 2019
Available online:
28 June 2019

Keywords:
Hauntology
Waste
Cohabitation
Das Unheimliche
Hospitality
Oikos

ABSTRACT

Modern-day waste, such as the microplastics in the water, particulate matter in the air and chemical waste in the soil, distorts notions of inner and outer, of familiarity and strangeness, of own and other, and turns our world into an unhomely (uncanny) place. This paper explores what it means to live with waste instead of trying to make it go away. When we explore the ontology of waste, we find that waste is never unambiguously (in the) present and invites us to take "being" as haunted and explore a "hauntology" (Jaques Derrida) of waste. This hauntology refers to being as ecological being or "being ecological" (Timothy Morton) and invites us to inquire after the "eco" in ecology: the Greeks *oikos* (home). When, as this paper suggests, we take cohabitation as a starting point, it becomes unclear who is the host and who has come to visit. Whose home takes central stage? And whose world? This paper argues we need to let go of an overarching concept of "world" and instead become familiar with (our) "habitat" and in so doing open it up to the non-humans we share our home with. Sticking with the metaphor of (un)home(liness) this paper argues that our house is a haunted house and explores ways in which we can become hospitable with these (unwelcome) guests.

Long ago, this violent planet of radioactive rock had learned to become home.
Atlas had loved the earth; the crumble of the soil between his fingers, the budding of spring, the slow fruit of autumn. Change.
Now the earth changed as Atlas had stayed still, feeling the tilted axis rotate against his shoulder blades. All his strength was focused into holding up the world. He hardly knew what movement was any more. No matter that he shifted slightly for comfort. The monstreous weight decided on everything.
Why?
Why not just put it down?

Jeannette Winterson in Weight (2005)

1. INTRODUCTION

"This is where we should start feeling at home," philosopher Slavoj Žižek remarks as he walks around a quite unremarkable garbage dump in the documentary *Examined Life* (2008). The porn magazine he finds there, the broken-down refrigerator with the orange juice carton in it, according to Žižek these are all things we should not disavow or try to simply make go "away", but feel at home with. This approach to waste is very different from the dominant discourse nowadays, in which waste is identified in terms of waste management and framed as an object of manageable sustainability (Valenzuela and Böhm, 2017). As manageable material waste is either taken as something to be treated technologically and be incinerated, landfilled or anaerobically digested, or as something to be re-used, recycled and become a resource once again (Gregson and Crang, 2010). But waste is unruly, it defies managerial control and "bites back" (Reno, 2015, 564) and in so doing reminds us that we are never fully in control. The microplastics that are roaming the earth these days are a constant reminder of this — we did not intend that material to be where it is (moreover, we did not intend that material to be in the first place) and they altogether escape our managerial grasp. In so doing they remind us of our "being ecological" (Morton, 2018). Or as Josh Reno puts it: 'as they circulate and deform, wastes mix with people and places, with which they mutually transform or become together' (2015, 561). When Žižek urges us to feel at home with waste, he urges us to explore the possibility of this being-with or living-with waste. Instead of trying to keep the bad things out, we should try and find a way of living with them.

In this paper I will explore what this living-with might entail. First, I will inquire into the ontology of waste and ask what it is and how it is. I will show that waste muddies our sense of being and of being present and in doing so invites us to explore a hauntology of waste, that is, to conceive of

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https://doi.org/10.31025/2611-4135/2019.13820

"being" as haunted. I will then argue that hauntology refers to being as ecological being or "being ecological" (Morton, 2013, 2017 and 2018) and urges us to inquire after the "eco" in ecology: the Greeks *oikos* (home). When we take cohabitation as a starting point, it becomes unclear who is the host and who has come to visit. Whose home takes central stage? And whose world? I will subsequently argue that we need to let go of an overarching concept of "world" and instead become familiar with (our) "habitat". In so doing we open up our world/habitat/home to the non-humans we are cohabiting with and try and find ways to become hospitable with these (unwelcome) guests.

2. THE ONTOLOGY OF WASTE

What is this thing called waste that we have to learn to live with? As the scholarship in waste grows, so does the variety of answers to the question "what is waste?" We should also keep in mind that the conceptualization of waste is predominantly based in and on Global North contexts (Bell, 2018) in which waste is usually taken and kept "out of sight", whereas in the Global South waste is very much "in sight" (Davies, 2019) and people are used (and forced) to living with waste, which leads to different conceptualisations (Millington and Lawhon, 2018). There seems, however, to be an agreement on the fact that the answer depends on why the question is asked (Moore, 2012) and who's doing the asking (Reno, 2015). There also seems to be an agreement on the fact that what we define as waste varies between societies, places and times (Divoudi, 2009).

When we ask what waste is, we're inquiring into the ontology of waste. Although we can easily affirm that there is such a thing as waste — the ontological question — it is less easy to determine how things become waste and how we should distinguish waste from non-waste. Waste comes to be by having been something else (Viney, 2014) and comes to be in and through our relating to it — there is nothing essential to these things we call waste (Kennedy, 2007). As I've argued in more detail elsewhere, when we inquire into the ontology of waste we find that waste muddies our sense of being and of being present and invites us to explore a "hauntology" of waste. I take the concept of hauntology from Jacques Derrida (2010 [1993]) who uses the figure of the specter to problematize the assumption that things are unambiguous spatially or temporally present and develops an ambiguous ontology, a paradoxical state of being and non-being, that is, a hauntology. This hauntology allows us to think of being as haunted (Doeland, 2019).

Where does this take us in relation to waste? For a long time, waste has been taken as something to be disposed off. Whereas in pre-industrial times everything was remade and used again, with the rise of consumer culture we needed to learn to waste (Strasser, 1999) and came in need of what Philip Slater (1970 [1990]) discerned as the "toilet assumption." He defines this assumption as the implicit belief that waste can be simply "flushed away" and that 'unwanted matter, unwanted difficulties and unwanted complexities, and undesirable obstacles will disappear if they are removed from our immediate field of vision.' (Slater 1990, 19). Out of sight, out of mind. More recently Timothy Morton has come up with a similar metaphor: the "ontological u-band" (2013). He remarks that when we flush the toilet, we imagine the U-bend taking away our waste into some "ontologically alien realm". We are, however, increasingly bedoming aware of the fact that there are no such things as ontological u-bands and that all these things we try and make go away come back to haunt us, either now or in the future. When we act as if an oil spill is no big deal, assuming the ocean is so big it will water it down and clean it up for us, as if the 99% of the plastic produced that we cannot account for (Cózer et al., 2014) is really gone, as if the nuclear waste and highly toxic waste that remains from the waste-to-energy process is savely tucked under ground, we assume an "away" or a "beyond", a spatially and/or temporally ontologically alien realm that will take care of it.

When we inquire into the ontology of waste we are, then, refered to ecology — the relationship between people, plants, animals and minerals that teaches us that there is no "away" and that everything goes somewhere. Or as Morton puts it: 'thinking ecologically 'isn't simply pushing preformatted pieces around, thought meets specters, which is to say, beings whose ontological status is profoundly and irreducibly ambiguous. To encounter an ecological entity is to be *haunted*. Something is already there, before I think it' (2017, 64, emphasis in original) to which he adds later on that 'the specter is an *ontological* aspect of the structure of how things are' (2017, 96, emphasis in original). To be is to haunt and to be haunted.

Although in the past few decades the emphasis has moved away from waste as something that is to be disposed of and towards waste as a potential resource (Divoudi 2009, 131) we are still a long way from an ecological approach to waste. We might have become aware of our being "ecological beings" and that in taking out the trash of our respective homes we are dirtying and polluting that other home — our environment — but that hasn't changed much yet. Notwithstanding the (Western) mantra of "reduce, re-use, recycle", the global waste production is continuously increasing and could grow with 70% by 2015 as the World Economic Forum warned in 2018. And maybe that should't come as a big surprise. The current (European) discourse of "zero-waste circular economy" (Valenzuela and Böhm 2017, 26) underpins the logic of sustainable growth. It normalizes "green growth" and "sustainable growth" and purifies the connection between waste and unsustainability from all traces of ambivalence (Valenzuala and Böhm, 2017) and in doing so strips waste of its critical potential. When waste is taken as a resource, a manageable material and not as the necessary by-product of a system that thrives on wasting, or put differently, as a specter that reminds us of our ecological being, we will be haunted even more fervently. As research shows, when the stain of wastefulness is removed, we start to consume even more (Catlin and Wang, 2013). Trying to remove this stain is just another way of trying to make waste go "away". Instead, we should try and listen to what it has to say, as is usually the best way to deal with specters — the more one ignores them and tries to keep them out, the scarier the haunting

gets. What if we stop fighting the feeling of unhomeliness and try and be at home in and unhomely world? What if we accept that being has become unhomely, uncanny, *unheimlich*? And how to (co)inhibit this strange place?

3. BEING AT HOME (IN THE UNHOMELY)

Let us therefore return to Žižek urging us to become at home in a world of waste. What does it mean to be "at home" somewhere? When we ask what it means to be at home, we ask what it means to be ecological. The Greek root of "eco" is *oikos*, which refers to "house" or "household". The Indo-European root of "eco" is *woikos*, that in Latin became vicus, a neighborhood or settlement. When we inquire into our (being at) home, we are inquiring into ecology and the way in which we are related to our environment. Or as Moron puts it, oikos is about how the way in which each thing is inevitably inside another thing (Morton, 2013, 17). Just as I am a house for the numerous bacteria that inhabit my microbiome and at the moment also for the child that is developing inside me, — and, I might add, for the microplastics and chemicals that inhabit both me and the fetus (Houlihan, 2005; Wright and Kelly, 2017; Kontrick, 2018) — I myself am part of the house that is my town, my nation, the world. All these houses are increasingly becoming unhomely places — *unheimlich*, uncanny. But what kind of unhomeliness is this?

In his 1919 essay "Das Unheimliche" Sigmund Freud famously explores the notion of das Unheimliche, the uncanny, and argues that it is not so much the unfamiliar that gives rise to the uncanny, but the familiar. We experience something as uncanny when something we know for a moment appears strange to us, or when something unknown appears strangely familiar. The uncanny, in short, refers to the strangely familiar or the familiarly strange. As Anneleen Masschelein points out in her study of the uncanny (2011), in English texts the German unheimlich is not always translated in order to pun on the root Heim (home), which is lost in the common translation: uncanny. Why not translate it as unhomely? As Masschelein argues the concept of the unhomely is used mostly in architecture and postcolonial studies and has evolved to take on a slightly different meaning (2011, 13). I would say, however, that in this case "unhomely" is to be prefered to "uncanny", which is rooted in the Anglo-Saxon ken and refers to "knowledge" and to "understanding". For it is the (un)homeliness of waste that I'm struggling with here, not the (un)knowability of waste.

We experience something as unhomely when something we know for a moment seems strange to us, or when something unknown for a moment seems familiar. The unhomeliness of our modern-day waste-dominated world is, however, not so much about our feeling unhomely because we cannot keep our distance from the wastes that surround us — microplastics in the water, particulate matter in the air, etcetera — but in how these wastes refer to the unhomeliness of being itself, be it human, animal, vegatal or mineral being, that is, to our "being ecological".

If we are ecological beings, how can we clearly distinguish between what is inside and what is outside, between what is own and what is other? The unhomely is all about the impossibility to clearly distinguish the two, or as Nicholas Royle puts it in his study of the unhomely, about "the crisis of the proper" (2003). The unhomely disturbs what we call "proper" (from the Latin proprius, "own") and in doing so it is also 'a crisis of the natural, touching upon everything that one might have thought as "part of nature": one's own nature, human nature, the nature of the reality of the world.' (Royle, 2003, 1) As Žižek too remarks at the aforementioned garbage dump, in romanticizing "nature" and construing it as a harmonious, balanced totality (that we humans are destroying) we are distancing ourselves from it. Nature is then taken as a biblical place, a garden of Eden, and humans as fallen beings that should try and make things right by finding their roots in it again. If nature is construed as such, it is put at a distance and on a pedestal. If we take nature from its pedestal or, even beter, conceice of an "ecology without nature" (Morton, 2007), we take being to be ecological and allow for the unhomely.

But how much room should the unhomely be given? As Derrida points, if we let in too much, our home is taken over and ceases to be our home (Derrida, 2000). Derrida: 'it is less a question of […] trying to master the *Unheimliche* or the uncanny so that it becomes simply the familiar, than it is of the opposite movement. But this is not to say that one has to turn oneself over, bound hand and foot, to the *Unheimliche*, because I don't believe in that' (1985, 156, emphasis in original). As Derrida argues elsewhere, a hospitable place relies not on openness only, but also on mastery — there is always a master of the house. Derrida distinguishes "conditional hospitality" from "absolute hospitality" and in one of his famous deconstructive moves shows that absolute hospitality would mean that all that is "own", that is "proper" that defines our house as our house, would cease to exist if we let the other in unconditionally, that is, there would be no home left in which to invite the other in. The notion of being "at home" is an unstable category that is caught up in the dialectic of both "host" and "guest" and also in the dialectic of "friend" and "enemy". He points out that there is hostility in hospitality (which he dubs hostpitality), which refers to the Latin root *hostis*, that can mean both friend and enemy (Derrida, 2000). Who is at home and who comes to visit? And how to distinguish friend from foe? When inviting someone in and telling them to make themselves "at home" we don't invite them to do as they please, but to play by our rules. Derrida: 'I want to be a master at home, to be able to receive whomever I like there. Anyone who encroaches on my "at home," on my ipseity, on my power of hospitality, on my sovereignty as host, I start to regard as an undesirable foreigner, and virtually as an enemy. This enemy becomes a hostile subject.' (2000, 55) Hospitality, then, depends on selecting and filtering, on categorizing and choosing who to invite in and on what grounds. This does, however, mean that the tables can turn and the host can become a hostage. Derrida: 'It is always about answering for a dwelling place, for one's identity, one's space, one's limits, for the ethos as abode, habitation, house, hearth, family, home.' (2000, 150) Who is to be invited in and who's barred from entrance?

Although, as I argue here, we cannot neatly distinguish ourselves from (our) waste, this does not mean we

should refrain from setting boundaries. As Josh Reno remarks: 'when waste management is lacking, people and wastes mix in ways that threaten human life and dignity' (2015, 561). We should keep in mind, however — and that is a lesson we can learn from Derrida, whose work is all about deconstructing the metaphysics of the proper and showing 'how difference operates at the heart of identity' (Royle, 2003, 24) — that what we call our own, our proper, is always already expropriated, because it is not our own or of our own. There was never anything proper, it was always already of the other too.

In making waste, however, we make things our own and mark them as our own. Appropriation and dirtying (or wasting) seem to go hand in hand. Although all living beings pollute to lay claim to their habitats, the emphasis on appropriation since the Industrial Revolution has let to unprecedented pollution (Serres, 2011). As Michel Serres point out, the verb 'to have' has the same root in Latin as 'to inhabit' (namely habere). Our language echo's the relation between the nest (or home) and appropriation. Serres: 'between the living space and possession: I inhabit therefore I have.' (2011, 8) Serres would therefore not invite us to regard the earth as our home, since it is because we regard it as our home, our own, that we dirty it. Serres: 'by generalizing and globalizing dirt and so erasing the borders where polluting starts or stops, and hence appropriation, the right to property suddenly reaches an intolerable threshold and becomes literally unbearable.' (2011, 71) Serres suggests that instead of marking places as our own (and in so doing, dirtying them) we should become tenants. Tenants too, inhabit the places they dwell in, but don't mark them as their own. Serres: 'The world, which was properly a home, becomes a global rental, the Hotel for Humanity. We no longer own I; we only live there as tenants.' (2011, 72) It is dispossession of the world that Serres is after. The "natural contract", a contract between the earth and its inhabitants, as Serres envisioned it twenty years prior in his book *Le Contrat naturel* (1990) turns out to be a rental agreement. Is that the way to go? Should we let go of the idea that the world is our home and conceive of it as a hotel? Will that allow us to share it with ghosts, specters, and other (unwelcome) co-inhabiters? Let's not forget that hotel rooms are anonymous places that we can easily make our own because all the traces of those who've come before us have been carefully removed. While we stay there, we mark these rooms with our waste. We do not, however take responsibility for our waste, but trust the housekeeping will clean up after us and take it "away".

4. FROM WORLD TO HABITAT

It is said that the world is a stage. But who's stage? Who are the lead characters and who play a supporting role? Whose world takes center stage? These days we are learning that humans are not the only actors. The décor is acting up and what used to be behind the scenes has crawled onto the stage, challenging humans' leading role (Latour, 2018). This changes the script and suggests different dénouements. It turns out we are not surrounded by passive, dead matter, but by active, "vital matter" (Bennett, 2010) such as waste, which is starting to "force thought" (Hawkins, 2009). How does this change the stage — the world? Up until now I have stuck to the notion of (un)homeliness and steered clear of the concept of "world". Although I cannot do justice here to this key philosophical concept, I will give some remarks on how I think we can and should open up (the concept of) "world" to non-humans.

Philosophers have been driven not so much by the desire to be in the world, but to find a vantage point on the world as a whole (Gaston, 2013). Such a notion of world does not leave room for much cohabitation. As Sean Gaston argues, it is Derrida who allows us to think of the possibilities of a concept of world beyond the logic of containment and in doing so allows for cohabitation of the world (2013, xiii). Why not let go of the concept of world altogether? Derrida does indeed argue for the end of the world. Drawing on a line of poetry from Paul Celan — "Die Welt ist fort, ich muss dich tragen" [the world is gone, I must carry you], he argues that with the disappearance of the world "as such" every death becomes an "end of the world." (2017, 170) However, a concept of the world remains of importance to Derrida insofar it indicates a domain in which people and animals live and die together, 'a place of common habitat' (2017, 264). In doing so, Derrida takes us beyond the logic of containment and allows for a world that is shared by, but not identical to all. Also, "the world is gone, I must carry you" holds a promise, a responsibility, a commitment, that we might not want to let go off.

Morton, however, does urge us to let go of (the) world. He argues that world is a "fragile, aesthetic effect" that depends on foregrounds and backgrounds, that has ended now that global warming (among others) has revealed that 'what we took for a reliable world, was just a habitual pattern.' (2013, 102) The world is not a passive, inert place that we can have our way with, but an active, alive environment that does not bend to our will. Morton: 'when we look for the environment, what we find are discrete lifeforms, nonlife, and their relationships. No matter how hard we look, we won't find a container in which they all fit; in particular we won't find an umbrella that unifies them, such as world, environment, ecosystem, or even, astonishingly, Earth.' (2013, 129) We are surrounded by global warming and nuclear radiation, not a world. As Morton points out we prefer the world somewhere outside us, where we can admire and reflect on it. We don't like finding ourselves in it, where there is no "away" and things come crashing in on us.

What if we let go of (the) "world"? What concept would be able to replace it? Since we're looking for a concept that takes being to be ecological and steers clear of vantage points, I would suggest a concept that lies at the root of ecology: habitat. Whereas the concept of world is overarching per se — even when we make the world as small as our individual world, it still presupposes a birds-eye view of that particular world — the concept of habitat is specific to these things (both living ánd non-living) that inhabit it. Although in ecological research the term "habitat" is highly problematic, as the definition of habitat varies a lot (Hall, Krausman & Morrison, 2007) and is used correctly only 55% of the times (Hall, Krausman & Morrison, 2007; Kirk et al., 2018), in a theoretical context it works perfectly. Al-

though originally a fundamental concept in plant ecology only (Yapp, 1922) it has evolved into a concept that can describe virtually any location occupied by organisms (Kirk et al., 2018) and can, as I would suggest, also include the dwelling places of non-living things, or more succinctly put: include non-living things such as waste into the dwelling places of living things.

If we want to learn to live with waste, we have te learn to not try and close our habitats of, but share them with the other things — both living and non-living — that are already there. And let's not forget that even if we wanted to, we cannot keep our distance from waste — waste-things are hard to keep at bay. Both the waste that is landfilled (but leaks), that is incinerated (but leaves toxic ash and dioxins) and that is anaerobically digested (but not all of it) leaves remainders. There are, however, two types of waste that stand out in being hardest to live with and could be construed as the wastes of our times: plastic waste and nuclear waste. It is on living with these two wastes that I would like to turn to finally.

5. AT HOME WITH GHOSTS AND MONSTERS

As two recents documentaries reflect — *Into Eternity* (2010), that follows the construction of the radioactive waste repository in Onkalo, Finland and *Containment* (2015), that focusses on the communication of what is stored at the WIPP (Waste Isolation Pilot Plant) near Carlsbad, New Mexico to future generations—it is not easy to live with nuclear waste. How to make sure the waste is contained there? How to make sure that nuclear waste sites remain undisturbed (for at least 10.000 years)? How to make universal warning signs that will still be legible and understood after all these years? As Peter van Wyck points out in his study of nuclear waste burial and ecological threat, where nuclear waste is concerned there is no such thing as containment: '[n]uclear material stand in relation to their containment only very imperfectly — there is always leakage' (Van Wyck 2005, 19). This is not just a technical matter. As *Containment* shows we can never be sure that signs that we erect to spell "danger, don't dig!" might in a future times by future creatures be taken for an invitation to dig — "there are treasures here"!

We will have to keep knowledge alive about the danger of what is stored at these sites. Morton somewhat jokingly suggest that instead of storing our nuclear waste below ground and (accidently) forgetting about it, we should keep it out in the open and care for it there. Preferably in city centers, where we cannot avoid being aware of it and knowledge of its danger and conservation can be handed down through generations. Although not located in a city center, the Dutch nuclear waste disposal site COVRA (Centrale Organisatatie voor Radioactief Afval) is very visible. The building is painted a bright orange and open to visitors through educational programs and guided tours, that even give access to the faults in which the containers with radioactive waste are stored. In the end the radioactive waste stored there might indeed find its way into a city center. This waste disposal site was never meant as a permanent location and since it is located at the coastline and sealevels are rising ever more rapid (Bamber et al. 2019) it will have to be relocated a lot sooner than anticipated. Whatever we do with it, we should keep it in sight and in mind.

On medieval maps unknown territory was marked by monsters. Nowadays we seemingly have no uncharted territory left; there are no monsters on Google Maps. But there are monsters nonetheless. Nowadays these monsters are not supposed to unhabit parts of the world that we have not made our own yet, but are said to live in our waste and wasteland for example, such as the ever-growing mineral residues that are brought forth by mining processes (Ureta and Flores, 2018). Monsters are also found in the remains of material we did not only dig up, but also processed: oil-based plastic. To raise awareness for the ever-growing problem of plastic waste, plastic is regularly framed as monstrous, most recently in the Greenpeace's "Plastic Monster Ship Tour" (2019). Charactaristic for these projects is that the people involved — both in activist and in educational settings — gather plastic and gather together to make monsters out of plastic. Although these monsters are usually rather friendly looking, the idea is that they warn us and scare us of (Figure 1).

If we delve into the etymology of "monster" we learn that the roots of the English "monster" can be found in the Latin *monstrare* (to show) and *monere* (to warn). But how do they warn? And what for? Monsters tell us something about the times they arise in. As fantasy fiction writer China Miéville puts it in his "Theses on Monsters", history can be written in monsters and 'epochs throw up the monsters they need' (Miéville, 2012, 142). These monsters don't, however, speak for themselves. They need "decoding" (Miéville 2012). As Stephan T. Asma points out in the epilogue of his extensive study on monsters, it is no coincidence that he doesn't give a single definition of "monster". Rather, monsters are 'environmentally specific archetypes for a clan's central threat [that] appear and reappear in our stories and in our artwork because they help us (and helped our ancestors) navigate the dangers of our environment [and] provide us with a ritualized, rehearsable simulation of reality, a virtual way to represent the forces of nature, the threats from other animals, and the dangers of human social interaction' (Asma, 2009, 282). Nowadays we need not fear gigantic sea monsters lurking in the depths of waters unknown, but (among others) the plastics that we have let slip into these waters. The plastic monster makes us pay attention to life's "symbiotic entanglement across bodies" (Tsing et al., 2017). Plastic goes places, our bodies included. The danger, however, lies not so much in this monstrous entanglement itself, but in the denial of it. As the editors of *Arts of Living on a Damaged Planet* put it, 'our continued survival demands that we learn something about how best to live and die within the entanglements we have. We need both senses of monstrosity: entaglement as life and as danger.' (Tsing at al., 2017, 4).

How should we deal with these plastic monsters? As Latour points out, there is an important lesson to be learned from the most famous of monsters: Frankenstein's. Dr. Frankenstein's great mistake was not that he created a monster, but that he got scared of what he had created and

FIGURE 1: Plastic monster; photocredit: Marten van Dijl / Greenpeace.

abandoned his creature. As the monster tells Dr. Frankenstein when he meets his creator later on, he wasn't born a monster but became one when he was abandoned and left to his fate (Latour, 2011). If we take this cautionary tale to heart and ask wherein the monstrosity of plastic lies, we find that it is not so much in the material itself but in our lack of care for it. We have put a material in this world and then abandoned it.

Living in an unhomely world means living in a world of monsters. It means living in a world in which an ever-growing amount of material demands our care. As Latour puts it, we are living in a time of cohabitation, a "monstrous time" (Latour, 2005). The question should no longer be: can we flush you? Or: will you go away? Or: are you going to disappear soon? But: can we cohabitate with you? This time of cohabitation is a radical break with the time of progress and of succession. As Naomi Klein points, out the time of progress, that has been central to modernity, takes the world as a frontier of conquest rather than as a home (2014). Moreover, this future-oriented thinking relies on sacrificing in the present and in so doing allows for "zones of sacrifice" (Kuletz, 1998; Lerner, 2010; Klein, 2014) such as abandoned tarsand fields and mines, chemically polluted areas and other wastelands. The time of cohabitation on the other hand, demands that we trade the temporal for the spatial and care for these spaces. This "spatial turn" (Morin 2015) requires that we relate to the space in which we dwell and invites us to consider matters of openness to that space and of border patrol — or in other words: hospitality.

Do we invite waste in? Derrida would probably argue that we need to set boundaries and decide on who to invite in and who to keep out. Monsters teach us that these boundaries are porous and that it is indeed of the greatest importance that we learn something about how best to live and die within the entanglements we have. As Tsing points out, 'we need both senses of monstrosity: entanglement as life and as danger.' (2017, 4) But most of all, we should become familiar with all these different kinds of things that are made to waste and decide if we are wise to let them in or not. A Derridian approach to this consists in the paradoxical relation between conditional and unconditional hospitality. Paradoxical, because they push in opposite directions and cannot be sublated into one imperative. The ethical gesture requires we do not ask which side is the "good one," for the good is always on the other side (Morin, 2015). When I invite in unconditionally and with open arms, the good is on the side of the conditional and vice versa. Morin: 'I can never stand firm on the side of the good; rather, finding myself on one side, ethics is what pushes me to the other side indefinitely.' (2015, 33)

Back to Latour, who in his "parliament of things" (Latour, 1993) asks who are at the table, assembled, and then asks if and how those at the table can coinhabit. In Latour it is those who are already in the collective who decide if those outside it can or cannot enter. The meaning of Latour's "can we live together?" seems to be 'can we, who already

belong to the collective, live with you, who do not (yet) belong.' (Morin, 2015, 37) Morin suggests we complement Latour with a Derridean understanding of unconditional hospitality, that is, with an openness that, I would add, is reminiscent of Morton's notion of "ecological being". When we take cohabitation as a starting point it becomes unclear who is the host and who comes to visit. As Morton asks in the opening chapter of Humankind: 'Am I simply a vehicle for numerous bacteria that inhabit my microbiome? Or are they hosting me?' (2017, 1) Who knows, in time we might become dependent on the microplastics that are invading our bodies and that, like it or not, we have become entangled with — there is no way to keep the bad things out. Our house is a haunted house and we'd better make ourselves at home.

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SUSTAINABLE RESOURCE MANAGEMENT BY STUDENTS AT HIGHER EDUCATION INSTITUTIONS

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Article Info:
Received:
19 July 2018
Revised:
21 November 2018
Accepted:
17 January 2019
Available online:
23 April 2019

Keywords:
Circular Economy
Reuse
Higher education institutions
Key performance indicator
Waste management
Sustainability

ABSTRACT

'Fast fashion' and a 'take, make, dispose' economy are fueling people's accumulation of goods, and their willingness to throw these goods away. The end of each University year sees tens of thousands of students clear out their homes, and lives, and move on to pastures new, leaving behind a wake of unwanted clothing, kitchenware and bedding, amongst other 'throwaway' items. Sustainable waste management schemes at higher education institutions (HEIs) are becoming more prominent, and are taking the opportunity to generate positive outcomes from this problem that affects most university cities across the country. This study comprises of an extensive review of HEI reuse schemes as well as the collection of primary data during the planning and running of "Shift Your Stuff", an end-of-term reuse scheme run by Southampton University Students Union (SUSU). The study develops an existing protocol, designed to provide a consistent method of running and analysing an end of term re-use scheme, and refining it to create a methodology which could be easily transferred to other universities. We critically analyse the methods, protocol and key performance indicators (KPIs) which have been used in previous years, and report on the success of this year's project, in comparison to previous years. The KPIs continued to be an effective way of displaying the success of the scheme, with a 92.4% reuse rate – an improvement on the previous year, along with a host of social benefits including the donation of food and clothing to the homeless or deprived and the contribution to ground-breaking heart disease research undertaken by the British Heart Foundation (BHF). These benefits were brought about by the donations themselves, and by the £3,474 raised through the sale of donations. Although the total amount of donations was lower in comparison to previous years, the quality was substantially higher, with a lower percentage of donations having to be sent for disposal. We make a series of suggestions on how to improve the scheme in order make it more effective. For universities to reach their potential in terms of donations, it is recommended that the running of the scheme is handed over to a charitable organisation that is set up for this purpose.

1. INTRODUCTION

"Shift Your Stuff" is an annual scheme run by the Southampton University Students' Union (SUSU). The scheme involves distributing collection bags to students at the end of the summer term, and providing a service to collect or drop-off donations of unwanted items. The SUSU then redistributes them for reuse or resale by charities. As well as charitable benefits to the community, the scheme works to alleviate complaints by residents of the city of Southampton about the mess left by students in streets when they move out of accommodation at the end of the academic year. It also retains materials and resources in the circular economy, reducing pressure on resources and promoting reuse and recycling to the student community. During 2015/16, a protocol was developed to set out a recommended timescale of what should be done to execute the project, and when each task should be completed (Patel, 2016).

1.1 Consumerism and resource consumption

Current consumerism is driving pressure on resources across the globe; 'fast fashion' is reinforcing the 'take, make, dispose' linear economy, in which resources are used few times before being sent for final disposal. For example, so-called "fast fashion" is a system that produces fashionable items in short times in order to respond to rapidly changing consumer demands (Diop and Shaw, 2018; Cachon and Swinney, 2011). The global retail textile industry reached a value of over $1.2 trillion in 2014 and,

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Detritus / Volume 06 - 2019 / pages 11-24
https://doi.org/10.31025/2611-4135/2019.13813

as one of the world's principal industries, this sector impacts significantly upon the environment (Diop and Shaw, 2018; Resta & Dotti, 2015) It is a highly turbulent industry - with trends changing regularly and 'fashion seasons' being short (Bruce and Daly, 2006) and with shops bringing in new garments in every few weeks (Joy et al., 2012), the perfect conditions are created to encourage customers to shop regularly and accumulate goods quickly. Some products may only have a 'fashionable' life cycle of weeks (Sull and Turconi, 2008) resulting in many items (between one third to a half of all items sold (Bruce and Daly, 2006) having to be sold at mark-down prices. This is making low quality garments even more accessible, driving consumers away from high quality, longer-lived products.

There is a growing recognition that resources must be conserved and an increasing understanding of the need for circular economy behaviour in order to protect resources and the environment. For example, Diop and Shaw (2018) highlighted that it is crucial citizens view end-of-use clothing and textiles in terms of their value as a resource in adherence to the aims and principles of the waste hierarchy. Indeed, consumers are starting to resist consumerism - people are becoming more aware of the impact of their purchases and are making conscious decisions to move away from consumption and its environmental impacts (Joy et al., 2012; Kozinets and Handelman, 2004). In some developed countries, reuse, refurbishment and repair of "pre-loved" or "pre-owned" items is increasingly becoming a lifestyle choice for many people (Williams and Shaw, 2017). However, the cost of ethically produced, long-life products may not be a priority for all, especially students who may prioritise spending on other items or activities. People with environmental values may utilise other methods of reducing their environmental impact through the responsible disposal of their unwanted goods (Joung and Park-Poaps, 2013). This environmental consciousness, and understanding of the impacts of production and consumption, is driving an increase in public participation in 'reuse' schemes in communities around the world (Bianchi and Birtwistle, 2010).

Young people are the most concerned with buying 'trendy', cheap clothing (Morgan and Birtwistle, 2009). Within this group the demand for cheap clothing is high, and the availability and access to these goods means that it is increasingly easy to buy and accumulate large amounts of cheap, low quality goods. Add to this the stash of free promotional t-shirts a student will receive during their time at university, and you have students with wardrobes stocked with garments they feel little attachment to. It has been found that consumers feel a low sense of attachment to cheap items, and do not feel guilty about disposing of them (Birtwistle and Moore, 2007). This mind-set is especially present in student communities (Students and Staff of the Centre for Environmental Science, 2017). Due to the nature of their living situation, many students tend to purchase cheap items which only need to serve a purpose over a couple of years. At the point of purchase these items are often destined to be thrown away within three years, if they haven't already broken. So, at the end of the academic year large volumes of reusable goods are disposed of by students who are leaving university. Birtwistle and Moore (2007) found that consumers were more likely to donate these unwanted items to charity, if prompted to do so. Shift Your Stuff was designed to appeal to the student population and encourage responsible disposal by providing a prompt to encourage students to donate their unwanted items, instead of disposing of them. If students have accumulated large amounts of cheap clothing that they do not feel attached to, they are likely to dispose of it when they move out of their house rather than moving it with them. Shift Your Stuff can intervene and divert these goods from landfill redirecting them towards reuse or recycling. completed (Patel, 2016).

1.2 Waste issues and reuse schemes

At the end of the summer term in university towns in the United Kingdom (UK), tens of thousands of students will move out of their rented homes and in the process, generate large quantities of waste (Zhang et al, 2011). It is hard to miss the waste issues that are generated by the student move-out period. It is publicised in local newspapers and often through local news and radio - overflowing bins are in abundance, which can in turn generate issues with vermin, odours, and health and safety (Bristol Post, 2016, Waddington, 2016). These issues are put down to student clear outs and can cause tension between permanent residents and transient student populations (Keep Britain Tidy, 2013). Many residents claim that students have a disregard for their area because they may only be there for a year or two, and feel little pride in keeping the area looking nice. But students may justify their actions by claiming that waste disposal options are limited. Bin size restrictions can cause issues for houses of multiple occupancy (HMOs) with only one wheeled bin serving up to seven or so residents, a particular issue during periods of high waste generation such as during the move-out period (Williams and Cole, 2013; Gosling, 2016). Access to civic amenity (CA) sites may also be limited for many students due to the restricted transport options of a typical student, meaning that they may find it impractical to dispose of their waste elsewhere, especially if they have large quantities that cannot be carried on public transport (Williams and Taylor, 2004). These combined factors cumulate in streets containing overflowing bins and extra bags of waste on streets, conditions that can attract both vermin and scavengers, often resulting in waste being scattered across the streets of student dominated areas.

Waste management schemes in higher education institutions (HEIs) can play an important role in alleviating the problems associated with the student move out (Zhang et al, 2011). Reuse schemes such as Shift Your Stuff can divert student waste from disposal, creating space in bins and alleviating pressure on waste collection authorities and councils that have to handle the responsibility of clearing up the waste. The positive impact of end-of-term reuse schemes has been seen in Bristol, where litter complaints fell by 28% once the 'Bristol Big Give' reuse scheme began in the city (Bristol Post, 2016).

A review of reuse schemes has found that a kerbside collection can come with associated problems, with a possible issue being scavenging. Between the point when

bags are left out for collection by students and them being collected by the team working for the scheme, scavengers may tear open bags in search of items which may be beneficial to them. This action can cause health and safety problems, as well as being an unsightly nuisance to the community.

Charity reuse schemes can aid many issues: litter problems can be alleviated; resources conserved; social benefits generated; and deprived members of the community helped (Horne, 2000). These schemes are becoming increasingly common in HEIs. The British Heart Foundation 'Pack for Good' campaign currently has 80 partnering universities and has raised over £1.5 million through 2,200 tonnes of donations since the scheme launched in 2011 (BHF, 2016b).

1.3 Key performance indicators

Key performance indicators (KPIs) are a way to measure performance in a repeatable way which will produce results which can be reliably compared (Williams and Shaw, 2018). They are a straightforward way of measuring results and presenting results in both technical and non-technical ways (Del Borghi et al., 2009). Typical KPIs for waste management include strategies include weight diverted from landfill (WRAP, 2016) and carbon footprinting (ICAEW, 2017). Shift Your Stuff uses KPIs to measure the success of the scheme. These have been developed to produce consistent datasets year on year, making the results easily and reliably comparable. They are also an effective way to relay the success of the scheme to the public (Del Borghi et al., 2009) without having to give complicated figures. KPIs were developed for the 2015/16 Shift Your Stuff; variations on traditional waste management KPIs had to be created to be appropriate for a reuse scheme (Table 1). This paper considers the effectiveness of KPI use for quantifying the success of a reuse scheme.

2. METHODS

2.1 Desk study

A thorough review of reuse schemes at UK HEIs (Appendix 1) was conducted to gain an understanding of the characteristics of some of the most successful schemes and to see how they may be applied to Shift Your Stuff. Summaries of these findings can be seen in Appendix 2.

TABLE 1: Key performance indicators (KPIs) for Shift Your Stuff.

Key Performance Indicator	Description
Summary	Measures the percentage of all the collected donations that is sent for reuse.
Environmental	The weight of each material category which is reintroduced to the circular economy through the scheme.
Economic	The amount of money raised for charity from the resale of the items donated through the scheme
Social	A qualitative summary of the social benefits that are brought about by the donations, or the money raised by the resale of the donations.

A SWOT (strengths, weaknesses, opportunities and threats) analysis from Patel (2016) was used as a framework and, from the report and undertaking discussions with people who have been previously involved with the scheme, additional strengths, weaknesses, opportunities and threats were identified. The SWOT analysis was amended accordingly (Table 2).

2.2 Protocol development

Analysis of the existing Shift Your Stuff protocol (Patel, 2016; available on request via idw@soton.ac.uk) was undertaken by discussing the tasks and timescales with the Shift Your Stuff team. Each task on the protocol was assessed for necessity, appropriateness of the timings and suitability of the person assigned to carry it out. If tasks were deemed to need additions and/or alterations to timings then the suitable changes were agreed on by the Shift Your Stuff team and noted on the protocol.

As outlined in Section 1.3, there has been some focus on developing methods for measuring the impact of reuse (Castellani et al., 2015; WRAP, 2011), although it has been recognised that there are complexities when doing this (Fortuna & Diyamandogly, 2016; Alexander and Smaje, 2008). Whilst KPIs for recycling are well-established and widely used, data collection for reuse is challenging. There is a clear need for reuse-related KPIs so that the impacts of reuse can be measured, monitored and demonstrated over time. In this context, the key performance indicators developed by Patel (2016) were reviewed and where it was found that there were problems with the KPI data collection in the previous year, appropriate changes were made.

2.3 Collection, sorting and processing of donations

Donations could be made in two different ways. These were promoted via posters and blogs written by officers of the Southampton University Students' Union (see examples at: https://blogs.susu.org/officers/2016/05/24/shift-your-stuff-2016/ or https://blogs.susu.org/blog/2017/05/15/shift-your-stuff-2017-a-spring-clean-is-the-dream/).

The scheme offered drop-off points at University of Southampton campuses where students could deposit their donations in branded donation bins. These points were available for deposits from the end of May to the end of June. The second method which could be utilised to make a donation was kerbside collection. Kerbside collections took place on specific days in June 2017. Students were advised to leave donations in branded red bags outside their property on the morning of each collection. Employees from the city council then collected any red bags from kerbside in the student areas of the city delivered them to the sorting rooms at an identified campus (Highfield). Details on how to donate were printed on leaflets and handed out at University of Southampton campuses.

Volunteers were recruited to assist with sorting and processing the donations. Recruitment was undertaken via posters, word-of-mouth, social media, and by email to selected groups of student (e.g. students studying Environmental Science; see Appendix 3). Each bag of donations was opened and visually checked to ensure the contents were of one category; in cases where categories were

TABLE 2: SWOT analysis of the 2015/16 Shift Your Stuff. Original evaluation carried out immediately after the previous year's project is in shaded boxes, additional strengths, weaknesses, opportunities and threats identified prior to this year's project are unshaded.

Strengths	Weakness
Having items pre-sorted into categories reduced sorting time required	Inconsistent data (can't compare years), digital scales were lost before the second round of sorting.
The quality of the items was increased (less waste produced than the previous year), potentially because of the categories.	Some bags picked up by Southampton City Council from the streets didn't have stickers on and contained a mixture of categories.
The charities agreed in advance what they would take, and came to collect it.	The quality of sorting was dependent on the quantity of bags that needed sorting at the time.
Less storage space was needed as the charities collected soon after items had been sorted.	If there were a lot of bags, the sorting became less thorough.
By taking photographs, data could be collected and stored in a quick way.	The marketing team put a picture of an electrical item on the category stickers meaning a lot of electricals were donated even though Union Southampton had asked for students not to donate these.
Data collection more consistent across the sorting days than the previous year (though still some issues).	The kitchen utensils category was quite fragile so the items would easily break if not handled correctly.
Baseline data started to be collected from the KPIs	A technological issue for storage of the photographs resulting in the photographs from the first round of sorting being lost.
	Total weight was lower than the previous year.
Student's unwanted items were diverted from final disposal for reuse.	Limited data- only the number of bags and their weight.
Waste issues on the streets were reduced.	Inadequate staffing/volunteering.
Many students dropped items off at drop off points, or left them for kerbside collection.	Relatively low participation in comparison to student population.
The scheme was supported by SCC and the BHF.	Low number of donations to the first collection, possibly arranged too early.
The scheme set a good example for resource management in the city	Some charities were unhappy with donation quality/sorting, Communications with charities could have been better to have a good idea of what they wanted and what was required of them.
	No data on the demographics of the participants.
Opportunities	**Threats**
Unsure of the number of students who took part in the project	Not all of the charities were happy with the items donated, for example, some items were not appropriate for the function of their charity.
There was some overlap with the categories and some confusion of what items should go into which category	Some issue with the number and reliability of volunteers
Results don't include halls of residence donations (as some other universities do); inclusion of this could improve results.	Poor marketing of the scheme- if participation is too low, the scheme may not be worthwhile running

co-mingled the items would be separated out into single category bags (Figure 1). At this stage, items that were not suitable for donation were separated and moved to a designated area; such items included electricals, sharps and waste. The next stage was to weigh each bag of donations (using Tefal mechanical scales). Each value was logged in kg on pre-prepared data sheets. Finally, each bag was re-opened and a digital photograph was taken of the contents. The bags were laid out in segregated categories (Figure 2), so that the charities would be able to easily take whichever categories they had agreed to take when they collected, meaning the process would run as smoothly as possible.

The purpose of inspecting the donations is to remove items that are unsuitable for donation to charity. Such items may include dirty, dangerous or broken items, or open items of food. The total weight of all unsuitable items was recorded on a spreadsheet in kg.

The contents of each bag of donations were also photographed in order to analyse the material composition to minimise the time required for data collection on the sorting days. Photographs were grouped and analysed one category at a time. In each photograph, the materials present were identified and percentage dominance of each material was estimated. The estimated weight of each material returned to the circular economy was calculated using the composition and the category weight (Table 5, Table 6).

Participants were provided with category stickers so that bags could be easily identified according to what was inside. Each sticker had space to fill in details on student type (UG, PGR, PGT), and nationality (UK, EU, international) and number of bags donated. This was provided in order to be able to collect demographic data on the participants of the scheme. In the cases where these details had been filled in, the responses were recorded on the pre-prepared data sheets.

By email, charities were provided with a simple questionnaire to fill out after the donations had been made to them. This questionnaire asked questions which would generate results for the social and economic KPIs, such as how much money they expected to raise from donations and what social benefits they can generate with that mon-

FIGURE 1: Example of a bag that contained co-mingled materials.

FIGURE 2: Categorising sorted bags ready for collection

ey (Table 7). After data collection was complete, a SWOT analysis was undertaken to identify the new strengths, weaknesses, opportunities and threats to Shift Your Stuff (Table 8).

As the bags of donations were collected by each charity, the numbers of bags from each category taken were tallied on a spreadsheet so that accurate donation figures would be known during follow up communication regarding social and economic benefits. Shift Your Stuff donated goods to various charities - the main beneficiary was the British Heart Foundation. Other charities that took donations were Southampton Refugee Action, Patrick House, St Francis Animal Welfare, Dogs Trust and Jamie's Computers. Most charities pre-arranged a collection time and date, and informed SUSU what they would like to take. Donations for Jamie's Computers and the Dogs Trust were delivered to their local sites.

During the sorting process, an unexpected, opportunistic partnership was formed with the student marketplace 'Stradents'. The director of the scheme came along to the second sorting day and an agreement was made that much of the kitchenware would be made available for a charity sale for pre-sessional, international students new to the university (Figure 3). The items were stored in a room in a SUSU building for two days until the sale took place. Kitchenware items were laid out by category and a donation bucket was provided. Students were encouraged to make a small charity donation in exchange for any kitchenware they wanted from the selection. The sale was advertised on the Stradents social media pages, and the university 'Me-chat' service, a messaging platform for pre-sessional students.

3. RESULTS

3.1 SWOT analysis

The results of the SWOT analysis are shown in Table 2.

3.2 Project development

The review of the existing protocol resulted in a number of alterations being made (Table 3); the revised protocol can be obtained upon request (to idw@soton.ac.uk). Changes to marketing timescales were made in order to maximise the time over which students were exposed to marketing for the scheme through social media and campus advertising. This is important because if advertising is launched after students finish lectures and exams, as the previous protocol advised, many may not be on campus regularly and therefore not see the advertising. Access to to the scheme was planned to be extended by launching the drop-off period earlier in the term. Many students com-

FIGURE 3: Set up for the pre-sessional kitchenware sale.

plete 'coursework only' modules in their final year, meaning that they finish university around mid-May. If the drop-off period is to start after this point an opportunity may be missed to gain donations from those leaving, final year students - the main target audience of the scheme. As the system for donations only requires for drop-off bins to be provided on campus, it is of no inconvenience to move this forward in the term, as long as the space is available.

After analysis of the KPIs, changes were suggested in order to improve the reliability of the data collection for the KPIs. Suggestions included collecting more detailed data, but it was agreed that the possible improvements were not realistically attainable - time and (wo)man-power limitations on the sorting day mean that data collection must be limited to methods that are fast and simple to conduct, making a more detailed analysis impractical. Because of this the indicators remained the same. This exercise did, however, draw attention to areas where it was most important to use consistent methods to collect full and reliable data sets. It also highlighted a need for more data on who is participating in the scheme and how much they are contributing. To collect this data the category stickers provided to participants to label their donation bags were designed to include space for the participants to fill in demographic data (type of student and nationality), as well as the number of bags they had donated. This was hoped to provide new data on which student groups are participating in the scheme, and which groups should be targeted with marketing in the future.

The analysis of last year's project revealed that there was confusion between the charities and SUSU regarding what type of item each charity could take, in what quantities and when they would like to collect. This problem was designed out through amendments to the protocol. Communication with charities before the event meant it was agreed in advance how much the smaller charities would like to receive, the BHF confirmed that they could take whatever else remained, and ensuring there would be minimal items that couldn't be taken by any charities.

A weakness identified from the 2015/16 Shift Your Stuff data collection was that the loss of weighing equipment, along with technological issues with a camera resulting in loss of photographic data, meant that weight and photographic datasets were incomplete and inconsistent. To avoid this happening again, it was ensured that all pieces of equipment (such as scales and cameras) had multiple back-ups, so that should one fail, data collection could continue. Volunteers were thoroughly briefed on the procedure to follow for the processing of the donations, to ensure that data collection was consistent.

3.3 Summary KPIs

3.3.1 Summary KPI: The percentage of all items collected that was sent for reuse

This KPI is designed to give a quick and simple way to measure and compare the success of the project year-on-year. The data used to generate a figure for this KPI was collected during the sorting process: the total weight of the incoming donations; and the total weight of items which could not be sent for reuse (waste). From these the total percentage reuse was estimated via Equation 1. Percentage reuse indicates the amount diverted from final disposal out of all the donations made. This figure can be compared to that of previous years to assess the success of the scheme, and the quality of donations being received.

During this year's project, a total of 2252.9 kg was donated, of which 170.7 kg had to be sent for final disposal due to being unsuitable for reuse. Using these figures, a total 92.4% reuse rate from this year's project was calculated. Last year the reuse rate came to 90.9%, meaning that in 2016/17 a slight increase in reuse rate has been achieved (Table 4). This suggests that students have provided a higher quality of donation to the scheme.

Equation 1: Calculation for Summary KPI: Percentage reuse

$$\text{Percentage reuse (\%)} = \frac{100^{*}\ \text{Total incoming donations(kg)} - \text{total waste (kg)}}{\text{Total incoming donations(kg)}}$$

3.3.2 Environmental KPI: The weight of each material put back into the circular economy through the scheme

The action of taking a photo of the contents of each bag takes very little time but provides a valuable source of data that can be analysed after the donation sorting has been completed. This KPI was developed as an alternative to carbon footprinting, which can be very time and data

TABLE 3: Problems identified during the review of the protocol, and the changes which were made to avoid, reduce or minimise these problems in the future.

Problem	Alteration
Many tasks were assigned to the 'Shift Your Stuff assistant', a role that no longer exists.	Task responsibilities had to be reassigned to various team members within the team running the scheme.
There was no opportunity to allocate tasks to team members.	A task was introduced which instructed the allocation of responsibilities to appropriate to team members.
The marketing launch date was deemed to be too late in the term- after the university summer exam period was over.	There was a general agreement that marketing should be launched when students return from the Easter break (week 30) to maximise exposure.
The drop off period for donations began too late in the term.	The drop off period was suggested to be extended earlier in the term.
Problems recruiting volunteers and increasing participation	It was suggested that the Shift Your Stuff team contacts charity groups within the university, such as Raise and Give (RAG), to see if any would be willing to participate in the running of the scheme. This would bring extra publicity, and extra volunteers, to the scheme
Charities were contacted too few times, and late on in the process, to discuss what donations they would like, and what volume they could take, resulting in confusion between charities and SUSU.	It was agreed that regular contact should be kept between SUSU and each charity, via email, to ensure both parties knew what their expectations were of what we would provide them, how much they could take, and to organise collection times.

TABLE 4: Summary KPI: Percent reuse for the 2015/16 and 2016/17 Shift Your Stuff.

	Sorting Round 1	Sorting Round 2	Overall percent reuse
Percent reuse 2015/16 (%)	94.63	81.41	90.88
Percent reuse 2016/17 (%)	93.25	92.24	92.42

intensive, making it impractical for this scheme. A breakdown of the materials returned to the circular economy from each category is shown in Table 5 as well as totals for each material (Table 6).

3.3.3 Economic KPI: A measure of the amount of money raised through the scheme.

This KPI was calculated by summing the total values of all donations collected through the scheme. As donations were split between 6 charities as well as pre-sessional students it is a challenge to agree upon a single, consistent value for each bag of donations. Each bag of donations varied in weight, quality and content- the bag weight ranged from 0.3-20 kg meaning that values varied greatly between bags. Due to the varying value of donations between charities it is important to speak to each charity to find out the value of the donations they were provided to them. So, after donations were passed over to the charities, follow up emails were sent to assess what value they typically place on each bag of donations they receive, these values vary between charities as they may sell items for different prices in their stores, or the perceived value may vary when the items are being directly used by the beneficiaries. The bag values and number of bags donated were then used to calculate an estimate of the amount of money each charity would generate from Shift Your Stuff. The British Heart Foundation were the only charity to respond to the economic KPI email, they stated that they place an average value of £10 on a bag of donations received from the University of Southampton, meaning that the total value of the donations made to the BHF through Shift Your Stuff was £1,690. The remaining donations were spilt between 5 other charities, from which responses were not received. These donations made up 49% of everything distributed to the charities, if these charities place a similar value of £10 on each bag of donations received, then the extrapolated value of all donations comes to £3,362. Additionally, the Stradents pre-sessional sale raised a total of £112 which went straight to the RAG charities, meaning that in total Shift Your Stuff raised £3,474 for a variety of good causes.

3.3.4 Social KPI: A qualitative summary of the social benefits brought about by the scheme.

Following the donation of goods to the charities, emails were sent to find out what social benefits are brought about by the charities as a direct result of the donations. During the 2015/16 project, the response rate from charities was low, so in 2016/17 it was advised that the team running the scheme kept regular contact with the charities to make sure it was clear all parties expected from the scheme. Instead of a formal questionnaire, which may be daunting to a busy charity, this year a simple e-mail was sent out asking about the social benefits brought about by the scheme. Again, the response rate from the charities was low with only the British Heart Foundation responding. Hence, it was necessary to consult the websites of the rest of the charities to gain a summary of the social benefits they create through their work (Table 7).

3.4 Pre-sessional sale

The Stradents pre-sessional sale provided a platform to sell donated kitchenware to pre-sessional students who had just arrived in Southampton. This initiative

TABLE 5: Environmental KPI: weight of each material returned to the circular economy. This was worked out using material dominance (from photographs) in each category and total category weights.

Category	Weight (kg)	Materials	% composition	Estimate of weight into CE (kg)
Clothing and Footwear	706.25	Textiles Leather Rubber Plastic	89 4 4 2	628.6 28 28 14
Bedding and Linen	305.45	Textiles	100	305.4
Kitchen and homeware	722.20	Textiles Plastic Ceramic Metal Glass Wax Wood Paper/Card	7 23 12 33 6 1 11 9	50.6 166 87 33 43 7 79 65
Food	102.95	Plastic Metal Glass Paper/Card	12 70 10 8	12 72 10 8.2
Books games disks	230.4	Plastic Metal Paper/Card	13 3 84	30 7 193.5

TABLE 6: Total values for the environmental KPI: the total weight of each material returned to the circular economy through the scheme.

Material	Weight back into CE (kg)
Textiles	984.6
Leather	28
Rubber	28
Plastic	222
Ceramic	87
Metal	106
Glass	53
Wax	7
Wood	79
Paper/card	266.7

meant that the kitchenware required minimal transportation, and that it was incorporated straight back into the circular economy, rather that possibly being sat in storage before being sold in a charity shop. Another benefit of this system was that it was directly helping new students at the university and spreading awareness of the scheme. As these beneficiaries are only at the university for one year, it is hoped that when they are finished with the items they will be donated back to the scheme during next year's run. This sale raised £112 and was attended by over 100 students.

3.5 Student demographics

Of the 548 bags that were donated to the scheme, only 35 people completed the demographics sticker provided, meaning that solid conclusions cannot be made about the types of students participating in the scheme. Twenty-eight out of 32 respondents were undergraduates, compared to only 4 postgraduates. Regarding nationality, 25 of 34 respondents were from the UK, 7 international and 2 EU. This data cannot be used to conclude who is participating in the scheme, and is unlikely to be of use in knowing which student groups to target in the future.

The addition of the 'Stradents' pre-sessional sale showed how effective targeting certain groups of students properly can be. This sale was organised in two days and was advertise to pre-sessional students through the Stradents Facebook page, Facebook groups for pre-sessional students, and the University's 'me-chat' messenger groups, over 100 students attended.

4. DISCUSSION

The 2016/17 Shift Your Stuff campaign had mixed results. Flaws encountered previously in the scheme were avoided through refinements to the procedure and being better prepared for the sorting process. A SWOT analysis of the 2016/17s scheme was generated (Table 8). The requirements of the scheme were better communicated, but some restricted items were still donated; this was not too much of a problem as it was anticipated and places to send such items were organised. Other issues continued to occur despite changes being made to try and avoid them. Again, contact with charities was limited, and in turn response to follow up questionnaires was low. In future a better method of communication should be organised with charities early on in the process, such as getting responses whilst they collect donations.

Sticking to the time schedule of the protocol was challenging, with some team members not knowing when tasks needed to be completed by, or failing to meet the deadlines set out in the protocol. Other deadlines which were agreed during planning meetings were also not met, particularly the agreed upon deadlines for when the team should begin marketing the campaign; marketing was not launched until over a month after the target launch date. Agreements were also made to launch the drop-off points earlier in the term to allow longer for drop-offs, but due to delays with bags this target was not achieved. Informal discussions revealed that the snap general election which was called in the run up to Shift Your Stuff diverted a lot of the time and resources which would have been invested in promoting and marketing the scheme. This may have had an impact on how many students heard about the scheme and reduced potential participation, as well as impacting the timeline of events during planning.

Another observation is that some of the most successful reuse schemes at other HEIs are at universities where

TABLE 7: Summary of the social benefits brought about by each charity that participated in the scheme, as reported through the charities by email or through websites.

British Heart Foundation	BHF use donations in a variety of ways including funding Nursing Care for those with heart problems, funding research into the diagnosis, prevention and treatment of heart disease, social support for heart disease sufferers. They also fund CPR training, distribute defibrillators, provide educational resources and keep BHF charity shops running (BHF, 2016a).
Southampton Refugee Action	Refugee Action provides advice for refugees and asylum seekers, gives practical support to those resettling in the UK and support organisations working with refugees. Donations went directly to be used by refugees and asylum seekers re-settling in and around Southampton (Refugee Action, 2017)
Two saints-Patrick House	Two saints-Patrick house offer safe accommodation for the homeless, and support those with homes to help them maintain their independence by offering free learning services and help with job hunting (Two Saints, 2016).
St Francis Animal Welfare	Care for, rehabilitate and re-home numerous domestic animals (St Francis, 2017).
Dogs Trust	Care for, rehabilitate and re-home dogs (Dogs Trust, 2017).
Jamie's Computers	A computer recycling social enterprise which aims to provide education and work experience through hands on training for its service users. It is run by the society of St James who address poverty, sickness, hardship and distress (Jamie's Computers, 2016).

TABLE 8: SWOT analysis of Shift Your Stuff 2016/17 project.

Strengths	Weaknesses
Donations were of a higher quality than previous years- higher reuse rate.	Lack of volunteers
Most items were pre-sorted in categories, meaning minimal sorting time was required.	Despite charities agreeing collection times, some did not turn up at their agreed time- causing issues for the sorting teams.
Charities organised collection times and amounts in advance.	Issues relating to kerbside donations being damaged by scavengers and bad weather.
Charities collected soon after sorting meaning less storage space was needed.	Some donations were unsuitable for re-use- i.e. dirty clothes, broken items, sharp knives and electricals.
Photographic data allowed quick data collection.	Total weight of donations was lower than in previous years.
A sale was set up to provide pre-sessional students with kitchenware and raise money for RAG charities.	Marketing was launched later than planned, meaning fewer students may have heard about the scheme than would have otherwise.
Data was collected for the KPIs, which allows comparison with the previous year.	Demographic stickers were filled in by very few students, meaning that the data isn't useful for future years
The scheme set a good example for sustainable waste management	Participation in the scheme was low, with only 548 bags donated.
Students' unwanted items were diverted from disposal.	Low response from charities to the economic and social questionnaires.
The scheme was supported by SCC and BHF, as well as several other universities.	Some bags contained items from a mixture of categories, meaning they took time to re-sort.
Scheme may have prevented litter on the streets.	
Opportunity	**Threats**
Possibility for expansion/more support if partnerships are formed with more charity groups within and outside of the University, and with other organisations such as Solent University.	Lack of student involvement with SUSU may be leading to low interest/participation.
Better support from the University could allow better recognition of the scheme from students.	Not including data from the halls collections may make the figures look lower than they are.
Large potential for food collections if that aspect of the scheme is pushed further.	
Large potential for food collections if that aspect of the scheme is pushed further.	

there is a strong relationship between the university/student's union and the student body. This close link may mean that students are more exposed, and pay more attention, to publicity for the scheme put out by the university. Students may be more likely to participate in the scheme if they have a good working relationship, and trust in, the organiser of the scheme.

The pre-assigned categories continued to minimise the amount of time required in the sorting process, with only a small amount of sorting/categorising required during the process. Additionally, the quality of donations was increased, resulting in a reuse rate of 92.4%. Sorting was carried out consistently, with the same equipment being used. Additional volunteers were brought in during the second round of sorting to deal with the larger quantities, but the same people were leading the sorting process to eliminate inconsistencies. Detailed guidance was given on instruction sheets handed out with the donation bags; these detailed which items fell into each category. For the most part, these categories were stuck to and very few items needed to be re-bagged. Despite the guidance, some prohibited items were still donated: 9 kg of sharp knives were donated, along with various other unsafe, non-accepted items (Figure 4). The safest disposal route for these was to the police knife amnesty, a contingency which should be planned for in advance in future years.

The kerbside collection was well utilised on both collection days: on the first collection day 151 kg was dropped-off compared to 225 kg collected through kerbside collection. During phase two 1094 kg was donated through campus drop-off points compared to 612 kg through kerbside collection. One problem which was relayed through the council's collection team was scavenging. To utilise the kerbside collection service students were advised to put their bags out on the morning of the collection day, evidently for some students this was not a suitable time and so they put their bags out earlier, some up to a week before the collection day. During the time that the bags were left on the streets some experienced damage through vandalism, bad weather and scavenging. Many bags were ripped open and contents strewn across pavements by people in search of useful or valuable items. The collection team estimated that around 50% of all bags left out for collection had been damaged or tampered with, meaning that they couldn't be collected for reuse. Street cleaning teams would then have to visit the student areas to clean up the resulting mess. So, in this situation, the scheme which has set out to reduce litter issues on the streets has inadvertently created mess on the streets; this was not the fault of the students. Discussions involved the consideration of working more closely with the council in future years to create 'safe' drop-off points where donations could not be tampered with. This action could dramatically increase the amount of successful donations made, and reduce costs involved in clearing the resultant mess, this should be followed up in future years to see if the city council would be willing to play a more active role in the scheme.

FIGURE 4: Prohibited items which were donated. electricals, knives, toilet brushes.

Volunteer recruitment continued to be a problem, with no-one signing up for the first session. For the second session, volunteers were recruited from the SUSU staff, meaning that there were fewer problems with people not turning up.

From analysis of similar schemes at universities across the country (Appendices 1, 2), it can be seen that these types of scheme have the potential to be hugely successful. Warwick University collected over 82 tonnes of donations from a student population of 24,000 (Warwick University, 2015). The collaboration of various HEIs and charities in Bristol produces outstanding results each year. The Bristol 'Big Give', a joint initiative between Bristol Waste Company, University of Bristol, Bristol SU, UWE, USESU, various student housing providers and the Charities BHF and Bristol HUB, brought in 96 tonnes of donations during 2017 (as of 24th June 2017). The Big Give has been running for 6 years, 3 of which has been in partnership with the BHF and is perhaps one of the best examples of a successful model for a HEI reuse scheme (Bristol, 2017).

The most successful HEIs have committed time and resources to develop and improve their schemes over years, due to the need for understanding and recognition of the scheme through the student community. These schemes seem to take several years to become well-established; during this time it is important to commit resources to the expansion of the scheme. A possible key factor in the scheme success could be the faith that the students have in the organisation that is running it. The Southampton University Student's Union is known to have a poor reputation amongst Southampton students, many students feel that they are poorly represented by the Union (HEFCE, 2017), and therefore may have little interest in the activities and schemes that are run by SUSU, and be less inclined to participate.

A possible way to boost the power of the scheme and expand to other universities would be to make use of a consortium such as the Southern Coast Affinity Group (SCAG). By collaborating with other HEIs and organisations such as WCAs and local authorities the scheme could have the potential to expand its coverage, participation and success- the more organisations are involved the more marketing and recognition the scheme will receive. A partnership such as this could boost Shift Your Stuff to a similar scale of projects such as the Bristol Big Give.

There are a number of options for the future of the scheme. The simplest option for SUSU is to carry on running the scheme. It is unlikely that this method will allow the scheme to grow (donations have decreased every year), but it will still allow the Union to exercise a sustainable waste management scheme and make charitable contributions, whilst contributing to litter management in Southampton. This method uses leaflets, posters and a small level of social media advertising to raise awareness of the scheme, but has limited reach to students, which may explain the limited participation.

Another option for SUSU to consider is increasing their effort in the scheme. By investing more effort into the marketing and advertising of the scheme, it could have a larger reach and therefore be more successful in terms of donations. This could be done by improving the student outreach of the scheme; due to restrictions the Union are limited on how much contact they can have with student groups, but with more involvement from the host university the scheme may be able to be advertised to more students who are not otherwise reached by SUSU's social media streams. Door-to-door contact could also be beneficial in increasing participation. In the 2014/15 Shift Your Stuff over 4 tonnes of donations were brought in, the main change to the scheme

being the loss of door-to-door bag drop-offs. Students may be more likely to participate in a scheme if it requires minimal effort from them. This method should also incorporate targeting of international students- when leaving University, most international students will get rid of most of their household items (anything that isn't worth travelling thousands of miles with), meaning that they are a good source of reusable items.

It is clear that some of the more successful schemes are those which involve partnerships between various organisations. The University of Southampton could explore the possibility of partnering with other HEIs in the area such as Solent University and the University of Winchester. If the opportunity arose it would also be beneficial to partner with a charity group such as Southampton RAG or Southampton HUB. By partnering with a charity group, Shift Your Stuff could be better supported in terms of resources for volunteers, have a wider reach for advertising through social media channels and have access to people who have good experience with such schemes who can advise in making it run more smoothly- all of which have been issues for Shift Your Stuff in the past.

Working more closely with Southampton City Council would be valuable for the scheme. This would be beneficial for both parties in that it could make the scheme more successful, reduce litter on the streets, and therefore reduce the costs and time involved with clearing it up. The council could aid the scheme by assisting with logistics and marketing, as well as possibly being able to help devise a way of making donations more secure on the streets. Collaboration between all possible parties could help the scheme meet its full potential. Involving organisation from across Southampton could make Shift Your Stuff a city-wide event rather than a scheme which is confined to student areas.

If SUSU feels that it cannot commit enough staffing and resources to expand the project to meet its full potential, it should be considered that the responsibility for running the charity collection could be handed over to the BHF. The BHF runs 'Pack for Good' for over 80 Universities and is hugely successful, raising over £1.5 million since 2011. The charity has dedicated teams to run the scheme in each region, meaning that they are experts at what they do and know exactly how to optimise the scheme to collect maximum donations. The BHF would take responsibility for all advertising, collections and sorting, meaning that SUSU would have minimal input of resources to the scheme. The only downfall to this scheme would be that all donations go to one charity, meaning that it cannot help local charities as it does currently.

5. CONCLUSIONS

The Shift Your Stuff scheme has successfully enabled reuse of household items that young people (in this case, students) would otherwise have thrown away, diverting waste from landfill and from the streets of a major urban centre (in this case, Southampton). The use of KPIs as a measure of success is providing a way to compare results year-on-year, and assess the impacts of the scheme. Despite succeeding in its general purpose, Shift Your Stuff has the potential to become a much more successful scheme, in that it could bring in more donations and in turn have better environmental, social and economic benefits. In comparison to other schemes at UK HEIs, Shift Your Stuff is bringing in a small amount of donations given the amount of students at the University of Southampton. From analysis of similar schemes, it is advised that the Students' Union looks to expand the scheme, either by collaborating with other organisations, or by passing on the responsibility of running the scheme to a charitable organisation that is set up for this purpose (e.g. the British Heart Foundation's Pack for Good campaign).

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APPENDIX 1: Summary table of reuse schemes at UK higher education institutions.

University scheme	Details
Aberdeen, University of	Student accommodation clearance scheme where local charities benefit from unwanted items. Permanent Swap Shop for exchanging/reuse.
Bangor University	Collaboration between Love Your Clothes, Bangor University Sustainability Lab and Bangor students' Union. Challenged residents to donate 1 tonne of unwanted clothing to the pop up 'swap shop' to be donated to Age Cymru, BHF and a local charity. Workshops also given on sewing, upcycling, and clothing care and repair.
Bath, University of	Bath University provide general guidance on their website with details on what and how to donate to permanent donation bins. Donation bags are provided in student accommodation.
Bristol 'Big Give'	Joint initiative from Bristol City Council, Bristol Hub, University of Bristol, University of Bristol Students' Union (Bristol SU), the University of the West of England, UWE Students' Union (UWESU) and the British Heart Foundation. Items collected from student halls. 21 donation points are placed around student areas from late April until mid-July. Aim of collecting £125,000 worth of donations. Summer 2016: 78 tonnes (£135,000 value) Donations benefit charities other than BHF including Lifecycle, St Peter's Hospice and Children's Hospice South West as well as other local charities. Door knocked to speak to over 1000 students about responsible waste management and the charity banks. The scheme has been described to have "[a] profound impact on unsightly waste and fly-tipping at the end of term" and to "aid relationship building between neighbours in areas with a high student population."
Durham 'Green Move Out'	The scheme was launched in 2005 and has grown every year since. Everyone in halls is provided with a bag and instructions ahead of move out period. Additional bags are provided at halls receptions. Guidance on how to donate: books, food, sharp/fragile items. Private rented houses provided with bags and instructions ahead of move out period. Place outside for kerbside collection on pre-specified day. Categories: clothing, shoes, bedding, electrical items, pots and pans, crockery. Local charities including County Durham Furniture Help scheme, Durham Palestine Educational Trust, East Durham Trust, A Way Out.
Imperial- end of term reuse scheme	Food collection boxes placed in halls kitchens during move out period, (2014- £8000 worth of food donations, around 2 tonnes). British heart foundation runs the main collection scheme with collection bin placed around halls. BHF bags delivered to all halls rooms. Collection point in common rooms/receptions.
LSE 'Relove'	Reuse program that runs at the end of academic year. Donations go to the BHF. Some selected items are stored over summer and offered to students in exchange for a donation at the start of the year (proceeds go to their Students' Union sustainability projects).
Sheffield 'donate don't waste'	Collaboration between Sheffield University and Sheffield Hallam University. Donation bags and donation points are provided in halls. Donation bags are also available from the SU for students living in private accommodation, donation points located in the SU. Scheme also incorporates a summer food drive, collection points at halls and SU. In 2016 this collected over 3000 items. The council provide 2 red sacks per person to be collected in addition to the regular household waste collection during move out period for anything that cannot be re-used/recycled. This is to help with street litter issues; this waste is sent for incineration.

University scheme	Details
Solent 'Eco Ernie'	This scheme was launched in 2008 and involves door to door collections around student areas at the end of the summer term using a milk float. Bags are distributed to students in private accommodation towards the end of the academic year. The scheme held 6 collection days for private housing, and 3 for student halls. 2012: around 3 tonnes of clothes and textiles and one tonne of electricals collected. The best items are resold at a 'church bazaar' and the rest is recycled or scrapped. Piggybacked by the 'Give a Bra' which provided bags specifically for bras to be sent to west Africa. Unlike many other collections, duvets were accepted to become bedding for horses. Textiles donated to the Hampshire and IoW Air Ambulance.
St Andrews 'St AndRe-Use'	Throughout the year 'St AndRe-use' open drop off points around the campus. Items collected are redistributed to students for free at 'The Big Giveaway' during Freshers' week and 'The Slightly Smaller Giveaway' at the start of Semester 2. Two major collection drives occur at the end of each term in Halls of Residences and one or two other University locations.
Warwick, University of	As well as participating in BHF Pack for Good, Warwick run a separate food collection at the end of the academic year, in 2015 this brought in 16.7 tonnes of food (RAWKUS project). Curtains and curtain poles from refurbished halls donated to BHF and 152 chairs to the Coventry city mission. 5,300 used duvets collected from campus rooms and donated to charities which help homeless women and refugees.

APPENDIX 2: Summary table of some successful 'Pack for good' schemes at various UK Universities.

University	Details
Birmingham University 'Junk-busters'	The pilot scheme for the "Pack for good" campaign. Aim to reduce waste left on the streets and raise funds for BHF. Amount raised to date: £86,000 2013/13 nearly 1800 bags were collected worth over £14,000 Collect bags on an eco-friendly milk float. 11 collection days between May and early July.
Cambridge, University of	2013: 1998 bags collected 2014: 3043 bags collected 2015: 6644 bags, equivalent to 53.1 tonnes. Collected between 1st April- 31st September. Value to BHF: £93,016. Net benefit by CO_2 emissions equivalent compared to landfill: 54,040kg
Chester, University of	Working together since 2012 2015 campaign: collaboration between Uni of Chester and 'Fresh Student Living' (private accommodation). April-September campaign period. 613 bags donated, equates to 4.9 tonnes (£8582 value). Net benefit by CO_2 emissions equivalent compared to landfill: 4987kg.
Coventry University	2015: Estimate of £30,000 worth of donations. From 2,265 bags.
Derby, University of	2015: 1041 bags collected, stock value £14,500. April-July.
Edinburgh, University of	Student led approach Partnership with BHF, SHRUB (student led swap and reuse hub), TIC international. Identify varieties of waste and which are valuable to each partner. Good communication: posters, leaflets and social media. Recruiting volunteers: student to student communications work best. Mailing list of student interested in sustainability- incentive of volunteers getting 'first dibs' on the donated items if they wish. Giveaway of essential Uni items during move in week- eg pots and pans.
Hull, University of	2013: 522 bags (3.1 tonnes) donated in total. Most donations early-mid June. Used a bag value of £20- estimates £10,440 value. 31,874 net benefit (kg CO_2 equivalent compared to landfill. Press released prior to collections in the hull daily mail with donation guidance. Local BHF gave a 15% discount for students wishing to purchase furniture/electrical to promote re-use.
Leicester, University of	2014/15 academic year: 674 bags of old clothing and utensils from the ten donation bins placed across the accommodation sites.
Loughborough University	2014: permanent donation points, full marketing campaign 1842 bags donated. 2015: Reps collected from halls, as well as donation points. 2,467 bags collected (£34,538 value) 2016: permanent donation banks, rep collections from halls, and pop up donation boxes during summer clear out. 2,838 bags donated (£39,732 value).
Newcastle University	Newcastle Uni, NUSU, Northumbria Uni, and Newcastle city council partnership with BHF. 2016 campaign: 3,776 bags donated (£87,220 value). Been running since 2012.
Nottingham, University of	Donation bins (at points between accommodation and uni), food drop off points, and 'horse and dray' doorstep collection.
Oxford, University of	2015: 56,336kg of donations (£99,288 value). Partnership between the City Council, Oxford Brookes and Oxford University Colleges. Red bins placed around the city. Regular collections from some accommodation services throughout summer term. Promoted through OUSU at the SU, on social media.
Warwick, University of	Top university collector three years running. 15 permanent collection banks, 22 pop up banks. 2015/16 academic year: 82.9 tonnes collected (£116,536 value).
York, University of	Council makes extra collections of general waste during moving out period. Students and locals encouraged to donate suitable items to clothing banks and drop off points at round uni. Info leaflets and maps of drop off points distributed prior to the event. 2014: 15.6 tonnes of donations, 1899 bags (£38,000 value). Council reminding students how to dispose of waste responsibly ahead of and during move out period, and through university social media.

APPENDIX 3: Email sent to students during volunteer recruitment drive.

Dear Students,

We are now recruiting volunteers to help us sort what we collect for Shift Your Stuff 2017. Last year we were able to sort over four tons of waste and this year we aim to recycle double that amount!

We are looking for student volunteers who available are anytime on June either 13th, 14th, 30th or July 1st to join us and help with the sorting of tons of stuff. Volunteers will receive free pizza.

You can sign up directly with this link and or through our Facebook event. Full details of Shift Your Stuff can be found at www.unionsouthampton.org/shiftyourstuff.

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PROPOSITION OF THRESHOLD FOR WASTE CONTAMINATED WITH MERCURY (COMPOUNDS) IN APPLICATION OF THE MINAMATA CONVENTION ON MERCURY AND IMPACT ASSESSMENT

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Article Info:
Received:
12 March 2019
Revised:
18 June 2019
Accepted:
20 June 2019
Available online:
28 June 2019

Keywords:
Concentration limit
GHS
Ecotoxicity
Environmentally sound management

ABSTRACT

The Minamata Convention on Mercury is a global treaty to protect human health and the environment from the adverse effects of mercury. Environmentally sound management of waste is under discussion. This note proposes a threshold for waste of category c) *Contaminated with mercury or mercury compounds* to be disposed of (Article 11 of the Convention), using the Globally Harmonized System of classification and labelling of chemicals of the United Nations (GHS - UNEP, 2017). Mercury and mercury compounds are classified as substances for the physical, health and environmental hazards categories. The thresholds of mercury and mercury compounds classifying a mixture as hazardous for the different hazard categories (physical, health, environmental) are "Presence", >0.3% and >0.0025% (25 mg mercury/kg of waste) respectively. For impact assessment, this threshold is then compared with large data set of hazardous (793 data), potentially hazardous (depending on the concentration of hazardous substances) (55 data), as well as natural or non-polluted anthropized media (composts, sediments, agricultural soils) (21 784 data) from France. This demonstrates that 75% of the hazardous waste have higher total mercury concentration, that potentially hazardous waste samples have lower concentrations, and that all composts, agricultural soils and marine sediments and 99% of the fluvial sediments have lower concentrations. So, this threshold will not classify common industrial waste or natural media as requiring special treatment for safe disposal, but well a large part of industrial hazardous waste.

1. INTRODUCTION

The Minamata Convention on Mercury (UNEP 2017) is a global treaty to protect human health and the environment from the adverse effects of mercury. Regular conference of the parties progress in technical recommendations to "make mercury history". Environmentally sound management of waste is one point under discussion.

The Convention defines in Article 11 "Mercury wastes":

"…2. For the purposes of this Convention, mercury wastes means substances or objects:
(a) Consisting of mercury or mercury compounds;
(b) Containing mercury or mercury compounds; or
(c) Contaminated with mercury or mercury compounds,
in a quantity above the relevant thresholds defined by the Conference of the Parties, in collaboration with the relevant bodies of the Basel Convention in a harmonized manner, that are disposed of or are intended to be disposed of or are required to be disposed of by the provisions of national law or this Convention. This definition excludes overburden, waste rock and tailings from mining, except from primary mercury mining, unless they contain mercury or mercury compounds above thresholds defined by the Conference of the Parties…"

The scope of the paper is to provide a reliable concentration limit to the Minamata Convention. This paper uses the Globally Harmonized System (GHS) of classification and labelling of chemicals of the United Nations to propose a threshold and compare it with concentrations observed in waste and natural media. The method is explained in detail in the paper, with a focus on ecotoxicity, which appears to be the property with the lowest dangerous ranking.

In this paper, the "concentration limit" used in the GHS is the equivalent of "threshold" of the Minamata Convention on Mercury, and "substance" used in the GHS is the equivalent of "compound" of the Minamata Convention on Mercury.

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Detritus / Volume 06 - 2019 / pages 25-31
https://doi.org/10.31025/2611-4135/2019.13822

2. MATERIAL AND METHODS

2.1 Properties of mercury and mercury substances in the GHS

It is proposed to use the Globally Harmonized System (GHS) of classification and labelling of chemicals of the United Nations (last version: UNEP, 2017). The European Union has adopted the GHS system in 2008 (CLP, 2008) and has developed an official list of the hazard properties of 4 249 substances, as well as a registration and self-classification system for producers and importers of chemical products (REACh system). This list is an Annex of the CLP regulation and can be downloaded in a spreadsheet template (CLP, 2018) and is used here. These "harmonized" (at EU level) data were built by working group of experts and have been used to collect consistent data on mercury and mercury compounds and to propose threshold consistent with the regulation.

2.2 Particular case: ecotoxicity in the GHS

The different categories of ecotoxicity are attributed to substances from experimental laboratory ecotoxicological standardized tests. Organisms are submitted to different concentration of the substance in their living medium, and the concentration producing 50% (or x %) of mortality or effect after a given time (called LC_{50} or EC_{50} or EC_x depending on test), or the highest concentration producing no observed effect (called NOEC) is measured. These concentrations expressed in mg of substance per liter of living medium are compared to concentrations of Table 1, and a category is attributed to the substance.

For substances with experimental results < 1 mg/l, the GHS uses multiplying factors called "M factors" to fine tune the classification of substances and mixtures with these substances. M factors are determined by the lowest $L(E)C_{50}$ and NOEC experimental values (Table 2). Mercury and mercury compounds are non-rapidly degradable compounds.

Experimental ecotoxicity data of mercury and mercuric substances were taken from a reference UE publication (UE 2005), to determine M-factors for mercury and mercury compounds.

The ecotoxicity of mixtures with ecotoxic substances at a given concentration can be assessed by the calculation rules presented at Table 3 (copy of Table 4.1.3 of the GHS).

The European Union updated criteria on Ecotoxicity classification of waste (Regulation 997/2017, EU 2017). For simplicity of classification of waste with most of the time unknown mineral substances composition, this regulation does not use the multiplying factors M-factors (that are attributes of substances). M-factors were developed in the GHS to fine tune the ecotoxicity of ecotoxic substances (EC_{50} or NOEC < 1 mg/L), and to avoid the creation of multiple hazard statement codes to reflect the different grades of ecotoxicity. The resulting concentration limit for waste by calculation method as proposed by the 997/2017 Regulation for chronic ecotoxicity for mercury and mercury compounds is 0.25% or 2500 mg/kg, hundred times higher than the concentration calculated here (see Results). Many experts believe that the abandonment of M factors for waste is irrelevant. This method has not been used here.

2.3 Data of concentration of mercury in waste and natural media

Data of hazardous waste, potentially hazardous waste

TABLE 1: Categories for substances hazardous to the aquatic environment for short-term (acute) and long-term (chronic) aquatic hazard (extract of Table 4.1.1 of GHS).

(a) Short-term (acute) aquatic hazard
- Category Acute 1: $L(E)C_{50}$ fish, crustacea, algae ≤ 1 mg/l
- Category Acute 2: $L(E)C_{50}$ fish, crustacea, algae > 1 but ≤ 10 mg/l
- Category Acute 3: $L(E)C_{50}$ fish, crustacea, algae > 10 but ≤ 100 mg/l

(b) Long-term (chronic) aquatic hazard

(i) Non-rapidly degradable substances for which there are adequate chronic toxicity data available
- Category Chronic 1: Chronic NOEC or EC_x fish, crustacea, algae ≤ 0.1 mg/l
- Category Chronic 2: Chronic NOEC or EC_x fish, crustacea, algae ≤ 1 mg/l

(ii) Rapidly degradable substances for which there are adequate chronic toxicity data available
- Category Chronic 1: Chronic NOEC or EC_x fish, crustacea, algae ≤ 0.01 mg/l
- Category Chronic 2: Chronic NOEC or EC_x fish, crustacea, algae ≤ 0.1 mg/l
- Category Chronic 3: Chronic NOEC or EC_x fish, crustacea, algae ≤ 1 mg/l

TABLE 2: Multiplying factors (M factors) for substances for highly toxic ingredients or mixture (categories Acute 1 and Chronic 1) (Table 4.1.5 of GHS).

Acute toxicity	M factor	Chronic toxicity	M factor	
$L(E)C_{50}$ value (mg/l)		NOEC value (mg/l)	Non-rapidly degradable ingredients	Rapidly degradable ingredients
0.1 < $CL(E)_{50}$ ≤ 1	1	0.01 < NOEC ≤ 0.1	1	-
0.01 < $CL(E)_{50}$ ≤ 0.1	10	0.001 < NOEC ≤ 0.01	10	1
0.001 < $CL(E)_{50}$ ≤ 0.01	100	0.0001 < NOEC ≤ 0.001	100	10
0.0001 < $CL(E)_{50}$ ≤ 0.001	1000	0.00001 < NOEC ≤ 0.0001	1000	100
0.00001 < $CL(E)_{50}$ ≤ 0.0001	10000	0.000001 < NOEC ≤ 0.00001	10000	1000
(continue in factor 10 intervals)		(continue in factor 10 intervals)		

and non-hazardous waste and natural media have been gathered:

- The concentrations of mercury, cadmium and lead in 793 hazardous waste (data from a hazardous waste management company) (Hennebert, 2012);
- The 55 available data concentrations from an INERIS database for the following waste: car shredding residue (11), epoxy powder (1), excavated soil underlying a road (3), excavated soil (11), foundry sand (1), municipal solid waste incinerator bottom ash - MSWI BA (20) , paint residue (1), phosphogypsum (2), sand blasting residue (1) and sand from incineration fluidized bed (4);
- The heavy metals in different composts (379 samples) from organic fraction of municipal wastes (separately collected or mechanically sorted) of 30 sites in France (Zdanevitch 2012);
- The results of routine quality monitoring of sediments (11 791 samples of fluvial sediments and 816 samples of marine sediments) of France by the Water Agencies (Padox and Hennebert 2010a, b);
- The data from routine analysis of agricultural soils (8 798 samples) are gathered at the French level by soil scientists (Gissol, 2018).

3. RESULTS AND DISCUSSION

3.1 Properties of mercury and mercury substances in the GHS

The mercury and nine mercury substances from the list are presented at Table 6. Two "generic entries" are also listed. Most frequently in waste, field measurement with fluorimeter and routine laboratory analysis will deliver total mercury concentrations (metallic or not) rather than mercury substances concentrations. If the exact mercury substances present in the waste is not known, which is frequently the case in waste, these "generic entries" are used in the EU (line 2 and 3 of the substances list in Table 6: "inorganic compounds of mercury with the exception of mercuric sulphide and those specified elsewhere in this Annex" and "organic compounds of mercury with the exception of those specified elsewhere in this Annex"). In total twelve entries are used for mercury and mercury substances, with their hazard statement codes (Hxxx).

Mercury and mercury substances are all classified Acute Toxic when the three routes (oral, dermal and inhalation) are considered, classified Specific Target Organ Toxic Single Exposure (1 substance: calomel) or Repeated Exposure (11 other substances), Ecotoxic Aquatic Acute Level 1 and Ecotoxic Aquatic Chronic Level 1. Additionally, mercury dichloride is classified mutagenic and reprotoxic, and elemental mercury is reprotoxic. Mercury difulminates and dimercury dicyanide dioxide are explosive (Table 6).

3.1.1 Classification of mercury and mercury substances (GHS, EU data) for ecotoxicity

Experimental ecotoxicity data of mercury and mercuric substances from a reference UE publication (UE 2005) are presented at Table 4. A summary can be found on the INERIS portal (INERIS 2018). The resulting M factors are

TABLE 3: Classification of a mixture for short-term (acute) and long-term (chronic) hazard based on summation of the concentrations of classified ingredients (Tables 4.1.3 and 4.1.4 of GHS).

Category	Sum of the concentrations (in %) of ingredients classified as:	Mixture is classified as:
Short-term (acute) hazard	Acute 1 * M ≥ 25%	Acute 1
	(M*10*Acute 1) + Acute 2 ≥ 25%	Acute 2
	(M*100*Acute 1) + (10*Acute 2) + Acute 3 ≥ 25%	Acute 3
Long-term (acute) hazard	Chronic 1 * M ≥ 25%	Chronic 1
	(M*10*Chronic 1) + Chronic 2 ≥ 25%	Chronic 2
	(M*100*Chronic 1) + (10*Chronic 2) + Chronic 3 ≥ 25%	Chronic 3
	Chronic 1 + Chronic 2 + Chronic 3 + Chronic 4 ≥ 25%	Chronic 4

TABLE 4: Experimental $L(E)C_{50}$ and NOEC of mercury and mercury substances (UE 2005) and corresponding ecotoxicity level and multiplying factors (M factors).

Water		Fresh			Marine		Classification	
Ecotoxicity Organisms	Tests results (mg/L)	Ecotoxicity category (Table 2)	M-factor (Table 3)	Tests results (mg/L)	Ecotoxicity category (Table 2)	M-factor (Table 3)	Category	Proposed M factor
Acute	$L(E)C_{50}$			$L(E)C_{50}$			Acute 1	100
Algae	0.010000	Acute 1	100	0.010000	Acute 1	100		
Invertebrate	0.010000	Acute 1	100	0.003500	Acute 1	100		
Fish	0.000700	Acute 1	1000	0.070000	Acute 1	10		
Chronic	NOEC			NOEC			Chronic 1	100
Algae	0.000200	Chronic 1	100	0.000900	Chronic 1	100		
Invertebrate	0.000290	Chronic 1	100	0.000100	Chronic 1	1000		
Fish	0.000620	Chronic 1	100	0.005000	Chronic 1	10		

most of the time 100 (with two values of 10 and two values of 1000 in a set of 12 values). It is proposed here to use M = 100 for acute and chronic ecotoxicity.

Using the calculation rules of Table 3 and the M factors obtained in Table 4, the concentration limits classifying a mixture containing mercury or mercury substances as ecotoxic acute and chronic are presented at Table 5. For full classification of mixtures, all the other ecotoxic substances must also be used, but this is not the question here. The level 3 has the lowest concentration: a mixture is hazardous if the concentration of mercury and/or mercury substances is greater than 0.0025% (mass/mass), or 25 mg/kg (sometimes expressed as 25 parts per million - ppm). These concentrations are reported as concentration limit for Environmental hazard in Table 6.

3.1.2 Proposed concentration limits for waste containing mercury or mercury compounds for physical, health and environmental hazard classification

The concentration limits are presented in the last row of Table 6, with the lowest by category of hazard in color. For human acute toxicity, concentration limits are derived from acute toxicity estimates for mixtures of the GHS. Some substances have specific concentration limits for some hazard class category in the EU list of substances, but they were not used here. For physical hazard, the requirement is "presence". For health hazards, the lowest concentration is 0.3% of mercury or mercury substances in the mixture (reproductive toxicity). For environmental hazard, the concentration limit is 0.0025% or 25 mg/kg.

3.2 Impact assessment of proposed threshold

Numerous data (> 22 000) from France, for different waste streams were collected to perform impact assessment of the proposed threshold:

- Hazardous waste (according to the EU List of Waste);
- Potentially hazardous waste (depending on the concentration of hazardous substances in it) (so-called "mirror entries" in the EU List of Waste), including municipal solid waste incinerator (MSWI) bottom ash;
- Composts, sediments and agricultural soils.

Regulatory considerations on municipal sewage sludge (biosolids) and waste intended to be used as fertilizers are also presented.

More than 75% of the hazardous wastes have more than 25 mg Hg/kg. Data of Cd and Pb are presented, to illustrate that other hazardous elements are most of the time also present in these wastes (Table 7).

The Hg concentration for 55 potentially hazardous waste are presented at Figure 1 (left) and for MSWI BA at Figure 1 (right).

All the reported concentrations are lower than 25 mg/kg. For the MSWI BA, 8 samples have concentrations lower than the quantification limits of the laboratory (between 0.175 mg/kg and 5 mg/kg).

The distributions of Hg in different composts from organic fraction of municipal wastes (separately collected or mechanically sorted) of 30 sites in France (Zdanevitch, 2012) are presented in Table 8.

The results of routine quality monitoring of sediments are joined to the table. The network aims to monitor pollution and hot spots are more intensively sampled. The regulatory concentration limits of Hg for these sediments for reuse in natural environment are 1 mg/kg and 0.8 mg/kg respectively. One percent of the fluvial sediments should be specifically managed for mercury, with the threshold of 25 mg/kg, and no marine sediment. These samples are contaminated by other heavy-metals (As, Cu, Cd, Pb, Zn - result not shown) and are in all the case identified in survey and separated during dredging.

Data from routine analysis of agricultural soils are presented (Gissol 2018). A detailed analysis is available (Baize et al 2007). Another publication deals among others with local geological "anomaly" of heavy metals (Baize 2000). According to the authors, the higher concentrations originates from the parent material of the soil (most of the case) or from industrial inputs. All the concentrations are lower than 25 mg/kg (Table 8).

For municipal sewage sludge, the EEC Council Directive on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (EEC 1986) (updated) states in its Annex I B the limit values for heavy-metal concentrations in sludge for use in agriculture for mercury: 16 to 25 mg/kg of dry matter. In France, the corresponding decree sets a concentration limit of 10 mg/kg of dry matter (RF 1998).

For reuse of waste as fertilizing products in the Circular Economy package (EC 2016), the EC has confirmed the concentrations of Hg in fertilizers, culture medium and soils improvers of 1 mg/kg and in one case 2 mg/kg, that

TABLE 5: Classification of a mixture for acute and chronic aquatic ecotoxicity based on summation of the concentrations of classified ingredients (source: UNEP 2017), containing mercury or mercury substances (M factor Acute and M factor Chronic = 100). In green: the lowest concentration limit.

Level of ecotoxicity	Ecotoxic Acute if sum of	Ecotoxic Acute (M=100) if sum of mercury and mercury substances	Ecotoxic Chronic if sum of	Ecotoxic Chronic (M=100) if sum of mercury and mercury substances
1	Acute 1 * M ≥25%	≥ 0.25%	Chronic 1 * M ≥25%	≥ 0.25%
2	(M*10*Acute 1) + Acute 2 ≥ 25%	≥ 0.025%	(M*10*Chronic 1) + Chronic 2 ≥ 25%	≥ 0.025%
3	(M*100*Acute 1) + (10*Acute 2) + Acute 3 ≥ 25%	≥ 0.0025% = 25 mg/kg	(M*100*Chronic 1) + (10*Chronic 2) + Chronic 3 ≥ 25%	≥ 0.0025% = 25 mg/kg
4	-	-	Chronic 1 + Chronic 2 + Chronic 3 + Chronic 4 ≥ 25%	≥ 25%

TABLE 6: Classification of mercury and mercury substances with the GHS in the EU, and corresponding concentration limits classifying a mixture containing mercury or mercury substances as hazardous. In color: the lowest concentration limits for physical (brown), health (blue) and environmental (green) hazards.

Hazard group		Physical		Health															Environmental	
Hazard Property		Explosive		Acute toxicity							Irritant			STOT*			Muta-genicity	Repro-ductive toxicity		Acute M=100	Chronic M=100
				Oral			Dermal		Inhal.					SE*	RE*						
Substances and hazard statement codes	CAS	H200	H201	H300 Cat 2	H301	H302	H310 Cat 1	H311	H330 Cat 2	H331	H314	H315	H319	H335	H372	H373	H341	H360	H361	Cat 1 H400	Cat 1 H410
Mercury	7439-97-6								X						X			X		X	X
Inorganic compounds of mercury with the exception of mercuric sulphide and those specified elsewhere in this Annex	-			X			X		X							X				X	X
Organic compounds of mercury with the exception of those specified elsewhere in this Annex	-			X			X		X							X				X	X
Dimethylmercury [1]; diethylmercury [2]	593-74-8 [1]; 627-44-1 [2]			X			X		X							X				X	X
Dimercury dicyanide oxide; mercuric oxycyanide	1335-31-5		X		X			X		X						X				X	X
Dimercury dichloride; mercurous chloride; calomel	10112-91-1					X						X	X	X						X	X
Mercury dichloride; mercuric chloride	7487-94-7			X							X				X		X		X	X	X
2-methoxyethylmercury chloride	123-88-6				X						X				X					X	X
Phenylmercury acetate	62-38-4				X						X				X					X	X
Phenylmercury nitrate [1]; phenylmercury hydroxide [2]; basic phenylmercury nitrate [3]	55-68-5 [1]; 100-57-2 [2]; 8003-05-2 [3]				X						X				X					X	X
Mercury difulminate; mercuric fulminate; fulminate of mercury	628-86-4	X			X			X		X						X				X	X
Mercury difulminate; mercuric fulminate; fulminate of mercury [≥ 20 % phlegmatiser]	628-86-4		X		X			X		X						X				X	X
Concentration limits = tresholds	Hazardous if	Presence	Presence	≥ 0.25%	≥ 5%	≥ 25%	≥ 0.25%	≥ 15%	≥ 0.5%	≥ 10%	≥ 1%	≥ 10%	≥ 10%	≥ 20%	≥ 1%	10%	≥ 1%	≥ 0.3% lowest	≥ 3%	≥ 0.0025%	≥ 0.0025%

** STOT SE = specific target organ toxicity – single exposure, STOT RE = specific target organ toxicity – repeated exposure*

TABLE 7: Distribution of Hg (in red: > proposed threshold of 25 mg Hg/kg), Cd and Pb in hazardous waste in France.

Element	Hg	Cd	Pb
Number of data	793	2266	2856
Mean	7121	1272	6961
Min	0.1	0.0	1.1
1%	0.3	0.5	6.3
5%	10	2	40
10%	10	4	69
25%	190	14	162
50% = median	836	55	1758
75%	5336	129	3464
90%	12228	394	8907
95%	16835	2481	29192
99%	121368*	9725	101794*
Max	544018*	728708*	550423*

** Probable analytical bias (X ray fluorescence analysis)*

TABLE 8: Concentration of Hg (mg/kg) in compost, fluvial and marine sediments, and agricultural soils (21 784 data) (in red: > proposed threshold of 25 mg Hg/kg).

Element	Compost	Fluvial sediments	Marine sediments	Agricultural soils
N samples	379	11 791	816	8798
Mean	0.39	1.22	0.50	0.08
Min	0.00	0.002	0.01	0.01
1%	0.02	0.005	0.02	
10%	0.09	0.03	0.05	0.02
25%	0.20	0.05	0.08	0.03
50% = median	0.30	0.13	0.13	0.05
75%	0.50	0.40	0.26	0.07
90%	0.80	1.30	0.53	0.11
99%	1.68	31.0	2.47	
Max	2.40	200	112	11.6

are in the Regulation relating to fertilizers (EC 2003).

4. CONCLUSION

The Globally Harmonized System of the UN classifies mixtures with concentration of mercury and mercury substances as hazardous (aquatic acute and chronic ecotoxicity of level 3) if their concentration is greater than or equal to 0.0025%, or 25 mg/kg. This concentration limit could be used as threshold for disposal of waste of category "c" according to the Minamata Convention.

The impact assessment shows that this concentration will not classify any of the common industrial waste or composts, sediments of soils to be managed specifically for mercury, but well the hazardous waste, that are already stabilized or solidified before landfilling in special landfills for hazardous waste, and very few (1%) contaminated fluvial sediments (that in all the case are also contaminated with other heavy metals), according to French data. It is also consistent with the present concentration limits set in the EU for reuse of municipal sewage sludge and other waste as fertilizing products.

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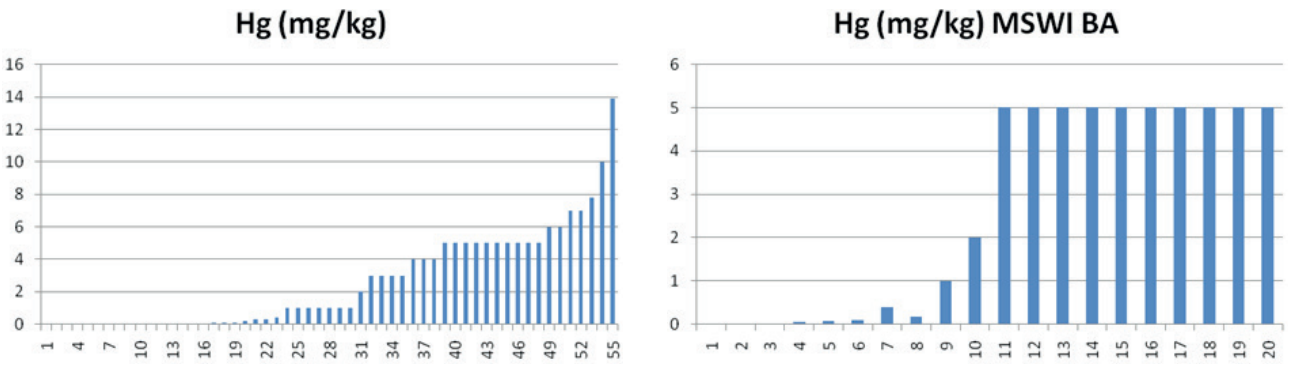

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ECOTOXICITY AND GENOTOXICITY OF STEEL SLAGS: PRELIMINARY RESULTS

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Article Info:
Received:
16 November 2018
Revised:
19 April 2019
Accepted:
7 May 2019
Available online:
19 May 2019

Keywords:
Steel slag
Biological assay
Chemical characterization
Ecotoxicity
Genotoxicity
Steel slag reuse

ABSTRACT

The European iron and steel industry produces a considerable amount of waste and by-products. Also in Italy the steel slag production is very high. Steel slags may be reused in recycled materials, such as materials for the construction industry, road base and asphalt mixtures, allowing to reduce the final disposal in a landfill. The reuse of this recycled material may generate potential release of toxic compounds for the environment and humans. The aim of this study was to assess the ecotoxicity and genotoxicity of steel slags by using an integrated chemical-biological approach. Chemical analysis of leachates obtained by using short-term leaching tests (UNI EN 12457-2) were performed, to evaluate the waste potential reuse according to the Italian legislation (Ministerial Decree 186/2006). Moreover, solutions obtained from leaching tests were assayed by using ecotoxicity tests on plant and animal organisms and genotoxicity tests on bacteria, plant and human cells. Chemical analyses of the eluates were within the limits of the Italian legislation. The preliminary results of the ecotoxicity and the genotoxicity tests demonstrated that this material has a low toxicity and therefore its potential use as a recycled material.

1. INTRODUCTION

The production chain of cast iron and steel generates scrap called steel slag (SS). Two types of slags are mainly generated from the iron and steelmaking industries: basic-oxygen furnace (BOF) steel slags and electric-arc furnace (EAF) steel slag. BOF-SS are residues from the basic oxygen converter, where steel is generated by the pig iron by injecting pure oxygen. EAF-SS are generated in high-power electric arcs where high quality steel is produced by melting recycled steel scrap (Chaurand et al., 2007; Yildirim & Prezzi, 2011). Thus, the production of SS is a result of the steel composition and of the steel production process (Yüksel, 2017).

In Europe, every year, more than 45 million tons of slag from both integral cycle furnace slabs and from electric furnaces are produced (European Slag Association and European Steel Association, 2012). The Italian production is about 6.5 million tons (Federacciai, 2012). According to the European regulations on wastes, this huge amount of material must be reduced in order to save primary resources and enhancedwaste disposal minimization. As a consequence, steel slag recovery is becoming a debated problem (Piatak, Parsons, & Seal, 2015).

Due to the mechanical and physical properties of steel, various applications can be implemented: in the construction industry, as in concrete (Rondi et al., 2016), or in road construction such as road base and asphalt mixtures (Sorlini, Sanzeni, & Rondi, 2012). Moreover, SS can be assimilated to natural hard rocks due to the composition similar to natural aggregates. In fact, SS are characterized mainly by oxides of calcium, iron, silicon, aluminum, magnesium, and manganese (Yüksel, 2017).

The classification of slag as a by-product or waste is variable within the European member states (Euroslag, 2018). The US Environmental Protection Agency classifies iron and steelmaking slags as non-hazardous considering their ignitability, corrosivity, reactivity and toxicity (National Slag Association, 1980). Despite that, according to steel production process characteristics (i.e. cooling conditions, blast furnace charges and temperatures), SS can also contain potentially toxic elements, such as chromium,

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Detritus / Volume 06 - 2019 / pages 32-38
https://doi.org/10.31025/2611-4135/2019.13815

molybdenum, vanadium (Komonweeraket, Cetin, Aydilek, Benson, & Edil, 2015; Primavera, Pontoni, Mombelli, Barella, & Mapelli, 2016; Tossavainen et al., 2007). Moreover, the SS industry remains for a long time in the environment and therefore has a prolonged contact with environmental mixtures such as deep water and soil. It may promote the release (for example through chemical weathering) of substances potentially dangerous both for the environment and for human health (Primavera et al., 2016).

Many mutagenic/genotoxic substances can be present and accumulate in the aquatic and soil environments, and have adverse effects on biocenosis and can affect humans through drinking water (Ceretti et al., 2016; Guan et al., 2017), surface water (Ohe, Watanabe, & Wakabayashi, 2004) and the food chain (Hamilton, Young, Bailey, & Watts, 2018). Consequently, the diffusion of these substances in water and soil could became a public health problem (Drzeżdżon, Jacewicz, & Chmurzyński, 2018; Rashtian, Chavkin, & Merhi, 2019).

The increasing presence of mutagenic/genotoxic pollutants in the environment has caused concern regarding the potentially harmful effects of xenobiotics on human health. Mutagenic compounds are very dangerous pollutants, since their effects may induce damage on an individual level as well as its progeny, throughout generations. Moreover, mutagenicity is related to carcinogenicity: the evidence of mutagenic activity suggests that a substance might be carcinogenic (Bajpayee, Pandey, Parmar, & Dhawan, 2005; Eastmond et al., 2009).

To date, Italian law requires the chemical analysis of these materials, by using short-term leaching tests in the case of re-use as supplementary cementitious materials or aggregates in concrete (Ministerial Decree n. 186, 2006). However, short-term biological tests allow an overall assessment of the analysed samples, that detect synergistic effects of complex mixture components and also predict the risks for the environment and for human health (Eastmond et al., 2009; Kirkland, Aardema, Henderson, & Müller, 2005). In particular, ecotoxicity, mutagenicity and genotoxicity tests are useful tools to predict and prevent risks due to the presence of toxic substances in the environment. The application of these assays in different cells/organisms allows for the assessment of mutagenic hazards for the environment and for humans (Escher & Fenner, 2011; Kirkland et al., 2005).

The aim of this study was to assess the ecotoxicity and genotoxicity of steel slags using an integrated chemical-biological approach, schematically summarized in Figure 1.

2. MATERIALS AND METHODS

2.1 Materials

Four samples of steel electric arc furnace (EAF-C) slags collected from different northern Italian steel-making plants were employed in this study. All the samples were stored from the factories in open areas and each sample collected was considered representative of the whole material storage.

Particle size ranged from 1 to 5 cm, whereas the density ranged between 2.5 and 3.3 g/cm^3.

2.2 Methods

Chemical analysis and biological assays were performed for all the samples.

Leaching tests according to the legislation (Standard CEN EN 12457-2, 2004) were performed on grinded steel slag samples.

Leachate solutions were assayed by using ecotoxicity tests on plant and animal organisms and genotoxicity tests were done on bacteria, plant and human cells.

In particular, the toxicity was tested through the investigation on the crustacean *Daphnia magna* (Standard UNI EN ISO 6341, 2013) and the common onion *Allium cepa* (G Fiskesjö, 1995; Ma et al., 1995). Mutagenicity in bacteria was evaluated by Ames test (Maron & Ames, 1983) and genotoxicity in plant cells was assessed by using micronuclei test in *Allium cepa* (Ma et al., 1995). Moreover, the micronuclei frequency in human leukocytes was studied (Fenech, 2000).

2.2.1 Leaching tests

Leaching tests were performed on naturally dried slag samples according to the CEN regulations (Standard CEN EN 12457-2, 2004), in order to quantify the leachable fraction of the wastes in water. Tests were performed by mixing the homogenized sample with demineralized water at a liquid to solid ratio of 10 l/kg. The mixture was placed on a tightly closed rotary shaker and agitated for 24 h, rotating at 10 ± 2 rpm. All the steel slags were previously subjected to a crushing treatment and a sieving process in order to obtain a particle diameter below 4 mm and the temperature was in compliance with the Directive (about 20°C). The solutions were filtered through 0.45 µm filters.

Chemical analyses of the leachate solutions were performed according to the Italian legislation for the recovery of non-hazardous waste (Ministerial Decree n. 186, 2006). Nitrates, fluorides, sulphates, and chlorides were analyzed through ion chromatography (ICS 1000, Dionex). Metals were measured by means of an optical plasma spectrometer (Optima 2000 DV, PerkinElmer) and cyanides were detected by using the colorimetric method DIN 38 (Nanocolor 400D).

2.2.2 Daphnia magna ecotoxicity test

A preliminary short-term acute test was performed on eluates obtained from the leaching tests using Daphtoxkits F (Ecotox LDS). The assays were conduced according to the European Directive (Standard UNI EN ISO 6341, 2013) and by following the manufacturer's instructions.

As a first step, in order to assess the toxicity of each sample, eluates were tested without any dilution. Totally, 20 neonates of *D. magna* (<24-h-old) were used for each slag leachate solution. Effects on crustacean movements or death were observed after 24 and 48 h of contact.

Eluates that resulted more toxic, were diluted (100%, 50%, 25%, 12.5%, 6.25%) in order to assess the lowest toxic concentration. For each dilution, 20 neonates were observed after 24 and 48 h of incubation.

Standard freshwater was used to dilute eluates and as well as a negative control in every test.

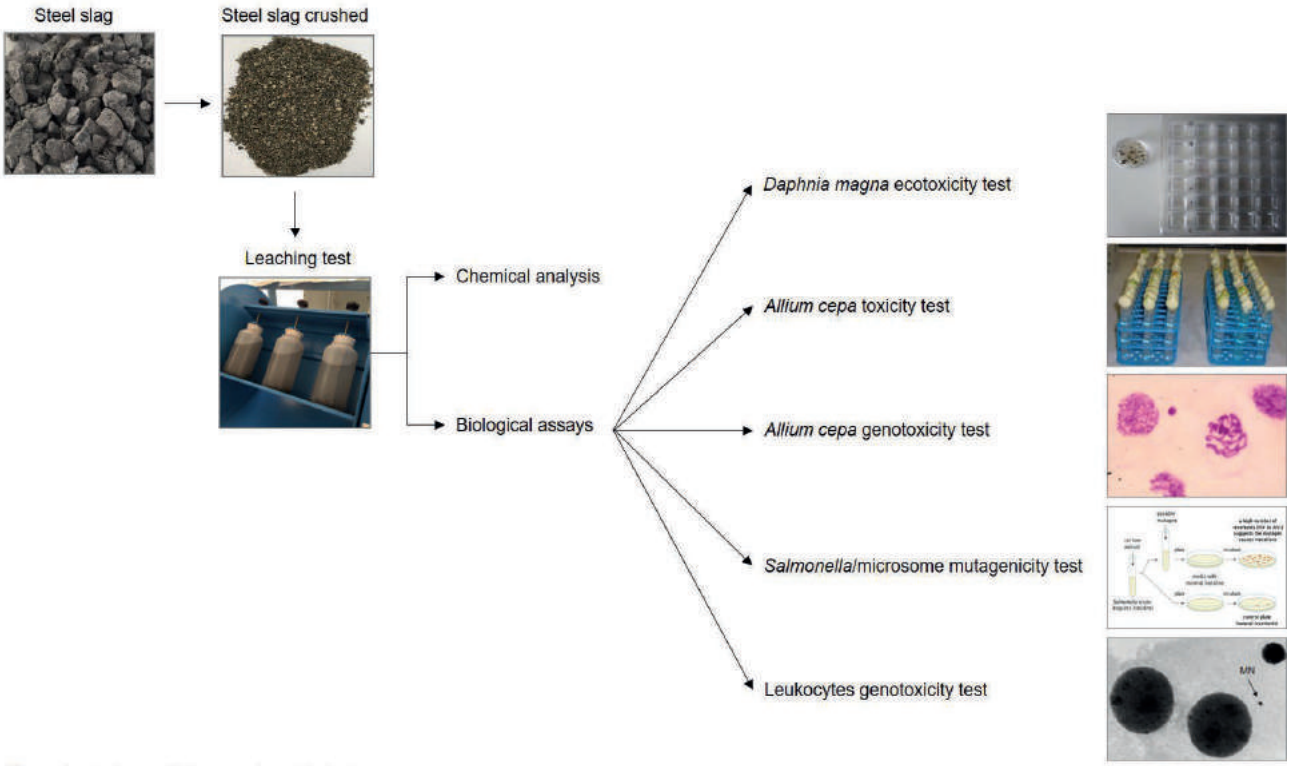

FIGURE 1: Scheme of the experimental design.

2.2.3 Allium cepa toxicity test

In a toxicity assay, 12 equal-sized commercial onion bulbs of A. cepa were cleaned and washed without destroying the primordial roots. They were exposed for 72 h in the dark to different dilutions of leachates (undiluted, 1:2, 1:10, 1:20, 1:100, 1:200 and 1:1000). Distilled water was used to dilute the samples and as a negative control. The roots mean length was used to calculate the EC50 value (Fiskesjö, 1985; Fiskesjö, 1995). Turgescence, consistency, colour change and root tip shape were also used as toxicity indexes.

2.2.4 Allium cepa genotoxicity test

The micronuclei (MN) *Allium cepa* test was performed by using six equal-sized young onion bulbs (2–2.5 cm in diameter) per sample. After 48-h pre-germination in a saline solution, the bulbs were exposed for 24 h to undiluted samples based on the results obtained in the toxicity test. They were then replaced in the saline solution (Rank's solution) for 44 h of recovery time. The roots were cut, fixed in 1:3 acetic acid-ethanol for 24 h and finally stored in 70% ethanol. Staining with acetic orcein was carried out on roots. Rank's solution was used as a negative control. A positive control was performed using maleic hydrazide (10^{-2} M, 6-h exposure) to ensure the effectiveness of the assay. Five roots of each sample were considered for the microscopic analysis (1,000X magnification): 5,000 cells (1,000 cells/slide) were scored for the mitotic index (MI), as a measure of the cellular division and therefore of sample toxicity; 10,000 cells (2,000 cells/slide) were scored for the MN frequency. All the *Allium cepa* experiments were performed in duplicate.

2.2.5 Salmonella/microsome mutagenicity (Ames) test

The leachates from steel slags were tested at increasing doses (2.5, 25, 125 and 250 µl/plate) using *Salmonella typhimurium* TA98 and TA100 strains, with and without *in vitro* metabolic activation (±S9) to detect direct and indirect mutagenic compounds. The experimental procedure was the standard plate incorporation method (Maron & Ames, 1983). The Salmonella TA98 strain detects frame-shift mutagens and the TA100 strain responds to base-pair substitution. Positive controls were 2-nitrofluorene (10 µg/ plate; more than 1,000 revertents/plate) and sodium azide (10 µg/plate; more than 1,000 revertents/plate) for the TA98 without S9 and the TA100 without S9, respectively, and 2-aminofluorene (20 µg/plate; more than 1,000 revertents/plate) for both strains with S9. The distilled water was a negative control. The results were espressed as a mutagenicity ratio (MR) obtained by dividing the revertants of each sample (computed by means of three replicates) by the spontaneous mutation rate (negative control). The test results were considered positive if two consecutive dose levels or the highest non-toxic dose level produced a response at least twice that of the negative control and at least two consecutive doses showed a dose-response relationship (APHA, 2012; Mortelmans & Zeiger, 2000).

2.2.6 Leukocytes genotoxicity test

MN frequency is a biomarker of early genetic effects which is often used in human biomonitoring studies. The MN frequency analysis was obtained with the cytokinesis-block micronucleus (CBMN) assay. The CBMN assay was performed according to the standard protocol de-

scribed by Fenech (Fenech, 2000). Briefly, whole blood was collected from a healthy donor and maintained at 37°C in darkness. Cultures were set up by adding 0.2 ml of whole blood to 2 ml of RPMI-1640 medium (Gibco), supplemented with 10% fetal bovine serum (FBS), 1% phytohaemagglutinin (PHA), 1% glutamine and antibiotics (penicillin: 10,000 U/ml; streptomycin: 10,000 g/ml (Biochrom)). After 24 h of PHA stimulation, leaching solutions were added and after 24 h of exposure, cytochalasin B (Sigma, Steinheim, Germany) was added at a final concentration of 4.5 µg/ml. Cells were harvested at the 72th hour after hypotonic treatment (0.5% KCl, 7 min) and fixed with the Carnoy's fixative 3:1 (methanol:acetic acid) with multiple changes till a clear pellet was obtained. Slides were prepared by the air-dry method and stained with 5% Giemsa for 20 min. With the CBMN assay, the frequency of MN in binucleated cells was measured. In this preliminary study, only the results obtained from the single dose of 2.5 µl of leachates are reported.

3. RESULTS AND DISCUSSION

The steel slag composition is reported in Table 1, in terms of leachable fraction derived from chemical analysis. All the four steel slag samples were characterized by similar concentration of the analyzed compounds. pH values indicated high alkalinity, as seen in the literature (Astrup, Mosbaek, & Christensen, 2006; Gomes et al., 2017). Sample A evidenced higher contents of nitrate, chloride, barium and vanadium with respect to other samples whereas sulphate concentrations were higher in sample B. Vanadium is a common critical element in steel slag, because it is a transition metal typically used in the metal alloy industries and can generate carcinogenic effects in humans (Gomes et al., 2017; IARC, 2006). Sulphate concentrations may affect the possible reuse of this waste in the building field from a mechanical point of view. Indeed, cracks and expansion problems can appear in concrete under sulphate attack (Ali et al., 2011). Another potentially toxic element commonly present in the steel slag leachate solution is chromium (Sas, Głuchowski, Radziemska, Dzięcioł, & Szymański, 2015). Sample C presents the highest concentration of this element (29 µg/l), although it is significantly below the limit value of 50 µg/l. Overall, the chemical analysis of all eluates obtained in our study were within the reference values required by the Italian decree for the recovery of waste (Ministerial Decree n. 186, 2006).

In this study, solutions obtained from leaching tests of four steel slags were assayed by using ecotoxicity tests on plant and animal organisms and genotoxicity tests on bacteria, plant and human cells.

Toxicity tests on steel slag leaching solutions through *D. magna* and *A. cepa* were performed.

Data obtained from the *Daphnia magna* toxicity test are summarized in Table 2. The validity criteria of the assay was respected, where a percentage of immobilization of the controls below 10%.

Preliminary results on crustacean immobilization using undiluted SS leaching solutions revealed differences among the samples: neither after 24 hours nor after 48 hours of observation, did samples A and B show toxic effects. Sample C resulted toxic with a mortality rate of 100% after 24 hours. Sample D showed an increase of immobilization from 30% (after 24 hours) to 75% (after 48 hours). According to the European Standard (UNI EN ISO 6341, 2013), a more detailed test was carried out on samples C and D using different dilutions. Sample C caused a high level of immobilization (90% after 48 hours) even at the lowest concentration (6.25%), while sample D showed a clear dose-response relationship between the sample concentration and Daphnia neonate immobilization. Immobilization by samples C and D could have been due to high pH values. As indicated by the European Standard UNI EN ISO 6341, tests should be performed without pH modification because "an adjustment of pH can alter the nature of the sample" (UNI EN ISO 6341, 2013). However, if toxic effects are observed and the pH of the tested solution is outside the range of the organism survival, experiments can be performed by modifying the pH with no more than 5% of chemicals (UNI EN ISO 6341, 2013). Further investigations are planned to clarify this point.

The *Allium cepa* toxicity test revealed the absence of toxicity in *Allium cepa*. In fact, all samples (both undiluted and diluted) did not influence the length of the roots. Moreover, no other sign of toxicity, as turgescence, consistency, color change and root tip shape, was observed in macroscopic parameters. Due to the absence of toxicity in roots, the undiluted samples were assayed in the *Allium cepa* genotoxicity test.

The results of *Allium cepa* genotoxicity test are reported in Table 3. The mitotic index (MI) showed that the eluates did not negatively influence cell division: MI values of samples are very similar to the negative control value and this allowed to consider the data of the micronuclei. No sample induced a statistically significant increase in the frequency of MN, thus highlighting the absence of genotoxicity in these cells.

Results of the Ames test by using *Salmonella typhimurium* TA98 and TA100 strains expressed as a mutagenicity ratio (MR) are reported in Table 4. No mutagenic activity was found for the SS eluates sample at all doses tested.

Table 5 reports the results of the preliminary micronucleus (MN) test performed at the dose of 2.5 µl of SS leachates. Data are expressed as the frequency of mononucleated (MONO), binucleated (BN), multinucleated (POLY) and the number of MN in each binucleated cell (MN/BN). Since the MN/BN was below 3, which is the cut-off number for the genotoxicity identification, no genotoxicity effects were detected at the tested dose.

The limited number of samples did not allow the authors to perform a detailed statistical analysis.

To date little is known about the toxicity of substances potentially released by steel slags (Suh et al., 2014; Radić et al., 2013), moreover there are no studies about genotoxicity of these materials.

For these reasons, we focused on the integration of chemical analysis, requested and codified by the legislation (Ministerial Decree n. 186, 2006), with biological assays to describe in detail the effects of steel slags on the environment and in humans. This combined procedure rep-

TABLE 1: Chemical characterization of leachates from steel slag. n.d: not detected.

		Sample				
Parameters	**M.U.**	**A**	**B**	**C**	**D**	**Limit value DM 186/2006**
Concentration						
Nitrate	mg/l	5.0	1.1	1.7	1.9	50
Fluoride	mg/l	0.21	0.26	0.31	0.64	1.5
Sulphate	mg/l	1.0	26.2	7.0	9.7	250
Chloride	mg/l	9.2	5.7	4.8	5.8	100
Cyanide	mg/l	< 5	< 5	< 5	< 5	50
Barium	mg/l	0.27	0.16	0.17	0.23	1
Copper	mg/l	< 0.01	< 0.01	< 0.01	< 0.01	0.05
Zinc	mg/l	< 0.01	< 0.01	< 0.01	< 0.01	3
Beryllium	µg/l	< 10	< 10	< 10	< 10	10
Cobalt	µg/l	< 10	< 10	< 10	< 10	250
Nichel	µg/l	< 10	< 10	< 10	< 10	10
Vanadium	µg/l	228	157	68	189	250
Arsenic	µg/l	< 5	< 5	< 5	< 5	50
Cadmium	µg/l	< 4	< 4	< 4	< 4	5
Total Chromium	µg/l	< 10	< 10	29	12	50
Lead	µg/l	< 10	< 10	< 10	< 10	50
Selenium	µg/l	< 10	< 10	< 10	< 10	10
Mercury	µg/l	< 1	< 1	< 1	< 1	1
Asbestos	mg/l	n.d.	n.d.	n.d.	n.d.	30
COD	mg/l	< 15	< 15	< 15	< 15	30
pH	-	10.2	10.5	11.3	10.9	5.5-12

TABLE 2: Results of *Daphnia magna* toxicity test on undiluted solution (samples A and B) and on undiluted and diluted samples (samples C and D).

		Immobilization %	
Sample	**Concentration %**	**24h**	**48h**
A	100	0	5
	control	0	0
B	100	0	0
	control	0	0
C	100	95	100
	50	100	100
	25	100	100
	12.5	90	100
	6.25	10	90
	control	0	0
D	100	30	75
	50	5	20
	25	10	10
	12.5	0	5
	6.25	0	0
	control	0	0

Control solution: standard freshwater

TABLE 3: Results of *Allium cepa* genotoxicity test: mitotic index (MI) and frequency of micronuclei (MN) in *Allium cepa* roots.

Sample	**MI** (%)	**MN** (mean ± SD)
A	12.7	1.0 ± 2.2
B	12.7	1.4 ± 0.9
C	11.8	2.8 ± 3.7
D	13.0	2.0 ± 1.7
Negative control	11.4	0.6 ± 1.3
Positive control	8.1	19.3 ± 14.2

resents an innovative methodological approach that could be improved in the standard procedure for the analysis of SS slag destined for re-use.

In conclusion, chemical composition of all the leaching solutions assayed respects the limits of the Italian legislation on waste recovery. The preliminary results of the *Daphnia magna* test on the leaching solutions of these wastes indicate in general a low inhibition of mobility in the majority of the samples. However, since one sample resulted very toxic for these crustacea, more detailed experiments are in course regarding the effect of different pH and dilution factors of the solutions.

All samples showed the absence of toxicity (length of roots) and genotoxicity (MN frequency) in the root cells in

TABLE 4: Results of Ames test using *Salmonella typhimurium* TA98 and TA100 strains, expressed as mutagenicity ratio (RM).

Sample	Dose (µl/plate)	TA98 - S9 RM	TA98 + S9 RM	TA100 - S9 RM	TA100 + S9 RM
A	250	0.94	1.22	0.75	0.65
	125	1.34	1.16	0.61	0.76
	25	0.90	1.22	0.52	0.69
	2.5	1.16	1.30	0.66	0.82
B	250	0.71	0.84	0.88	1.07
	125	1.19	0.90	1.02	1.07
	25	1.21	1.13	0.90	0.90
	2.5	0.81	1.18	0.83	1.00
C	250	0.77	1.09	0.88	0.98
	125	1.13	1.43	0.94	0.77
	25	0.84	1.48	0.98	0.86
	2.5	1.06	1.24	0.90	0.85
D	250	0.69	1.36	0.69	0.91
	125	0.88	1.39	0.87	0.97
	25	1.17	1.09	0.86	0.93
	2.5	0.73	1.16	0.97	0.96

Negative controls are expressed as revertants/plate: 26.0±8.9 (TA98 – S9), 36.8±6.2 (TA98 + S9), 109.8±11.2 (TA100 – S9), 116.8±11.3 (TA100 +S9).

TABLE 5: Micronucleus test in human cells: frequency of mononucleated (MONO), binucleated (BN), multinucleated (POLY) and the number of binucleated cells with micronuclei (MN/BN).

Sample	Dose (µl)	MONO	BN	POLY	Total cells	MN/BN
Control	2.5	725	165	110	1000	0
Ctrl+ H_2O	2.5	747	157	96	1000	1
A	2.5	776	114	110	1000	3
B	2.5	779	124	107	1000	2
C	2.5	770	128	102	1000	3
D	2.5	769	121	110	1000	1

the *Allium cepa* test. In line with our results, a pot experiment on growth of maize demonstrated the usefulness of EAF steel slag as a non-phytotoxic nutrient supplier (Radic et al., 2013). Moreover, no mutagenic activity was observed in bacteria with the Ames test.

Regarding the biological assay on human cells, the presence of MN in binucleated cells within the limits (normal range 0-3; > 3 genotoxicity index), demonstrated that the cultured cells did not undergo genotoxic damage from the tested leachates. Further concentrations will be evaluated. Also Suh and coworkers characterized the potential toxicity of EAF slag using an *in vitro* human dermal model, discovering that EAF slags were not a dermal sensitizer (Suh et al., 2014).

4. CONCLUSIONS

Our investigation is still preliminary, but results of genotoxicity tests indicate a low toxicity of steel slags. Naturally, it is necessary to further investigate the genotoxicity in other types of cells, especially mammalian cells. Due to the very low number of steel slag samples, a whole battery of biological tests will be repeated on a larger number of samples.

The reuse of steel slag wastes will undoubtedly have an environmental and economic advantages. Knowledge about the potential toxic element release and especially the effect it has on the human and environmental health, is therefore fundamental in finding new applications for steel slags.

AKNOWLEDGEMENTS

The authors would like to thank the PBR s.r.l., Maclodio - Brescia, Italy, for the chemical characterization of the slag samples.

This work was supported by the "Health and Wealth Strategic Programme" of the University of Brescia.

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CISA

INFLUENCES AND CONSEQUENCES OF MECHANICAL DELABELLING ON PET RECYCLING

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Article Info:
Received:
17 January 2019
Revised:
26 April 2019
Accepted:
3 May 2019
Available online:
24 May 2019

Keywords:
PET recycling
Sensor-based sorting
Label
Roughness
NIR Spectroscopy

ABSTRACT

The recycling of polyethylene terephthalate (PET) is an important issue of today's society. Mechanical recycling makes more sense from an ecological point of view than chemical PET recycling. However, mechanical recycling still is highly susceptible to defilements. Therefore, intensive pre-treatment is necessary to ensure the mechanical production of high-quality recycled PET. An important step in this process is to separate the PET bottles from their labels/sleeves. For this purpose, a newly developed label remover was studied. In this study, it was found that the machine had a delabelling efficiency of 90 w%. The PET bottles that were not sufficiently delabelled (10 wt.%) on average had a significantly smaller bottle size. This means that a sharp screening step, prior to delabelling, could improve the delabelling efficiency furthermore. Additionally, the applicability of near-infrared sorting technology was tested to find out, whether it can be used for quality control. Tests showed that state-of-the-art technology could differentiate between labelled and delabelled PET bottles, enabling separation of labelled PET bottles from delabelled bottles via sensor-based sorting. Hence, the proportion of contaminated PET bottles could be reduced furthermore with additional processing steps.

1. INTRODUCTION

Polyethylene terephthalate (PET) is one of the most common and prevalent thermoplastic polymers in today's society. It is used for the production of beverage bottles, fibres, moldings, sheets and other packaging material. Especially its worldwide usage as a container for beverages can be explained by the, in comparison to other plastic types, superior properties such as chemical, physical, mechanical, oxygen and carbon dioxide barrier features. The high clarity of PET constitutes a major advantage in comparison to many other packaging polymers. These properties contributed to the increased consumption of PET since the 1950s (Shen et al., 2010; Burat et al., 2009; Welle, 2011).

Due to the high quantities of PET bottles, this material presents a significant amount of today's waste. Since PET is not degradable under normal conditions and therefore occurs in aged waste excavated during landfill mining, expensive procedures would be needed in order to degrade PET biologically. In contrast, recycling processes constitute a relatively cost-effective method to reduce landfilling or incineration of PET waste. Therefore, its recycling is driven forward constantly (Awaja and Pavel, 2005).

Usually for recycling, first, mechanical pre-processing steps are applied to generate PET flakes that can be recycled chemically via depolymerisation or mechanically via extrusion. Chemical recycling offers the advantage that the recycled PET (RPET) has better properties than mechanically recycled PET, enabling a wide-ranging variety of possible applications. These superior properties come at the cost of a worse environmental profile of the chemical recycling process. During this process, the PET polymer is stripped down into monomers or oligomers using depolymerisation, resulting in an economically inferior process (Shen et al., 2010).

To receive better product qualities of mechanically manufactured recycled PET (RPET), the quality of their PET flakes must be improved. One of the main influencing factors on quality is the number of contaminants that enter RPET. These contaminants can be reduced by sorting out other materials, such as polyethylene (PE), polypropylene (PP) as well as metals. In order to separate PE and PP that are used for labels and sleeves from PET, pre-conditioning in form of delabelling can be necessary (Awaja and Pavel, 2005).

Especially due to marketing requirements, labels and sleeves become more popular and their size is often in-

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Detritus / Volume 06 - 2019 / pages 39-46
https://doi.org/10.31025/2611-4135/2019.13816

creased for promotional actions like enhancing packaging decoration. The variety of labels used on PET bottles is significant. Mainly low-density polyethylene (LDPE) and/or polyvinyl chloride (PVC) labels are used. Nevertheless, also labels and sleeves made out of 2-phenylphenol, polypropylene, and polystyrene can be found on the market. Such labels cannot only have an enormous effect on the quality of RPET but also affect the mechanical processing and sorting of PET bottles resulting in decreased machine efficiencies and recycling rates. If labels and sleeves are successfully removed from the PET bottles, they can be sold as by-products or be incinerated. The separation of labels/sleeves and bottles can also be accomplished by a washing process (Shen et al. 2010; Cotrep, 2012).

In this work, the separation efficiency of an innovative delabelling stage is tested and assessed at the pilot scale. Furthermore, its intelligent utilization in combination with sensor-based sorting machines is discussed. At last, the effects of the delabelling stage on the efficiency of downstream sensor-based sorting machines, applying near-infrared (NIR) technology, are studied.

2. PET RECYCLING - AN OVERVIEW

In order to recycle PET bottles, they have to be collected first. In Europe, this usually happens under schemes which follow the rule of producer responsibility. In some countries, PET bottles are collected within the household waste or via deposit-refund systems like in Germany. Either way, the collection of PET bottles is carried out on a local scale to transport the PET bottles to separation centres (Arena et al., 2003).

In waste separation centres, the bottles undergo several mechanical processing steps. Since the bottles often arrive in bales, a bale opener is used to disperse the bottles. Afterwards, either pre-washing or delabelling is necessary to remove labels and sleeves, enabling successful and efficient sorting of the bottles. In case of a washing step, an 80°C hot solution with 2% NaOH can be used. In the dry mechanical delabelling step, assessed in this study, mechanical friction is applied to tear the label or sleeve of the PET bottles (Awaja and Pavel, 2005).

The sorting of the material is often conducted via sensor-based sorting machines but can also be done manually. Magnetic and eddy current separators can be used to separate ferrous and non-ferrous metals. After separating undesirable materials and contaminants, the bottles can be sorted, e.g. according to their colour. At last the bottles are shredded into flakes, washed and have to be dried carefully. For the final washing step of the PET flakes, solvent washing with tetrachloroethylene is suitable. Since the minimization of the moisture content is most important to reduce hydrolytic degradation, the drying stage is essential after washing. Usually drying temperatures between 140 and 170°C, with a retention time between 3 and 7 hours are chosen in order to reach < 50 ppm water in PET flakes. To ensure the required purity of the PET flakes, a sensor-based sorting step might be necessary (Shen et al., 2010; Kranert, 2017; Awaja and Pavel, 2005; Assadi et al., 2004).

In this way, about 75 w.% of the baled PET bottles are processed to PET flakes and can be used for mechanical or chemical recycling. Losses occur during mechanical treatment, e.g. in the form of defilements, plastic and paper labels/sleeves, PE-/PP-caps and metals. 11-14 w.% of these fractions can be sold as by-products (PE caps, PVC/LDPE sleeves, etc.) while 14-18 w.% resemble solid waste and have to be treated furthermore (Shen et al., 2010).

The described mechanical pre-processing steps are necessary to prepare the PET for its further processing. Especially the quality characteristics of PET flakes must be achieved to ensure successful mechanical recycling. In Table 1, the minimum requirements for RPET flakes are given.

The degradation of RPET is increased by contaminants such as polyolefins or PVC, causing a reduction of the molecular weight and intrinsic viscosity of PET. This leads to a deterioration of the RPET properties. Reinforcing fillers and toughening modifiers then have to be applied to counteract the drop in molecular weight. (Srithep et al., 2011; Awaja and Pavel, 2005)

Once the minimum requirements for RPET flakes are met, they can be converted to granules or finished products at 280°C via melt extrusion. In comparison to chemical recycling, extrusion is a relatively simple, environmentally friendly and cost-effective process. However, to reduce the main disadvantage of mechanical recycling (reduction of molecular weight), mechanical processing must be improved (Shen et al., 2010).

In accordance with the topic of this study, a special focus lies on the influence of labels and sleeves on the recycling process of PET bottles despite their negative impact on RPET quality. During the sorting stage, labels and sleeves often remain on the PET bottles and can end up in the PET stream as well as in the PE or waste stream. Depending on the type of plastic used for the labels/sleeves, their thickness and size, PET bottles might not be identified correctly as PET and could be sorted out wrongly as undesirables. In this case, the PET yield would be significantly decreased since e.g. all full-sleeve PET bottles might be lost. Because of this reduction of the PET yield Cotrep (the technical committee for recycling of plastic packaging in France) recommends the use of partial labels and sleeves (Cotrep, 2012).

PVC labels are classified as unfavourable because PVC has a significant negative impact on RPET. It decomposes

TABLE 1: Minimum requirements for post-consumer-PET flakes to be reprocessed (Awaja and Pavel).

Property	Value
Viscosity [η]	> 0.7 dl g^{-1}
Melting point [Tm]	> 240°C
Water content	< 0.02 wt.%
Flake size	0.4 mm < D < 8 mm
Dye content	< 10 ppm
Yellowing index	< 20
Metal content	< 3 ppm
PVC content	< 50 ppm
Polyolefin content	< 10 ppm

during extrusion, clogs extruder fillers and causes further quality problems. Hence, if a PVC flake is detected in the PET flake stream, the separation of PVC has to be ensured. For a singular separated PVC flake up to 100 flakes are ejected. Because of this, more losses are generated and the amount of waste to be disposed of is rising (Cotrep, 2012). Cotrep recommends that labels and sleeves that are made out of polystyrene (PS) and PET-G should be substituted because they tend to deteriorate, form impurities (PS) and create yellowing (PS and PET-G) in RPET. Shrink LDPE labels are classified as favourable since they do not disrupt the recycling process significantly (Cotrep, 2012).

3. MATERIAL AND METHODS

PET bottles from a public collection system were obtained as input material for the delabelling trials. To generate reliable data, only empty bottles with fully attached labels were chosen for the trials. In total 98 kg of PET bottles with labels or sleeves were handpicked. An exemplary picture of the handpicked PET bottles is given in Figure 1. One can be seen that most of the bottles are deformed or crushed. The samples had a bulk density of around 50 kg/m³.

FIGURE 1: Input material for delabelling trials - PET bottles from the public collection system.

Delabelling trials were conducted with the "STADLER label remover" (max. throughput 8 t/h, dimensions 2,733 × 1,862 × 2,317 mm (L x W x H) stator diameter of 1,600 mm and drive power of 37 kW, rotor speed of 200 rpm) at the Stadler Technology Centre in Krško, Slovenia. As can be seen in Figure 2, the label remover is equipped with rotating arms that have jagged knives made from high-tensile steel. The length of these arms can be adjusted via slot holes. So, the distances between the knives on the rotating arms and the knives on the inner wall can be adjusted to fit the size of the input material. The general principle is that less space between the knives causes more delabelling at the risk of bottles being torn. Two types of knives are mounted to the inner wall:

- Vertically mounted knives
- Knives with an adjustable angle

The knives with adjustable angle enable the machine operator to modify the retention time of the input material: the more obtuse the angle, the longer the retention time.

For the trials, the input material was divided into two equally sized samples each weighing 49 kg. Two trials were run at a throughput of about 4 t/h. In the first and last seconds of each round, a continuous feed into the label remover could not be ensured. Particles at the beginning and the end of a round could falsify the results due to higher retention times. Therefore, only the delabelled product that was generated while a steady feed of the machine could be ensured was further studied. As a result of this approach, of the 49.0 kg input material per trial, 33.2 kg and 34.9 kg could be analysed respectively. It has to be mentioned, that

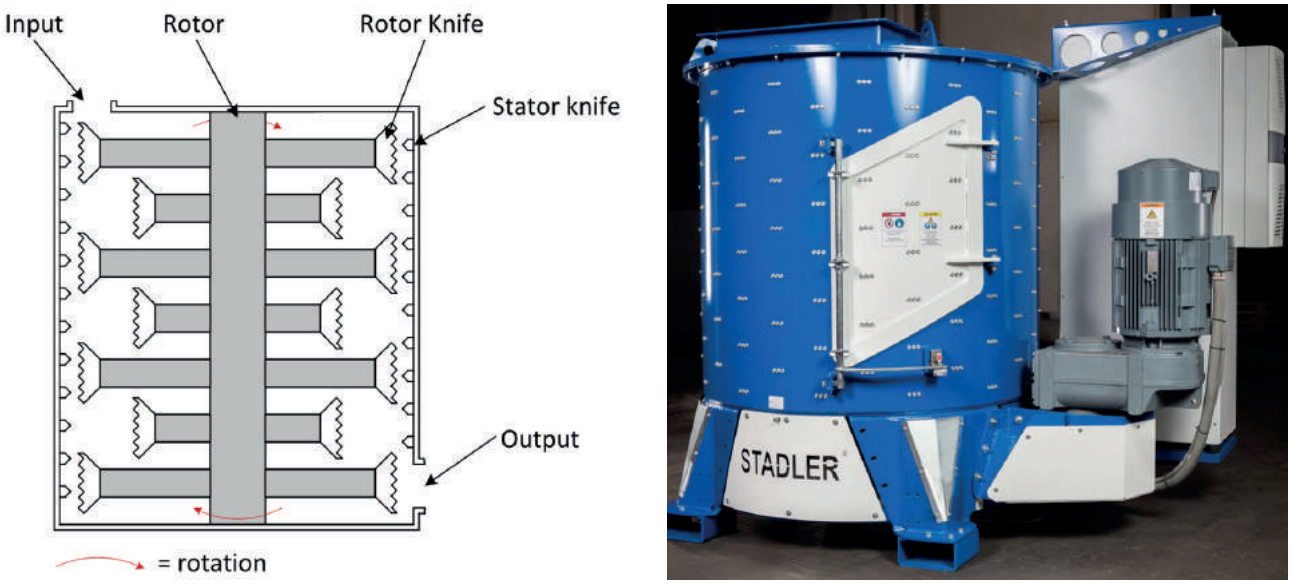

FIGURE 2: Scheme of the grinding chamber and picture of the "STADLER label remover".

the separated labels were not weighed after the trials since too many of them would remain in the delabeller or be lost throughout the trials to make sufficiently sound conclusions.

To evaluate the influence of the delabelling process, samples were taken before as well as after the trials and screened with a laboratory polygonal drum sieve. These screenings were conducted at a mesh size of 80 mm for 90 seconds since this is the typical screening time of packaging material in a technical drum screen of 10 m length (Go et al., 2018). The mesh size of 80 mm was chosen because this screen cut is used industrially to enrich PET bottles in the coarse fraction. PET bottles with a volume of 0.5 l and less can be lost into the fines. Therefore, the number of bottles in the coarse and fine fraction provides information about the predominant bottle size in the screened sample and potential shredding effects of the delabeller. Additionally, the delabelled bottles were sorted manually after the trials and divided into three different categories:

- Good: > 98% of the labels/sleeves were separated from the respective bottles (sufficient)
- Middle: 90-98% of the labels/sleeves were separated from the respective bottles (sufficient)
- Bad: < 90% of the labels/sleeves were separated from the respective bottles (insufficient)

The allocation of the delabelled bottles to these three categories was carried out by manual separation after the trials. Bottles that ended up in category 1 either contained no label at all or only small label pieces at the joins. Category 2 mainly contains bottles with label pieces on the joins. Bottles in category 3 primarily showed labels that were ripped open or sleeves that were sliced in pieces but not separated from the bottle. After the delabelling trials, samples of each category were taken and a screening analysis was conducted with a mesh size of 80 mm.

Before and after the delabelling process, samples of bottles were taken for further investigations with NIR (near infrared) technology. For these analyses, a sensor-based sorting machine from Binder+Co AG, equipped with a hyperspectral imaging (HIS) NIR sensor from EVK (HELIOS NIR G2 320) with a wavelength range from 950 nm to 1700 nm was used. Pictures of the samples, taken before and after the delabelling trials, were captured to analyse the raw spectra of the samples and to classify the different materials contained in the samples using state of the art algorithms. These algorithms consist of the processing steps given in Table 2.

For a classification of each object pixel, the y-values of each spectral band (width of one band is approx. 3.2 nm) were compared with the material specific spectral information implemented in the algorithm. This way, each pixel can be provided with a false colour and less computing power for the evaluation of each particle is necessary. Hence, the classification of each bottle can be performed.

TABLE 2: Preprocessing and spectral processing steps of spectra for classification.

Preprocessing	Spectral Processing
Spatial correction	1st Derivative
Bad pixel replacement	Normalization
Intensity Calibration	Smoothing
Noise suppression	

4. RESULTS AND DISCUSSION

The delabelling efficiency results from the composition of the output of the delabelling trials. The results are given in Table 3.

After visual inspection of the output, it could be found that about 90 wt.% of the bottles were delabelled sufficiently (60 wt.% Good, 30 wt.% Middle), meaning, the number of labels on PET bottles was reduced drastically. About 10 wt.% of the bottles were not delabelled successfully. The visual result can also be withdrawn from Figure 4.

An apparently large number of small bottles was sorted into category 3 (Figure 3). The visual observation can be confirmed with the results of the screening analyses presented in Figure 4. It can be seen, that compared to the

TABLE 3: Output composition - label remover.

	Good	Middle	Bad
Trial 1	62 wt.%	29 wt.%	9 wt.%
Trial 2	59 wt.%	33 wt.%	8 wt.%

FIGURE 3: Output fraction of the delabeller - from left to right: category 1 (Good), category 2 (Middle), category 3 (Bad).

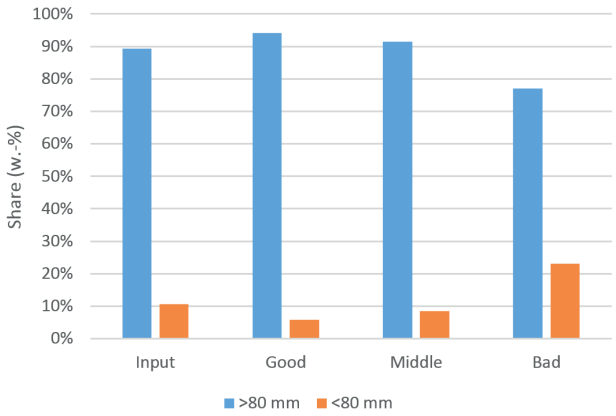

FIGURE 4: Results of screening analyses before and after delabelling.

input analysis, bottles in categories 1 and 2 (Good and Middle) show smaller amounts of material <80 mm (less than 10 wt.%) while category 3 contains more than 20 wt.% of bottles <80 mm.

It must be stated, that no shredded or compacted bottles were found. This suggests that most small bottles were delabelled insufficiently while most big bottles (>0.5 l) were processed successfully. The inverse conclusion of this is that a sieving step prior to the delabelling step would increase the efficiency of the delabeller furthermore, which is in accordance with findings of Go et al., 2018. Additionally, it must be mentioned that the input for the above-shown trials consisting of 100% labelled bottles is not the case in reality. This affects the quality of the output positively by increasing the percentage amount of label-free bottles in the output of the delabelling stage. Besides that, fully affixed paper labels underwent little to no change during the treatment. An example is given in Figure 5.

To determine the impact of labels and of the delabeller on the detection as well as classification of PET bottles, HSI NIR pictures of the bottles, prior and after delabelling, were taken. The different average spectra that were used to distinguish PET from PET covered with a label (PETL) and bottle caps are given in Figure 6. Significant differences between HDPE and the other spectra can be registered. To distinguish PET from PETL pixels, two different spectra for PETL had to be included due to variations concerning the intensity of the peaks, typical for PETL. Therefore, a

FIGURE 5: Impact of delabeller on the fully affixed paper label.

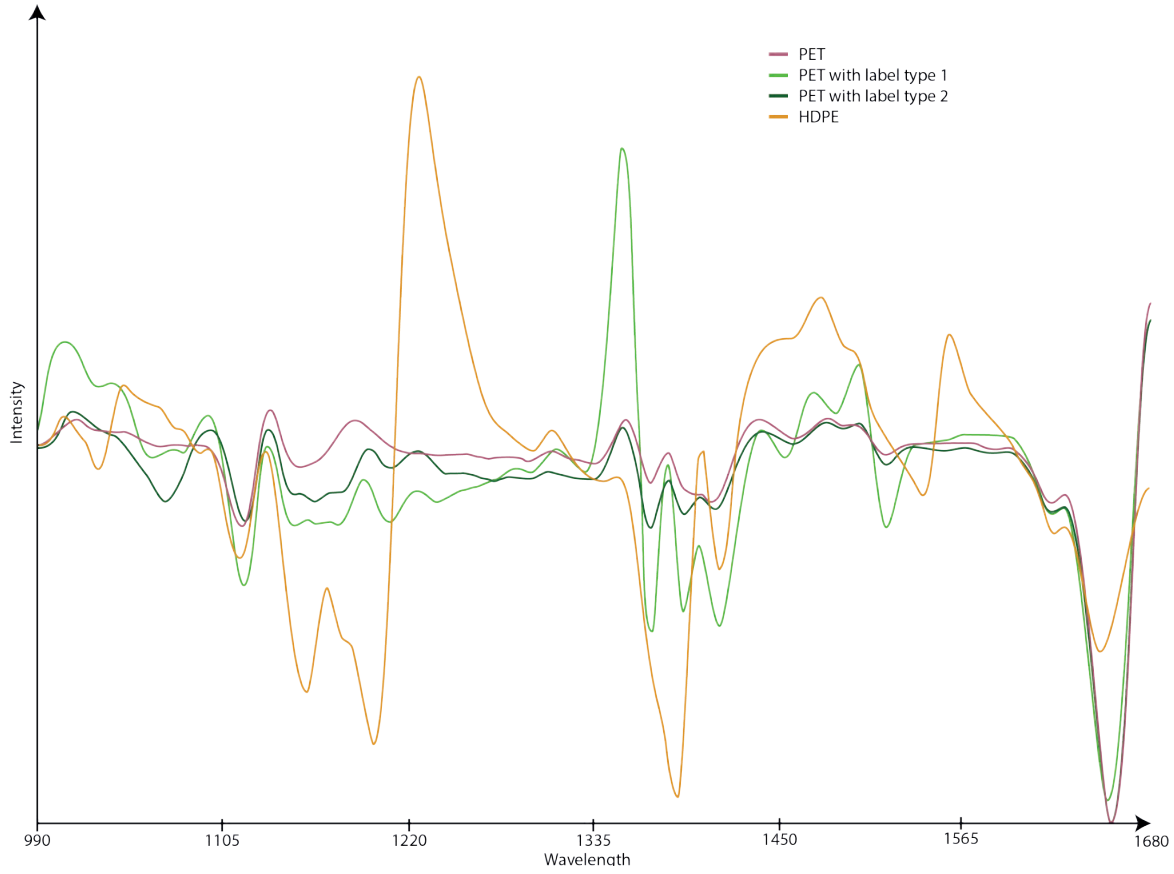

FIGURE 6: Qualitative spectral course (first derivative, normalized) of PET, PET with label type 1, PET with label type 2 and HDPE.

classificator with four different spectra was developed to distinguish between PET, PET with label and HDPE.

Examples for classified bottles are given in Figure 7. It can be seen that PET, HDPE and PETL can be distinguished from each other very well. It should be noted that even though some pixels of the fifth bottle were wrongly classified due to the influence of water, in this trial, all labelled bottles could be correctly classified as such.

To double-check the functionality of the created classificator, pictures of delabelled bottles were taken and classified as well. The result can be seen in Figure 8. All bottles are classified as not labelled PET and the caps (on bottles 1 and 4) are also correctly classified as HDPE. Only a few pixels on the edges of objects in Figure 7 and 8 are falsely classified as PETL due to edge effects. The amount of incorrectly classified pixels is insignificant and differentiation between PET bottles with and without labels can be expected.

Additionally, the extent of the PET spectrum before and after delabelling was analysed as well as the signal-to-noise ratio. In total 60,096 spectra were analysed for this purpose. The results are given in Figures 9 and 10. The spectra before and after delabelling are displayed. Apart from outliers (grey), it can be seen that 90% of the derived spectra (interquantile deviation) show significantly higher extents and marginally higher averaged standard deviations after the delabelling process than before. Prior to delabelling, the characteristic and most important absorption for classification of PET at a wavelength of about 1650 nm is barely noticeable let alone smaller peaks, e.g. between 1110 nm and 1180 nm. This complicates the classification significantly because the spectra have to be normalized for consistent sorting efficiency, which results in enhanced background noise.

Despite the fact that correct classification before and after delabelling is possible, mechanical treatment during label removal simplifies the classification and therefore enhances sorting of PET bottles. The trials showed that the differentiation between labelled and delabelled PET bottles is possible. This can be used for processes aiming for high product purities by installing a downstream sensor-based sorting unit after the delabelling step. The downstream sensor-based sorting unit separates the remaining labelled PET bottles from the delabelled bottles to recirculate them as input for the delabelling step once again.

5. CONCLUSION

For mechanical recycling of PET bottles with the aim of high-quality RPET production, the reduction of defilements is of utmost importance. An important part of this process is the separation of the labels and sleeves from the PET

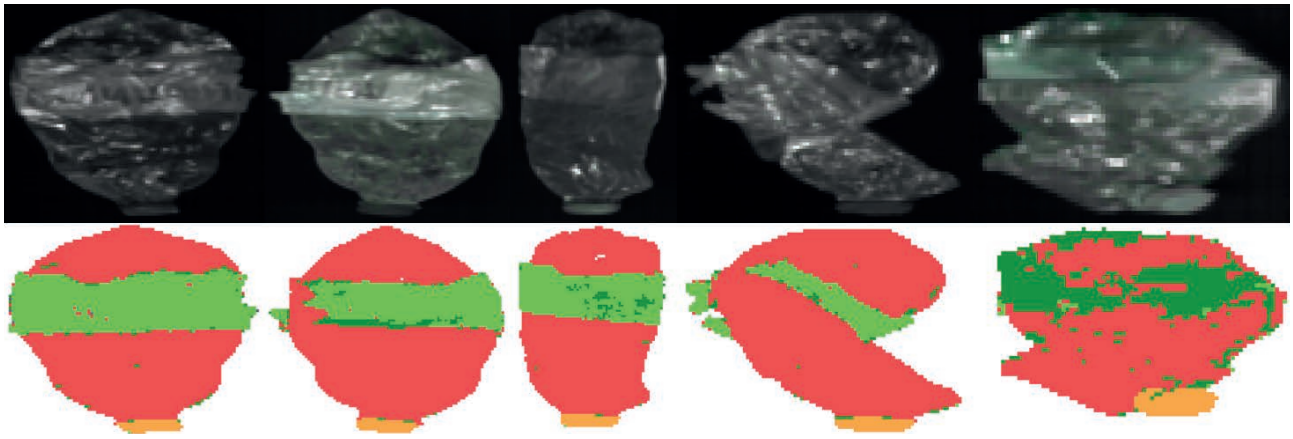

FIGURE 7: Comparison of live picture (upper row) and classified picture with false colours (lower row) of labelled PET bottles; red=PET, green=PETL, orange=HDPE.

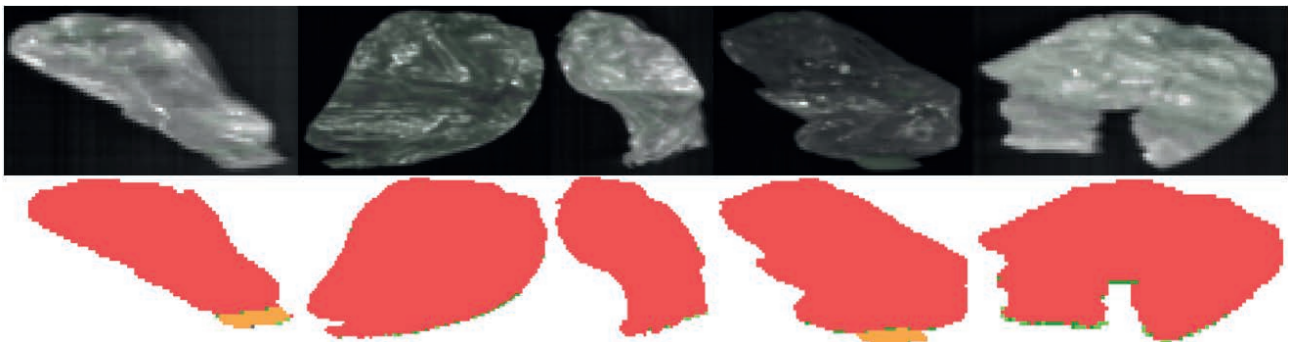

FIGURE 8: Comparison of live picture (upper row) and classified picture with false colours (lower row) of delabelled PET bottles; red=PET, green=PET with label, orange=HDPE.

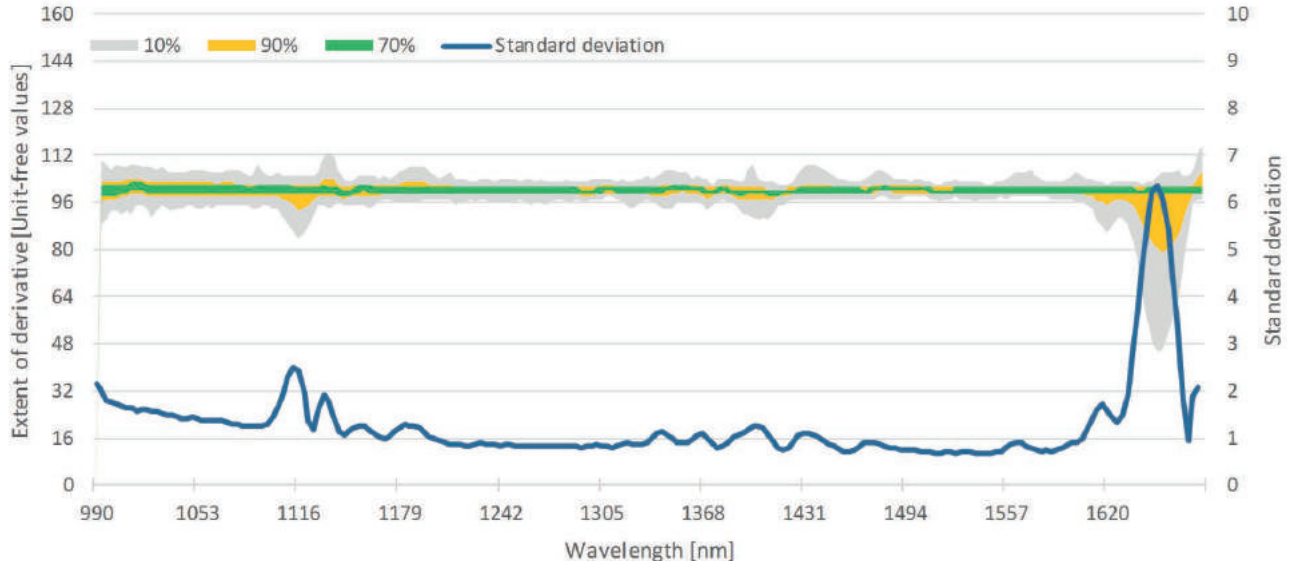

FIGURE 9: Interquantiles (90%, 70%) and outliers (10%) of derivatives basing on the raw spectra, recorded of PET pixels prior to delabelling (primary axis) and standard deviation of derivatives (secondary axis).

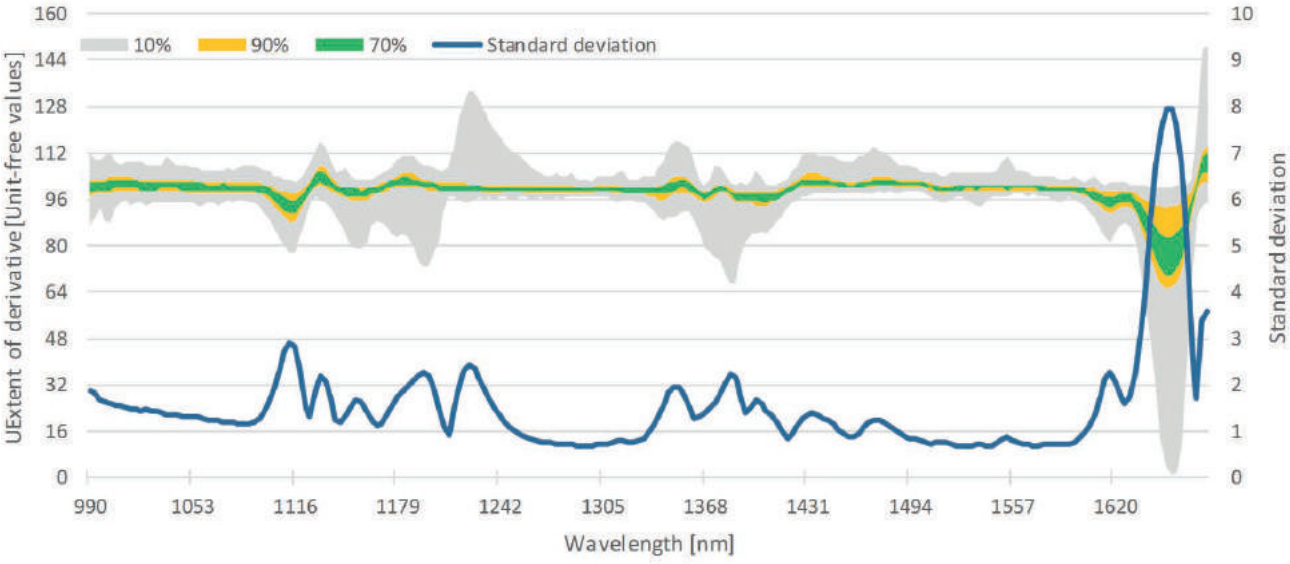

FIGURE 10: Interquantiles (90%, 70%) and outliers (10%) of derivatives basing on the raw spectra, recorded of PET pixels after delabelling (primary axis) and standard deviation of derivatives (secondary axis).

bottles. This can be achieved by the application of a mechanical delabelling step.

The studied "STADLER label remover" showed a delabelling efficiency of 90% at a throughput of about 4 t/h. It was found that the number of bottles unsuccessfully treated was strongly affected by the number of small bottles, <0.5 l filling volume. Therefore, in an industrial process, a screening step prior to delabelling would improve the efficiency of the delabeller furthermore.

Findings showed that the bottles were neither shredded nor significantly deformed during delabelling, enabling high efficiencies of downstream machinery, e.g. sensor-based sorting units. It was found that PET bottles with and without labels/sleeves could be classified and separated when applying HSI NIR technology. A sensor-based sorting unit could be installed downstream a delabeller to sort out PET bottles still containing labels, improving the purity of the PET stream. Additionally, it was found that the mechanical treatment roughens the bottle surface, resulting in an enhanced peak extension and, consequently, improved PET bottle classification.

ACKNOWLEDGEMENTS

This research has been funded by the European Union´s Horizon 2020 research and innovation programme under the Marie Sk³odowska-Curie grant agreement No. 721185 "NEW-MINE" (EU Training Network for Resource Recovery through Enhanced Landfill Mining; www.new-mine.eu). The authors wish to express their special gratitude to Renewi

Belgium SA/NV, Stadler Anlagenbau GmbH and the Department of Processing and Recycling (IAR) of the RWTH Aachen University for their straightforward collaboration and support.

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ACID HYDROLYSIS AS A METHOD TO VALORIZE CELLULOSIC FILTER CAKE FROM INDUSTRIAL CARRAGEENAN PROCESSING

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Article Info:
Received:
14 January 2019
Accepted:
17 June 2019
Available online:
28 June 2019

Keywords:
Acid hydrolysis
Acid sites
Carrageenan filter cake
Reducing sugar
Sulfonation

ABSTRACT

Wastes generated from carrageenan processing industry include cellulosic filter cakes (CFC) which are mainly composed of structural sugar (0.25 w/w) and ash (0.75 w/w, primarily perlite). This study investigated the possible valorization of CFC by recovering the available sugars as glucose through direct acid hydrolysis. Five different sulfuric acid concentrations (5% v/v to 15% v/v) were used as catalyst for hydrolysis done at constant temperature and solvent-to-solid ratio of 95°C and 8 mL/g, respectively, over a reaction time of 5 to 300 minutes, to determine the effect of acid concentration on the hydrolysis yield. The maximum sugar yield achieved was only ~0.06 w/w, corresponding to a recovery of ~24%, for hydrolysis done with 10% v/v sulfuric acid for 120 minutes. Although the amount of sugar recovered was relatively low, hydrolysates obtained have a sugar concentration of ~7 g/L, a level considered adequate for substrates in some fermentation processes. In addition, none of the inhibitory compound, 5-hydroxymethylfurfural, was present in the hydrolysate. Drying of residual solids obtained after hydrolysis was found to result in the sulfonation of the remaining organic fraction, producing a sulfonated residue (with total acid density of 4 to 7 mmol H+/g) which may be used as solid acid catalyst.

1. INTRODUCTION

Philippines leads the global carrageenan production with a production capacity of at least 36, 400 MT, sharing 40% of the total world production (Nobleza, 2013). Refined carrageenan is produced by alkali treatment of red seaweeds genera Gigartina, Eucheuma (previously Kappaphycus), Chondrus, and Hypnea, using sodium hydroxide or potassium hydroxide for carrageenan extraction, followed by separation of the cellulosic residue from the extract through filtration (Rhein-Knudsen et al., 2015; Stanley, 1987). During filtration, perlite is added at a ratio of 500 kg for every 1700 kg of raw seaweeds processed (Lhonneur, 1992), which results to about 31% of perlite in the cellulosic filter cake generated if assumed that the carrageenan extraction yield is ~35 kg per 100 kg of raw seaweed (Manuhara et al., 2016). Considering that the annual production of red seaweeds is ~1.6 million tons, with 22% processed to produce refined carrageenan (FAO, 2015; Philippine Bureau of Investments, 2011), there is about 16.2 ktons of perlite-containing cellulosic filter cake waste that needs to be disposed. A relevant action would be to find ways of valorizing said waste as an approach to solving the disposal problem and make possible a zero-waste carrageenan production industry.

Cellulosic biomass, like that of macroalgae for carrageenan proprduction are composed of cellulose, hemicellulose and lignin (Fan et al., 1987). However, compositional analysis done by Tan and Lee (2015) found that 68% of macroalgae carrageenan residue (MCR) is cellulose and has no acid insoluble lignin and hemicellulose, making MCR attractive for hydrolysis and subsequent fermentative processes, since delignification would not be required. Cellulose is composed of anhydrous glucose which are linked together by glycosidic bonds (Badger, 2002). Glucose is a useful fermentation substrate in the production of bioethanol and other high-value platform chemicals like 1,2-propanedieol (Douglas and Cooney, 1986), lactic acid and acetol (Hang, 1989), 2-keto-L-gulonic acid (Matsuda and Kageyama, 1982), and succinic acid (Li et al., 2010; Zheng et al., 2009). It can be recovered from cellulosic materials by breaking down β-1,4-glycosidic linkages through a process called hydrolysis (Kumar et al., 2009). However, the

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Detritus / Volume 06 - 2019 / pages 47-58
https://doi.org/10.31025/2611-4135/2019.13823

ease by which cellulose can be hydrolyzed is a function of its crystallinity and intermolecular bonding. The structure of cellulose can be grouped into six polymorphs namely cellulose I, II, III_1, III_{11}, IV_1, IV_{11} and VII. Only cellulose I is found in nature and is further classified into I_α and I_β (Rinaldi and Schüth, 2009). Cellulose I_α, a dominant structure in algal biomass, have a metastable triclinic structure, whereas, cellulose I_β which is commonly found in wood and cotton fibers have a stable monoclinic structure (O'Sullivan, 1997; Rinaldi and Schüth, 2009). In a triclinic structure, only one cellulose chain is present per unit cell whereas in monoclinic structure, there are two chains in each unit cell. The latter is more packed, resulting to more interactions between cellulose molecules. Thus, algal biomass have lesser degree of crystallinity and are relatively easier to hydrolyze than wood and cotton fibers (O'Sullivan, 1997). Hydrolysis of algal biomass can be enzymatically or chemically catalyzed.

Enzymatic hydrolysis uses a mixture of different types of enzymes called endoglucanase, exoglucanase, and β-glucosidase (Zhang and Zhang, 2013). Enzymatic hydrolysis of algal biomass have been investigated by Kumar et.al (2013), Trivedi et.al (2013) and Tan & Lee (2014). Masarin et.al (2016) and Tan & Lee (2014) particularly looked into the enzymatic hydrolysis of carrageenan residue. A mixture of enzymes (92 FPU/ml Cellic Ctec II and 1800 UI/ml β-glucosidase) were used by Masarin et.al (2016) to hydrolyze the Eucheuma cottonii extraction residue for 72 hours at temperature and pH of 45°C and 4.8, respectively, achieving a sugar yield of 65%w/w (g sugar per 100 g residue). Tan & Lee (2014) achieved ~64%w/w sugar yield from the hydrolysis of residue using an enzyme loading of 82.08 FPU/ml (Celluclast 1.5L) and 326.12 U/ml (Novozyme 188) for 54 hours at a temperature and pH of 50°C and 4.8, respectively. Although high sugar yield was achieved and no inhibitors (5-HMF and furfural) were formed during hydrolysis at relatively mild conditions, the long reaction times and the cost of enzymes are considered the main drawbacks to actual industrial application.

Chemical hydrolysis of cellulose is carried out in the presence of acids, commonly sulfuric acid, to catalyze the reaction. Depending on the acid used, hydrolysis reaction can be categorized as heterogeneous or homogeneous. Heterogeneous acid hydrolysis employs solid acid catalyst which can be recovered, regenerated and reused. A study was conducted by Tan and Lee (2015) on a two-step hydrolysis of carrageenan residue. The pre-treatment step done for 30 minutes and at 120°C employed the solid acid catalyst, Dowex ™ Gr-D8, at a solvent-to-solid ratio of 10 mL/g, and 4% w/v catalyst loading. The residue left after pre-treatment was then subjected to enzymatic hydrolysis which achieved a sugar yield of 67.8%w/w after 30 h. Although, the use of solid acid catalyst is a promising technology for sugar recovery because it allows catalyst reuse, its particular application in the hydrolysis of cellulosic filter cake poses a challenge because the recovery of the catalyst would be difficult due to the presence of perlite and residual solids.

Homogeneous acid hydrolysis employs acids in liquid state to catalyze the reaction. Several studies have been reported on the dilute-acid hydrolysis of algal biomass. Hydrolysis of K. alvarezii using sulfuric acid (0.2 M) and hydrochloric acid (0.2 M) at 10 mL/g solvent-to-solid ratio, temperature of 130°C for 15 minutes achieved a sugar yield of 38.5%w/w and 22.7%w/w, respectively (Dyah and Meinita, 2012). Hydrolysis of Gracilaria tenuistipitata, Gracilariopsis chorda, and Gelidium amansii using sulfuric acid at same operating conditions had a sugar yield of 26.6%w/w, 23.4%w/w, and 29.2%w/w, respectively (Meinita et al., 2013). Although acceptable sugar yield is achieved for dilute-acid hydrolysis, the main drawback for this technology is the further degradation of sugars to furans (ranging from 0.81 g/L (Dyah et al., 2014) to 5.9 g/L (Meinita et al., 2012)) which can inhibit the activity of microorganisms used in the subsequent fermentation process. Nonetheless, the process has the advantage of having relatively shorter reaction time and using less expensive catalyst, making it more economically attractive for producing reducing sugar from algal biomass.

Studies reported in literature on the acid hydrolysis of algal biomass focused mainly on raw seaweeds and macroalgae cellulosic residue (MCR), which does not contain the perlite or filter aids that are used in actual industrial process. Perlite is a glassy, volcanic rock (Maxim et al., 2014) mainly composed of silica (72%w/w) and alumina (13%w/w) (Samar and Saxena, Shweta, 2016). Considering that these are basic oxides, these may hydrolyze the acid used and may interfere the hydrolysis of CFC. Hence, this study aimed to produce reducing sugar-containing hydrolysates from CFC through hydrolysis using sulfuric acid as the catalyst. Specifically, the objectives of this study were to determine the effects of acid concentration (5% v/v to 15% v/v) and reaction time (5 to 300 minutes) on the yield and concentration of sugar during the hydrolysis of CFC at a constant temperature and solvent-to-solid ratio of 95°C and 8 mL/g, respectively. In addition, the amount of glucose degradation product (5-hydroxymethyfurfural) formed and the proximate composition of hydrolyzed CFC were also determined. Furthermore, the incidental carbonization and sulfonation of the solid residue after hydrolysis were investigated to look into its possible application as a solid acid catalyst.

2. MATERIAL AND METHODS

Cellulosic filter cake (CFC) samples were collected from a carrageenan processing company in Cebu, Philippines. Chemical reagents used were obtained through local distributors: sulfuric acid (Ajax Finechem, 98% w/w), hydrochloric acid (Ajax Finechem, 36% w/w), anhydrous sodium carbonate (Ajax Finechem, 99.8% w/w), sodium hydroxide pellets (Qualikems, 98% w/w),phenol crystals (Qualikems, 99.5%w/w), and acetonitrile (HPLC grade), glacial acetic acid, Rochelle salt (HiMedia Lab, 99% w/w), dinitrosalicylic acid (HiMedia Lab, 98% w/w), D-glucose, and 5-hydroxymethyl furfural (Sigma).

2.1 Collection, storage and characterization of carrageenan filter cake

Collected filter cake samples from a local carrageenan processing, with a wet, pale-white appearance, had an as received moisture content of ~89% w/w. These were dried

at 105°C in a convection oven (Memmert UM 500 F) until a moisture content of <10% was achieved. The dried samples were milled using Wiley mill, determined of its mean particle size following ASTM D1762-84 (ASTM, 2011) and then stored at room temperature (25 to 28 ºC) in a sealable plastic container for further analyses.

2.1.1 Proximate analysis and chemical composition

Filter cake samples were characterized for its proximate composition (moisture, volatile matter, ash and fixed carbon) following NREL/TP-510-42622 (Sluiter et al, 2008) and ASTM D1762-84 (ASTM, 2011). To determine the water soluble fraction of the sample, water extractives of filter cake sample (~2 g) were first determined using Soxhlet extractor for 24 hours with 4 to 5 siphon cycles per hour, following NREL protocol (Sluiter et al., 2008) with slight modification. The receiving flask containing the extract was cooled to room temperature. Contents of the flask was transferred into a 250 mL Erlenmeyer flask. The filter cake sample inside the thimble was thoroughly rinsed with distilled water, with the filtrate collected and added to the flask containing the extractives. The total mass of the collected extracts was recorded. The thimbles containing the extracted samples was then dried and the change in weigh of samples was taken to be the water extractives.

Soluble sugar recovered in the extractives were also analyzed using DNS (Miller, 1959) and phenol-sulfuric method (Dubois et al., 1956) for reducing sugar and total sugar, respectively. Residual base as dissolved in water extractives were also determined by titration with 0.01 N HCl until the pH meter indicated a neutral pH of 7 reading. Direct determination of residual base in CFC samples (~1 g) were also carried out by direct titration. Water of 50 mL volume was poured into an Erlenmeyer flask containing the sample and was heated to 100°C using a heating plate and stirred continuously with a magnetic stirrer for an hour. The mixture was then cooled, and a pH probe was immersed into the cooled mixture and titrated with 0.01 N standardized hydrochloric acid (HCl) solution until a pH of 7. Residual base were then expressed in equivalents of NaOH.

Total recoverable sugar of the samples was determined by employing modified NREL protocol (TP-510-42618) for algal biomass by Kostas et.al (Kostas et al., 2016), with some modifications. About 90 mg (weighed to the nearest 0.1 mg) of the sample was first hydrolyzed using 3 mL of 11 M sulfuric acid at 30°C for an hour. The mixture was then diluted with 30 mL distilled water to achieve a concentration of ~ 1 M sulfuric acid and incubated for two hours at 95°C in a water bath (Memmert One 10). Neutralization of the reaction mixture was done by adding sodium carbonate until effervescence ceased. The neutralized reaction mixture was filtered using ordinary filter paper into a 100-mL volumetric flask and diluted to the mark with distilled water. Hydrolysates obtained after filtration was then analyzed for its total sugar and total reducing sugar (Section 2.2.1).

2.2 Direct dilute acid hydrolysis

Hydrolysis of cellulosic filter cake was carried out using varied sulfuric acid concentration (5, 7.5, 10, 12.5, and 15% v/v) over different reaction times (5, 30, 60, 120, 180, 240, and 300 min) at constant temperature of 95˚C and solvent-to-solid ratio of 8 mL/g. About 10 g of sample was hydrolyzed with 80 mL of sulfuric acid solution in a 100-mL screw-capped media bottles, maintaining a solvent-to-solid ratio of 8 mL/g. Reaction bottles were incubated in a water bath at 95˚C and intermittently mixed every 30-minute interval. After the pre-determined hydrolysis time, the media bottles containing the hydrolysis mixture were immediately quenched in an ice bath. Vacuum filtration with Whatmann No.2 filter paper (8 m pore size) of the hydrolysis mixture was then carried out immediately to separate the liquid hydrolysates from the solid residues. The residues remaining on the filter paper, referred to as post-hydrolysis residue (PHR), were washed with 20 mL distilled water, transferred to an evaporating dish (120 mm in diameter), and dried at 100˚C to constant weight. Residual solids yield, Y_{RS} (g solids/g dry sample), was calculated using Equation 1, where m_{rd} (g) was the overall weight of dried filter paper, residue and evaporating dish, m_{dd} (g) the mass of dry evaporating dish, m_{df} (g) the mass of dry filter paper and M the moisture content of the sample being analyzed. Dried PHR were then stored in a sealable plastic for further proximate analysis (section 2.1.1) and total acid density determination (section 2.2.2).

$$Y_{RS} = \frac{m_{rd} - m_{dd} - m_{df}}{m_s \times (1 - M)} \quad (1)$$

The hydrolysate separated from the solid was pooled together with the washing and the resulting volume was measured using graduated cylinder. Hydrolysates collected as such were then analyzed for pH, transferred into a sample bottle, and stored at 4˚C for further analysis. The hydrolysate yield (Y_H) was calculated based on the volume V_h (mL) of collected hydrolysate and washing using Equation 2, where V_w (mL) was the volume used for washing and (mL) the volume of hydrolyzing medium.

$$Y_H = \frac{V_h}{V_w + V_{hm}} \quad (2)$$

2.2.1 Hydrolysate analysis

Hydrolysates were then analyzed for its total reducing sugar and total sugar content by employing DNS method (Miller, 1959) and phenol-sulfuric method (Dubois et al., 1956), respectively. Analysis of inhibitor concentration in the form of 5-hydroxymethyl furfural (5-HMF) in the hydrolysates was also done using high performance liquid chromatography (HPLC) following the protocol described by Ahmed et. al (Ahmed et al., 2013).

Total Reducing Sugar Analysis. DNS solution was prepared by dissolving about 10 g of dinitrosalicylic reagent and 2 g of crystalline phenol in 800 mL of 2% sodium hydroxide (NaOH) solution. Sodium sulphite solution was prepared separately by dissolving 5 g of sodium sulphite in 200 mL of 1% NaOH solution. Total reducing sugar content of the standard glucose solutions and hydrolysates was analyzed by pipetting about 2.4 mL of DNS (dinitrosalicylic acid) and 0.6 mL sodium sulfite into the sample (3 mL). After incubation of the vial containing the sample at 95°C in a water bath for 5 minutes, the vial was then removed from the water bath and 1 mL of 40% Rochelle salt solution was added into the vial to stabilize the color of the reaction

mixture. The vial was quenched in a water bath (~30°C) for 10 minutes, mixed thoroughly for 10 seconds using a vortex mixer, and analyzed using spectrophotometer (UV-1700, Shimadzu, Japan) at 540 nm. From the calibration curve, total reducing sugar concentration, C_{TRS} (mg/mL) was first determined, fractional yield, Y_{TRS} (g total reducing sugar/g dry filter cake sample), and the recovery (%), R_{TS} (g total reducing sugar/g total sugar content) in hydrolysate were calculated using Equation 3 and Equation 4.

$$Y_{TRS\ or\ TS} = \frac{C_{TRS\ or\ TSC} \times V_h}{m_S \times (1 - M)} \quad (3)$$

$$R_{TRS\ or\ TS} = \frac{Y_{TRS\ or\ TS}}{TSC\ or\ TRS} \times 100\% \quad (4)$$

Total Sugar Analysis. Phenol solution was prepared by dissolving approximately 50 g of phenol crystals with water and was diluted to 1 L. Total sugar concentration of the standard solutions and hydrolysate was determined by adding about 1 mL of phenol solution and 5 mL 96% sulfuric acid into the vial containing the sample (1 mL). The reaction mixture was allowed to stand for 10 minutes and mixed thoroughly using vortex mixer for 10 seconds. After incubation of the sample at 30°C for 20 minutes in a water bath, its total sugar content was then analyzed using a spectrophotometer (UV-1700 UV Visible Spectrophotometer Shimadzu) at a wavelength of 490 nm. The concentration of total sugar in the hydrolysate, C_{TS} (mg/mL), fractional yield, Y_{TS} (g total sugar/g dry filter cake sample), and the recovery (%) of total sugar, R_{TS} (g total sugar/r g total sugar content) in the hydrolysate were calculated using Equation 3 and Equation 4.

Analysis of inhibitor (5-HMF). The degradation product of glucose, 5-HMF, formed during acid hydrolysis was determined using HPLC equipped with C-18 Reverse-phase Column (ODS-3V, Inertsil, Europe), LC 10AT high pressure pump, CTO-10A oven and UV-VIS detector, SPD-10AV (Shimadzu, Japan). A mixture of acetonitrile, water and acetic acid at 11:88:1 v/v/v proportion was used as the mobile phase. Hydrolysate samples were filtered using sterilized membrane filter (0.45 µm) and injected (20 µL injection volume) into the HPLC system at 30°C, with elution at constant flowrate of 1 mL/min, and absorbance measured at 276 nm wavelength. The inhibitor concentrations (g/L) were calculated from the calibration curve (peak area vs. mass of analyte) obtained from the response detection of the standard 5-HMF solutions.

2.2.2 Yields of proximate components and acid sites in PHR

To understand the mass gain or loss of the post-hydrolysis residue (PHR), yields of proximate components were calculated using the data obtained from proximate analysis of the PHR and residual solids yield. Post hydrolsate residue samples were also analyzed for its total acid density adopting the procedure employed by Boehm et.al (Boehm et al., 1964). About 0.2 g of the sample was added with 50 mL (V_B) standardized 0.05 M NaOH solution and mixed thoroughly for 24 h at 200 rpm in an incubator shaker. The solids were then separated from the solution by filtration using Whatman filter paper (8 m pore size) and the volume of the filtrate (V_s) was measured using a graduated cylinder. An aliquot of about 10 mL (V_a) of the filtered solution is pipetted and added with 25 mL (V_A) standardized 0.035 M HCl solution. The acidified solution was added with three drops of phenolphthalein indicator and titrated with the standard 0.05 M NaOH solution using a buret until the color of the solution changed from colorless to pink. Volume of the titrant dispensed (V_T) was calculated by taking the difference of the final and initial buret reading. Total acid density of the residual solids in mmol H^+/g dry residue (ρ_{AD}) was then calculated using Equation 5, where C_B (mmol/mL), C_A (mmol/mL), and C_T (mmol/mL) were the concentrations of NaOH, HCl, and the titrant (NaOH) used, respectively.

$$\rho_{\mathrm{AD}} = V_B \left\{ \frac{C_B - \frac{[C_A V_A - C_T V_T]}{V_a}}{m_{RS}} \right\} \quad (5)$$

3. RESULTS AND DISCUSSION

Carrageenan filter cake (CFC) samples as received had a moisture content of 89.22±1.00% w/w (wet basis). Characteristics of dried and milled CFC are summarized in Table 1. Dry filter cakes contain ~75% w/w ash and ~25%w/w organics based on the amount of volatile matter and fixed carbon present. Considering that majority of what is left as residue during the production of refined carrageenan is cellulose, the organic fraction was expected to be purely composed of sugars. Upon determination of the total recoverable sugar content, it was found to contain 26.78± 5.80% w/w, with 15% of that fraction determined to be soluble sugar. This is consistent with the estimated organic fraction based on the proximate analysis confirming that the organic fraction of CFC is primarily cellulose.

The amount of cellulose as determined is much lower than those reported in literature about residues from carrageenan extraction. In a study by Tan and Lee (2015) wherein they performed laboratory-scale carrageenan extraction,

TABLE 1: Characteristics of the collected carrageenan filter cakes (CFC).

Proximate Constituents	Composition (g /100g) [a]
Moisture content	3.83 ± 0.10
Volatile matter	22.18 ± 0.70
Fixed Carbon	3.44 ± 0.20
Ash	74.38 ± 0.30
Total recoverable sugar	
Total sugar	26.78 ± 5.80
Total reducing sugar	24.52 ± 0.25
Water extractives	18.85 ± 1.00
Soluble sugar	
Total sugar	4.39 ± 0.70
Total reducing sugar	1.07 ± 0.00
Residual base	0.17± 0.04[b]/ 0.09 ± 0.01[c]

[a] *relative to filter cake in dry basis*
[b] *1g of sample suspended in 50 mL hot water while being titrated*
[c] *titration of water extractives*

the resulting residue, referred to as macroalgae carrageenan residue (MCR), was found to contain ~68%w/w cellulose. However, in their work, perlite or filter aids were not added during the separation of the extracted carrageenan; this explains why MCR only had approximately ~32%w/w ash. In a separate study by Masarin et.al (2016), residues (23 to 28 g residue per 100 g seaweed) left after laboratory-scale carrageenan extraction were reported to contain 59 to 62%w/w sugars and only 10 to 15%w/w ash. Given that perlite is added as filter aid in industrial carrageenan extraction at a minimum of 500 kg per 1700 kg of seaweeds processed, the total ash in CFC may add up to at least 56 to 67%w/w, which explains the high ash content of the CFC obtained in this work. The large fraction of ash in CFC is mainly contributed by the presence of perlite added during the filtration process, which is not incorporated in laboratory-scale carrageenan extraction. Other sources of ash include the inherent minerals found in the algal biomass and the residual base.

Considering CFC was obtained from alkali carrageenan extraction, the amount of residual base expressed as NaOH equivalent were also determined as this could neutralize the acid and interfere in the hydrolysis process. Analysis of the water extractives revealed that only a small amount of residual base (~0.09%w/w NaOH equivalent) was present in the CFC. However, perlite, primarily made of alumina and silica, can possibly react with the acid. Thus, a second approach of determining the equivalent base (–OH) through direct titration of the solids suspension with hydrochloric acid was carried out. About 0.17%w/w of NaOH equivalent was determined using this method, which is twice than the amount of base dissolved in the water extract. Nevertheless, this amount of residual base does not significantly reduce the actual concentration of the acid during the determination of total recoverable sugar and is considered not to cause interference during the analysis.

3.1 Acid hydrolysis of carrageenan filter cake

The effect of acid concentration (5% v/v to 15% v/v) and hydrolysis time (5 min to 300 min) on the hydrolysis of CFC at a temperature of 95°C and solvent-to-solid ratio (SSR) of 8 mL/g were investigated in this study. The responses that were determined to evaluate the effect of the variables were the amount of total sugar and total reducing sugar in terms of yield and concentration. Generally, it can be observed that at all acid concentrations (5% v/v to 15% v/v) used, the total sugar yield is greater than the total reducing sugar at any time (Figure 1). This indicates that not

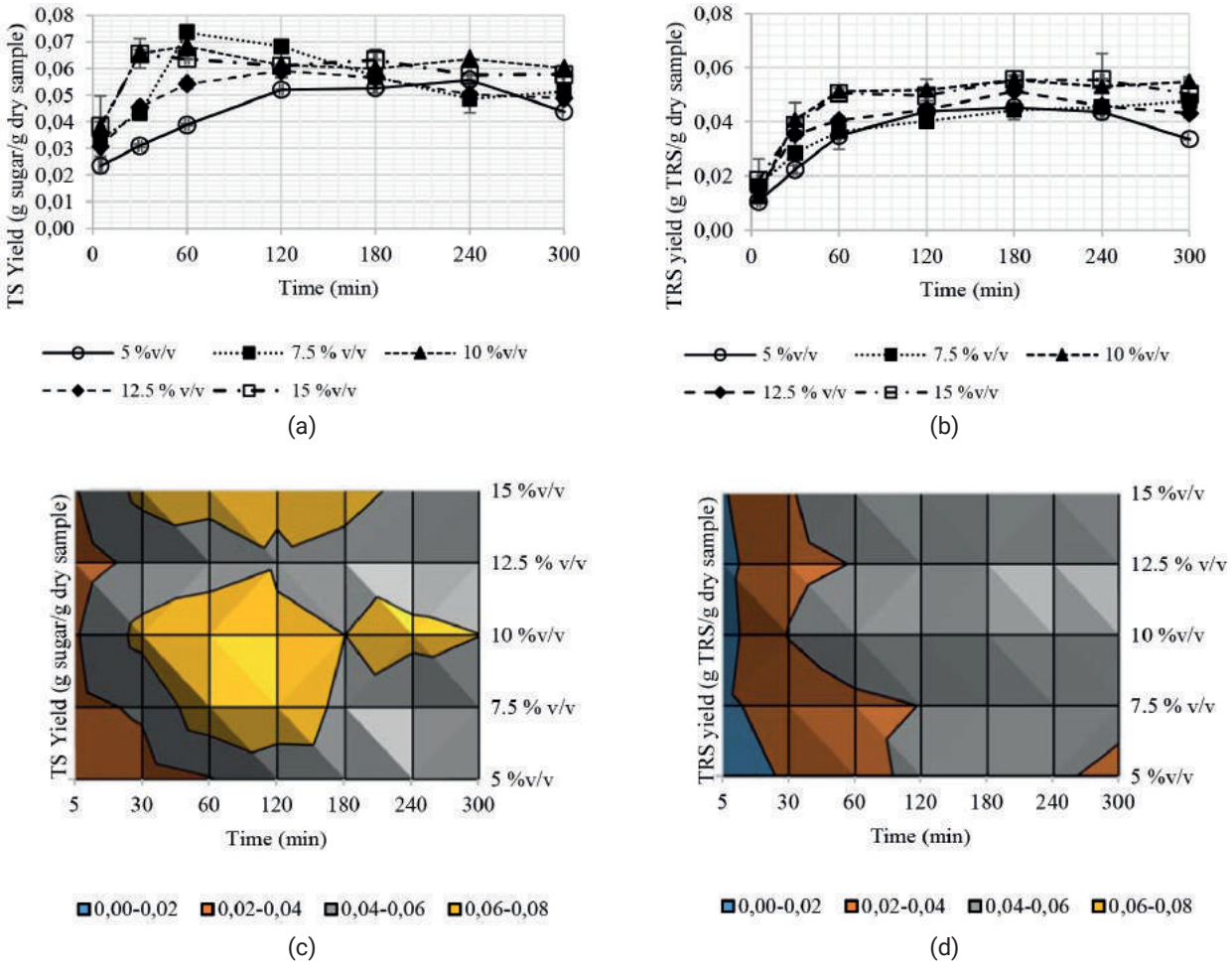

FIGURE 1: Sugar yields during hydrolysis at 95°C and SSR of 8 mL/g (a and c) total sugar (TS) yield versus time (b and d) total reducing sugar (TRS) versus time (This figure is to be printed in black and white in the printed version but colored in the online version).

all sugars released from the cellulosic structure are in its monomeric form.

It can be observed that across the different acid concentrations used, a maximum sugar yield of ~0.065 w/w was achieved, corresponding to a total sugar recovery of ~20% to 24%, respectively. Use of higher acid concentrations apparently results in faster rate of release of sugars from CFC as evident by a steeper curve during the earlier part (5 to 30 minutes) of the hydrolysis. For instance, the rate of release of structural sugars at 7.5 to 15% v/v H_2SO_4 ranges from 0.006 to 0.007 g/g.min, whereas a rate of 0.004 g/g.min at 5% v/v was obtained. At lower acid concentrations (5%v/v) used, the maximum sugar yield was achieved at 240 minutes while at higher acid concentrations (7.5%v/v to 15% v/v) used, maximum was achieved at 30 to 120 minutes (Figure 1a); using 10%v/v H_2SO_4 for 60 to 120 minutes appears as a good combination for acid concentration and hydrolysis time as this can achieve a sugar yield of ~ 0.06 w/w. However, this yield translates to relatively low sugar recovery of ~24%. The low recovery may be due to the crystallinity of cellulose, which made degradation into its monomeric constituents difficult. It is possible that the major fraction of the sugar hydrolyzed were the soluble sugars (3.5%w/w) originally present in the filter cake. The additional sugar recovered could be from the hydrolysis and breakdown of amorphous cellulose. Cellulose structure has amorphous and crystalline region. Crystalline cellulose is more stable due to its stable structure making it highly recalcitrant towards acid hydrolysis (O'Sullivan, 1997). In addition, no traces of 5-HMF were found during the analysis of inhibitor which suggests that the hydrolysis conditions employed may not be severe enough to further degrade the reducing sugars to form furans or perhaps the furans have already been further degraded to other compounds which were not detectable in the analytical conditions employed.

Several studies on the hydrolysis of algal biomass (Table 2) have been conducted. A study of Meinita et al. (2012) on the hydrolysis of K. alvarezii using 0.2 M (1% v/v) sulfuric acid at a temperature and SSR of 130°C and 5 mL/g, respectively, for 15 minutes achieved a sugar yield of ~20%. Hydrolysis of other algal biomass such as Gracilaria tenuistipitata, Gracilalriopsis chorda and Gelidium amansii, at the same hydrolysis conditions achieved a sugar yield of 27%, 23%, and 29%, respectively (Dyah et al., 2014; Meinita et al., 2013). Higher sugar yield was expected for the hydrolysis of raw seaweeds considering that its structure contains hemicellulose and no lignin unlike carrageenan filter cakes which are predominantly composed of cellulose (Daroch et al., 2013; Tan and Lee, 2014). Hemicellulose are easily hydrolyzed into its monomeric sugars in contrast to cellulose which have high degree of crystallinity (Kumar et al., 2009).

Enzymatic hydrolysis of CFC has been studied by Masarin et.al (2016) and Tan & Lee (2014) who reported a sugar yield of 64% to 68% w/w which corresponds to a sugar recovery of >99% (Table 2). This sugar recovery was achieved at mild conditions (50°C and 4.8 to 5.4 pH) but

TABLE 2: Hydrolysis of different algal biomass.

Substrate	Type of catalyst	Catalyst concentration	Solvent to solid ratio (ml/g)	Temp. (°C)	Reaction Time (h)	Sugar Yield (% w/w) / Sugar Recovery (%)	Total Sugar (g/L)	Furan (g/L)	Ref.
CFC	H_2SO_4	2.7 M	8	95	0.5	~6/25	7.3	Not Detected	This study
Kappa Alvarezii	H_2SO_4	0.2 M	10	130	0.4	8.9[a]/17[a]	~8.9	nd	(Maria Dyah Nur Meinita et al., 2010)
	H_2SO_4	0.2 M	5	130	0.4	20.2[a] /51.6[a]	40.4	5.9	(Meinita et al., 2012)
	H_2SO_4	0.2 M	10	130	0.4	-	38.5	3.6	(Dyah and Meinita, 2012)
Gracilaria tenuistipitata	H_2SO_4	0.2 M	10	130	0.4	26.6[a]/65.1[a]	26.6	1.2	(Meinita et al., 2013)
Gracilariopsis chorda	H_2SO_4	0.2 M	10	130	0.4	23.4[a]/68.0[a]	23.4	2.8	
Gelidium amansii	H_2SO_4	0.2 M	10	130	0.4	29.2[a]/48.7[a]	29.2	4.8	
Gelidium latofolium	H_2SO_4	0.2 M	20	130	0.4	22.2[a]/37.0[a]	11.1	3.5	(Dyah et al., 2014)
MCR	Enzyme[b]	0.09 % v/v	50	45	72	68.5[a]/100[a]	13.7	None	(Masarin et al., 2016)
MCR	Enzyme[c]	5.8 % v/v	50	50	54	63.9/99.8	12.8	None	(Tan and Lee, 2014)
MCR[d]	Solid catalyst (Dowex ™ Gr-D8)	4 % w/v	10	120	0.5	nd	nd	nd	(Tan and Lee, 2015)
	Enzyme [c]	0.2 %v/v	50	50	30	67.8/99.8	13.6	none	

Note: nd-no data; [a] *calculated value based on available data; 0.2 M is equal to ~1%v/v; 2.7 M is equal to ~15 %v/v;* [b] *92 FPU/ml (Cellic Ctec II) & 1800 UI/ml (β-glucosidase);* [c] *82.08 FPU/ml (Celluclast 1.5L) & 326.12 CBU/ml (Novozyme 188); dtwo-step hydrolysis*

a relatively longer reaction time (54 to 72 hours) was required. To hasten the enzymatic hydrolysis, pre-treatment of the biomass was done to reduce the crystallinity of cellulose. Tan & Lee (Tan and Lee, 2015) pre-treated MCR using a solid-acid catalyst (Dowex Gr-D8). This reduced the reaction time for enzymatic hydrolysis from 72 hours to 30 hours. Sugar concentration in hydrolysates achieved from enzymatic hydrolysis of MCR was ~13 g/L. In this study, a maximum sugar concentration of ~7.2 g/L was achieved at an acid concentration of 10% v/v (Figure 2). Considering that the achieved substrate concentrations range from 2 to 7 g/L, the sugar-rich hydrolysate recovered in this study may serve as potential substrate for fermentative processes, specifically in the pre-cultivation stages, such as the production of ethanol using Saccharomyces cerevisiae (ATCC 24860) (which requires 1.6 g total sugar/L substrate concentration, fermented at 3.7 pH, 30°C for 24 hours) (Ergun and Ferda Mutlu, 2000) and succinic acid production using Corynebacterium glutamicum (which requires 3.6 g glucose/L substrate concentration fermented at 30°C for 24 hours) (Okino et al., 2008).

Hydrolysate yield is also an important parameter to be looked into considering that the hydrolysate obtained in this study can be a potential substrate for subsequent fermentation process. The amount of hydrolysate available for fermentation is a useful information in later process design. The average hydrolysate yield from CFC was found to range from 65 to 78% of the total liquid used in the process. The lower hydrolysate yield (65%) observed when using 5%v/v H_2SO_4 (Figure 3a) was possibly due to the absorption of hydrolysate by CFC. Apart from the quantity of recoverable hydrolysate and sugar content, inhibitory substances in the hydrolysate should also be accounted. Considering that cellulose is made up of cross-linked glucose sugars, hydrolysates may contain degradation products of glucose such as 5-HMF (Kanchanalai et al., 2016). However, analysis of inhibitor using HPLC showed that no traces of 5-hydroxymethyl furfural in the hydrolysates, which suggests that the hydrolysis conditions may not be severe enough for the reducing sugars to be degraded to form 5-HMF. The presence of 5-HMF in the acid hydrolysates is undesirable in subsequent fermentation process as it in-

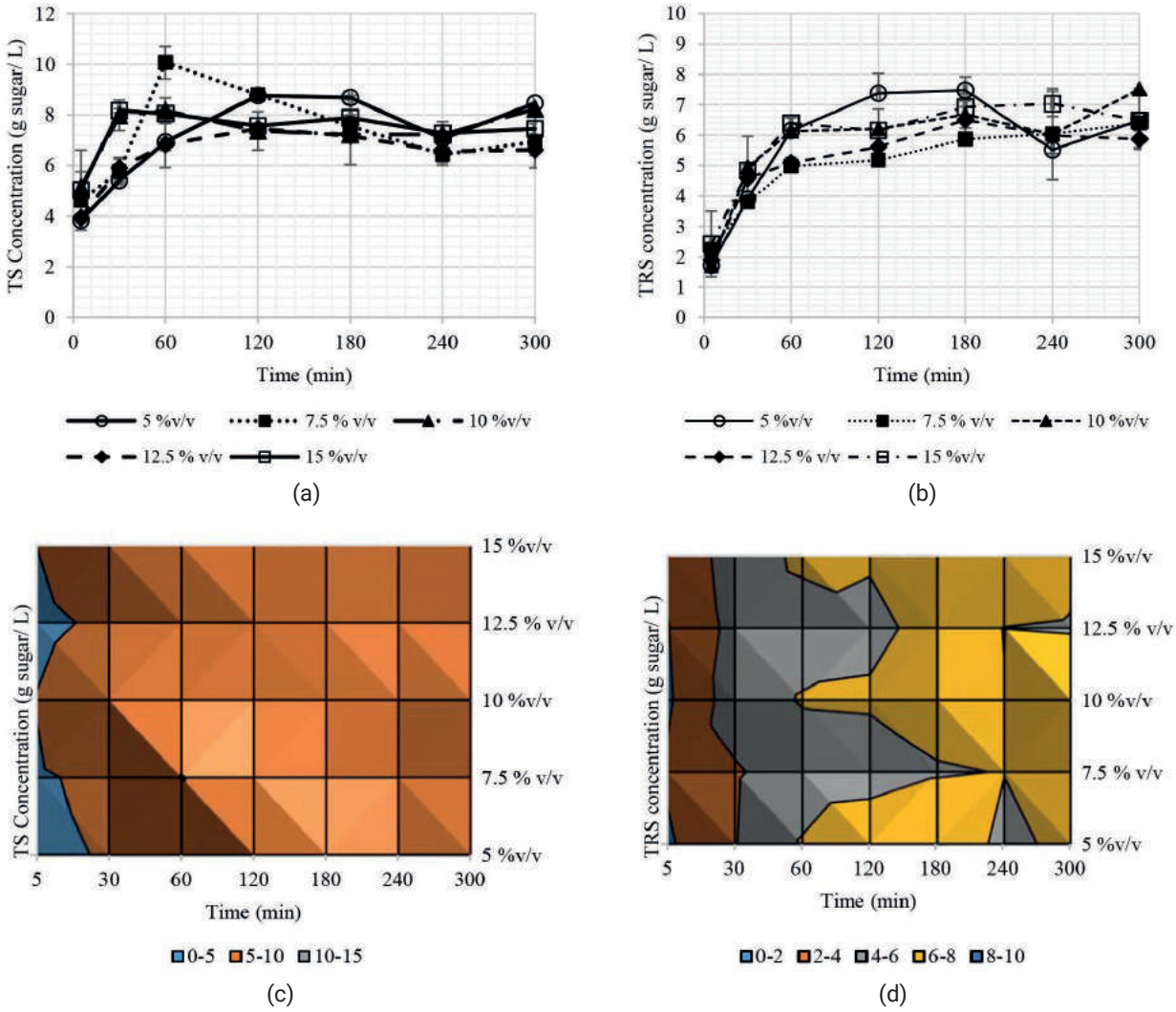

FIGURE 2: Sugar yields during hydrolysis at 95°C and SSR of 8 mL/g (a and c) total sugar (TS) yield versus time (b and d) total reducing sugar (TRS) versus time (This figure is to be printed in black and white in the printed version but colored in the online version).

hibits growth of cells, and prolongs the lag phase, thereby decreasing the productivity of microorganisms (Almeida et al., 2007). Inhibitory effects of 5-HMF to oleaginous yeast (Cryptococcus curvatus) is significant for concentrations over 3 g/L (Yu et al., 2011). Meanwhile, for Rhodosporidium toruloides yeast strain, complete inhibition was found even at 0.1 g/L 5-HMF. Another consideration is the presence of high sulfate ions present owing to the amount of sulfuric acid used. The presence of sulfate groups leads to osmotic stress to the fermentation bacteria. Nonetheless, its inhibitory effects on glucose consumption of S. cerevisiae is found significant beyond concentrations of 0.2 M concentration (Casey et al., 2013). which could be addressed during the neutralization of the hydrolysate prior to fermentation as addition of lime would result in the precipitation of the sulfate ions in the form of calcium sulfate.

The hydrolysates obtained when using acid concentrations of 5% v/v to 15% v/v have a pH ranging from 0.02 to 0.58 (Figure 3b). The pH of the hydrolysate is higher than the pH of stock solution which ranges from -0.34 to 0.22. The increase in pH can be attributed to the loss of hydronium ions due to its consumption during hydrolysis and possible absorption of the acid in the matrix of the biomass. Nevertheless, the acid concentrations remained low enough (pH < 1) and is not completely neutralized, which eliminates the possibility that hydrolysis did not proceed owing to neutralization of the available acids. In addition, the pH of the resulting hydrolysates decreases with prolonged hydrolysis time. This may be owing to the adsorption of the ions on the solid matrix and was not recovered during the washing step after filtration. As could be observed, the hydrolysate yields were also lower at shorter hydrolysis time, indicating part of the hydrolysates being trapped in the post hydrolysis residues and was later easy released and recovered at longer hydrolysis times.

3.2 Proximate constituents of PHR

Aside from hydrolysate yield, PHR solids had to be examined to validate the extent of hydrolysis. The residual solids left after hydrolysis of carrageenan filter cake using sulfuric acid (5% v/v to 15% v/v) at reaction time of 5 to 300 min, and at constant temperature and SSR of 95°C and 8 mL/g, respectively, had moisture contents of ~ 0.20 to 0.30 w/w. The PHR yields has a range of 84% to ~117%. Considering that carrageenan filter cake has a total sugar content of ~0.25 w/w, the PHR yield being above 75% could mean that there were still sugars left unhydrolyzed. At increasing acid concentration, at any time, PHR yield was observed to increase (Figure 4), which is contrary to what is expected because the mass of residual solids should decrease since part of the solid is broken down and solubilized during hydrolysis. This increase in PHR yield may be due to the residual sulfuric acid that remained with the PHR after the filtration step. When the PHR was dried at 105°C, the residual sulfuric acid did not volatize but instead may have sulfonated the residue and added up to the dry solid mass. At higher acid concentrations, more acids were available to sulfonate the solid, further increasing the residual solids yield.

From the proximate analysis of PHR, it was further found that volatile matter and ash yields increased, whereas fixed carbon yield decreased with the increase in acid concentration used (Figure 5). Volatile matter is comprised of hydrocarbons and some sulfur components that are removed upon heating at elevated temperature (950°C). The increase in volatile matter can be due to the attachment of sulfonic groups (-SO_3H) on the residues, thereby increasing volatile O, S and H elements. At higher acid concentration, more sulfonic groups are available for sulfonation, which also corresponds to the increase in the available volatile matter. The decrease of the fixed carbon

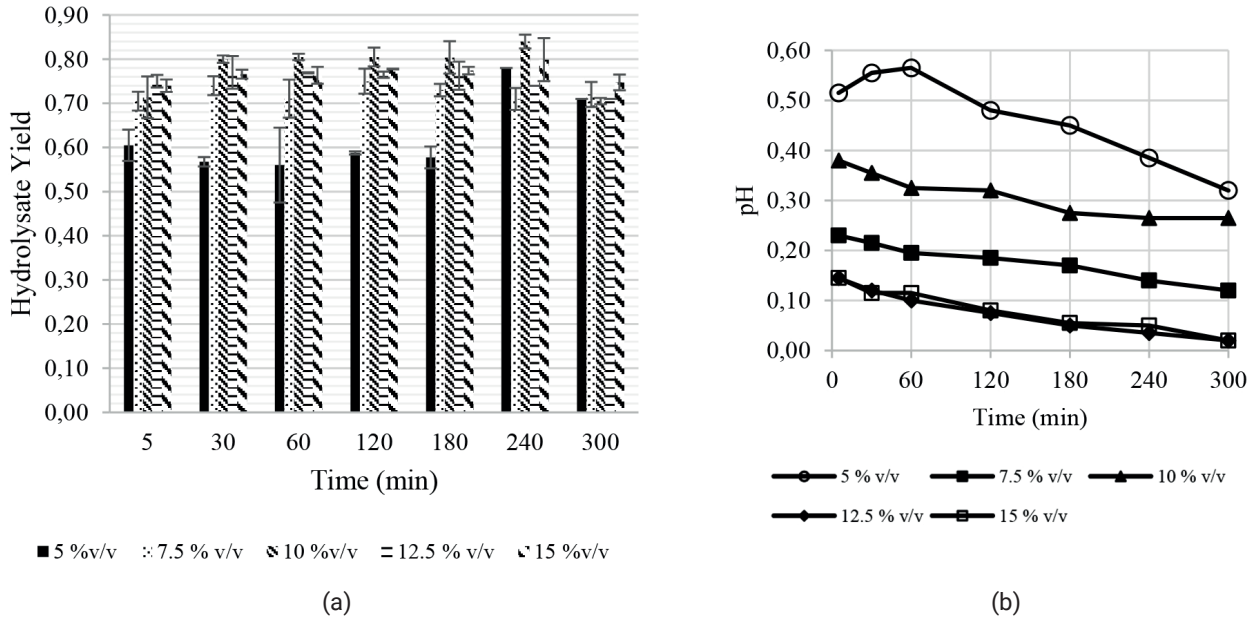

FIGURE 3: Hydrolysate yield (a) and pH (b)of hydrolysates at constant temperature and SSR of 95° C and 8 mL/g, respectively, and varying acid concentration (5 % v/v to 15 % v/v) and reaction time (0 to 300 min).

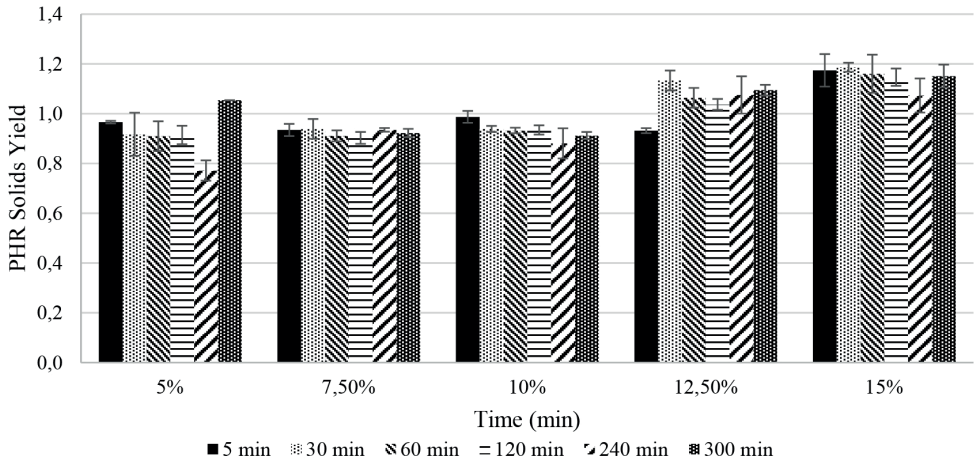

FIGURE 4: PHR solids yield obtained from the hydrolysis of carrageenan filter cake at 95 °C, 8 mL/g solvent-to-solid ratio, 5 to 300 min reaction time, and 5 to 15 % v/v acid concentration.

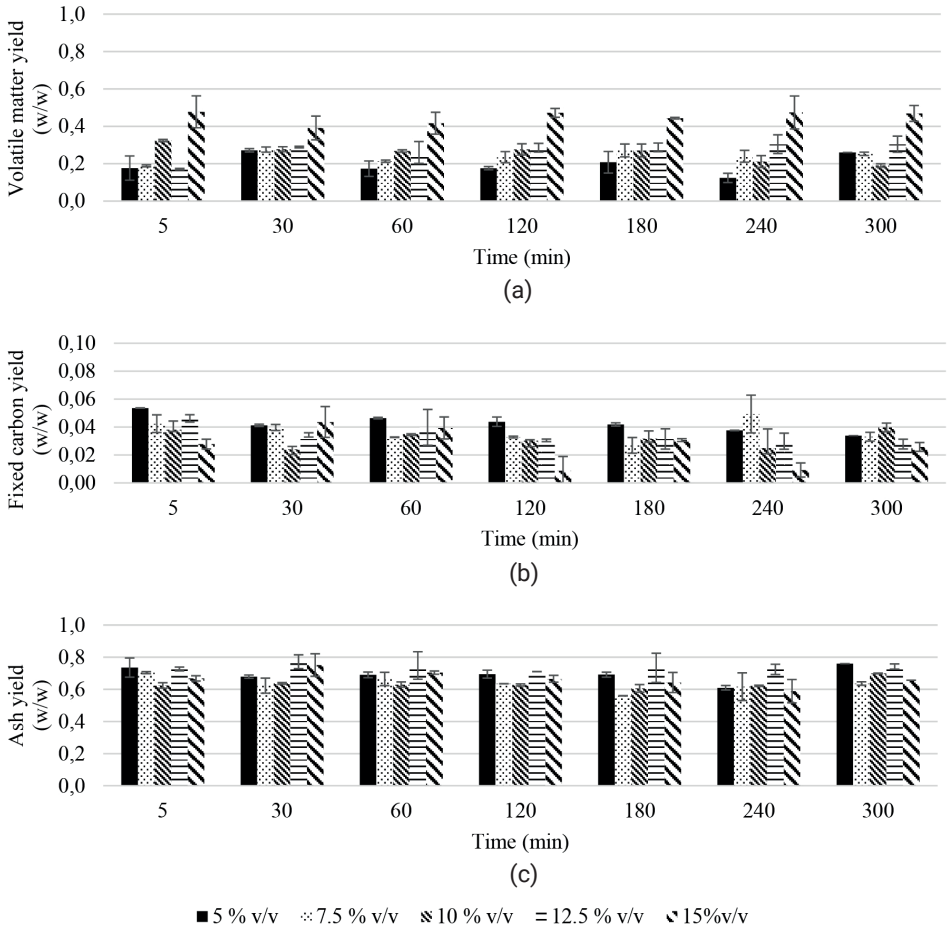

FIGURE 5: Proximate yields of the PHR (a) volatile matter (b) fixed carbon (c) ash at 95C, 8 mL/g solvent-to-solid ratio, 5 to 300 min reaction time, and 5 to 15 % v/v acid concentration.

yield is due to the fact that some of the solids were hydrolyzed during the hydrolysis process, and also possibly due to further degradation of the residue during drying at 105°C. The PHR ash yield, ranging from 0.6 to 0.8 w/w with an average of 0.67 w/w, is very similar to the initial ash content of the CFC, indicating the possibility of recovering perlite as the bound sulfonic groups can be easily removed during the calcination of perlite, which is carried out at elevated temperatures of 800 to 850°C (Samar and Saxena, Shweta, 2016).

In a study by Al-Dulaimi et.al (2015), hydrolysis of cellulose using concentrated acid (58%w/w) at constant solvent-to solid ratio of 12 mL/g and temperature of 45°C, resulted to an increase in molecular weight due to the en-

hancement of sulfonic group on the surface of the cellulose. Although the reactions in this study was carried out under lower acid concentration (5 to 15% v/v), it is still possible for sulfonation to have occurred considering that higher reaction temperature (95°C) was employed. Moreover, sulfonation by residual acid could have been further promoted during drying of the wet PHR at 105°C.

Sulfonation and carbonization of the PHR during drying is also supported by its appearance. As can be seen in Figure 6b, after filtration, the sample is still wet due to the retention of some acid hydrolysates. After drying at 105°C for 24 hours, PHR turned black (Figure 6c) probably due to the dehydration of the cellulosic structure and other organics in the presence of sulfuric acid. Sulfuric acid is a dehydrating agent that removes oxygen and hydrogen molecules in the cellulose structure in the form of water and leaves behind a carbon backbone as can be indicated by its black appearance (Woishnis and Ebnesajjad, 2012).

To further validate the possible sulfonation of the PHR, total acid density was determined. It was found that a total acid density ranging from 4 to 7 mmol H^+ for every gram of sample was obtained at an acid concentration of 5% v/v to 15% v/v during hydrolysis for 120 minutes at 95°C and solvent-to-solid ratio of 8 mL/g. As presented in Figure 7, there is an increasing trend of total acid density at increasing acid concentration (5% v/v to 15% v/v). This is understandable because at higher acid concentration, more sulfonic groups from sulfuric acid is available for sulfonation. During hydrolysis and drying of the residue, dehydration and sulfonation possibly occurred simultaneously. It is observed that the actual total acid density determined is within the range of the theoretical amount of acid (in moles based on the amount of hydrolysates left in the residue after filtration) in the solid residue after hydrolysis (Figure 7). Considering that there are two hydronium ions in one mole of sulfuric acid, one may have facilitated the dehydration of the residue leaving behind the black carbon back bone while the other hydronium ion remained as part of the sulfonic group (SO_3H) formed during the sulfonation process.

Several studies on the use of partially carbonized and sulfonated cellulose as a solid acid catalyst has been published in literature. The total acid density of the dried

FIGURE 6: Appearance of carrageenan filter cake (a) before hydrolysis (b) after filtration (c) after drying at 105°C for 24 hours.

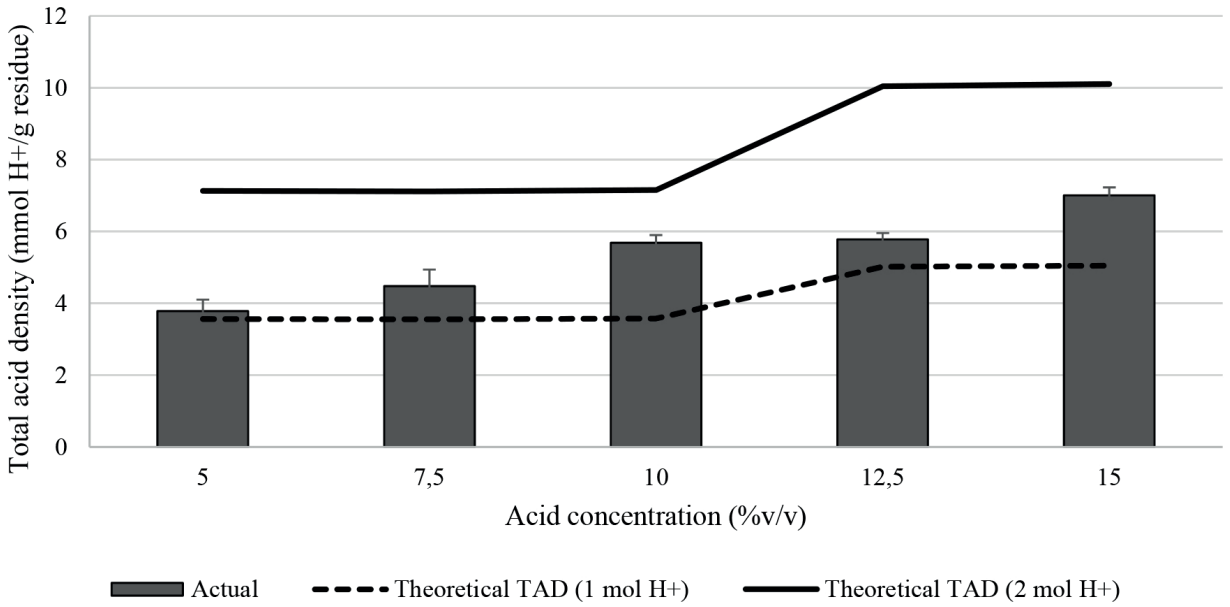

FIGURE 7: Total acid density (TAD) of PHR obtained at 5 to 15 % v/v acid concentration, 8 mL/g SSR, 95°C for 120 minutes (theoretical TAD was calculated based on the amount of hydrolysates left in the residue).

residues obtained after hydrolysis is comparable to those which have been successfully employed in the synthesis of biodiesel and hydrolysis of cellobiose. Carbon-based solid acid catalyst derived from sugars like glucose (Lokman et al., 2015), sucrose, and starch (Lokman et al., 2016) which had a total acid density of 4.2, 7.0, and 12.5 mmol H^+/g, respectively, and were successfully used in the synthesis of fatty acid methyl ester having high biodiesel yields of 89 to 94%. Suganuma et.al (Suganuma et al., 2010) also synthesized solid acid catalyst, but using microcrystalline cellulose as the raw material, which resulted in a catalyst with a TAD of 7.3 mmol H^+/g and was successfully used in the complete hydrolysis of cellobiose. Hence, the potential of the PHR from this study as a bio-based solid acid catalyst for biofuel (biodiesel and bioethanol) production may also be looked into in the future.

4. CONCLUSIONS AND RECOMMENDATION

Cellulosic filter cake obtained from carrageenan processing industry still contains sugar of as much as 25%w/w, with the remaining fraction being ash, which is primarily comprised of perlite. Hydrolysis of CFC can be carried out as means of recovering the sugars and at the same time prepare the residues after hydrolysis for subsequent sulfonation to produce a carbon-based solid acid catalyst. An increase in acid concentration during hydrolysis resulted to faster rate of recovery of sugars from cellulosic structure. A maximum sugar yield of ~0.05 to 0.07 w/w was obtained which corresponds to a sugar recovery of ~20 to 25%, respectively. Moreover, a maximum reducing sugar concentration of ~7 g/L was obtained at all acid concentrations with no traces of inhibitors (5-HMF) present in the hydrolysate, making it a potential substrate in subsequent fermentation processes. Drying of residues from the hydrolysis not only removed the remaining water but also resulted in the sulfonation of the PHR. The total acid density of PHR ranged from 4 to 7 mmol H^+/g and was found to increase as acid concentration during hydrolysis was increased from 5% v/v to 15% v/v, respectively. While hydrolysates can be used as substrate in subsequent fermentation process to produce high-value platform chemicals and biofuels, hydrolyzed residue can be further processed to produce heterogenous acid catalyst. Furthermore, perlite may potentially be recovered by simply subjecting the dried residue at high temperatures during calcination.

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POSSIBILITIES FOR THE USE OF SLUDGE FROM A DRINKING WATER TREATMENT PLANT AT GGABA III IN KAMPALA, UGANDA

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Article Info:
Received:
6 April 2019
Accepted:
30 May 2019
Available online:
28 June 2019

Keywords:
Bricks
Compressive strength
Heating value
Loss on ignition
Temperature

ABSTRACT

Sludge from the drinking water treatment plant at Ggaba III, located in Kampala (Uganda), was tested to evaluate the feasibility of two valorization routes, for building material and Solid Recovered Fuel (SRF) production. The aim of the research was to divert the huge amount of sludge produced every year, approximately equal to 2,140 metric tons of TSS/year, from landfilling. The average high heating value of the sludge was 8.44 MJ/kg TS, corresponding to the lower value of the interval of variation typically reported for other biosolids (8.0-23 MJ/kg). Different bricks were prepared at sludge to clay ratios of 0, 0.05, 0.1, 0.3 and 0.5 by weight. For each mixture composition, bricks of nominal size 215 x 102.5x 65mm were prepared by hand and fired for 6hrs in a Hoffman kiln at temperatures: 850°C, 900°C, 950°C, 1000°C and 1050°C. The bricks produced with a sludge to clay ratio of 0.1 fired at temperatures of ≥980°C met the compressive strength of 3N/mm^2 for common bricks according to Ugandan Standard (US) 102:1995. These results suggest that water treatment sludge at Ggaba is more suitable for the production of common bricks than using it as an energy source. Given the encouraging results that make the studied valorization route applicable in an emerging economy country as Uganda, further investigations are required to assess the leaching behaviour and stability of the mechanical properties over time.

1. INTRODUCTION

The Ggaba III drinking water treatment plant is managed by the National Water and Sewerage Corporation (NWSC), a public utility company 100% owned by the Government of Uganda (NWSC, 2014). The plant provides part of the drinking water distributed to Kampala, the Capital city of Uganda. The water, collected from the Inner Murchison Bay of Lake Victoria, is treated by means of the following sequence of processes: screening, pre-chlorination, coagulation, flocculation, clarification, filtration, pH adjustment and post chlorination. During coagulation, aluminum sulphate (alum, $Al_2(SO4)_3.14H_2O$), is used and alum sludge is produced. In order to meet the increasing demand of high quality drinking water, the sludge production is expected to increase, in accordance with the rate of increase in water production at this plant.

As noted by Babatunde and Zhao (2006), the costs of handling the enormous quantities of water treatment sludge (WTS) can account for a significant part of the overall operating costs of drinking water production. The Authors also noted problems of limited land available for WTS disposal in sanitary landfill sites. As one of the ways of managing sludges, Diener et al. (2014) suggested to identify valorization routes, to minimize the overall amounts to be landifilled and to use the recovered cost in funding sludge management interventions.

In the literature, the following WTS valorization options have been proposed and studied: application in building and construction materials (Mohammed et al., 2008; Badr et al., 2011; Mageed et al., 2011), utilization to improve particulate pollutant removal from sewage (Lai and Liu, 2004; Guan et al., 2005), application for phosphorus adsorption (Huang and Chiswell, 2000; Zumpe et al., 2002; Yang et al., 2006; Babatunde and Zhao, 2007); soil remediation (Elliot and Dempsey, 1991; Roy and Coulliard, 1998); application in pavement construction and geotechnical works (Raghu et al., 1987; Carvalho and Antas, 2005). According to Hegazy et al. (2011), the use of sludge in construction industry is considered to be the most economic and environmentally sound option.

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https://doi.org/10.31025/2611-4135/2019.13824

WTS has also been used to make bricks with binders such as clay (Weng et al., 2003; Ramadan et al., 2008). The most relevant factors affecting the quality of bricks were found to be sludge dosage into the mixture and the firing temperature (Weng et al., 2003). In other studies, incinerated WTS was used in combination with rice husks (Chiang et al., 2009) and with shale (Mageed et al., 2010). Hegazy et al. (2011) used WTS in combination with silica fume as a complete replacement of clay in brick making. In the study by Hegazy et al. (2011), which was performed at the temperatures commonly practiced in the brick kiln (900-1200ºC), the optimum sludge addition to produce brick from a mixture of sludge and silica fume was 50%. The produced bricks properties were found to be superior to the 100% clay control-brick and to those available in the Egyptian market (Hegazy et al., 2011).

It is important to identify valorization routes that are applicable under the conditions (availability of technologies, skills, funds for operation and maintenance) of an emerging economy country. Moreover, it is imperative that the proposed management of WTS is aimed at "closing the cycle" by reducing landfilling and promoting a local reuse of the material, otherwise considered to be waste. This research identifies valorization routes for the use of WTS in production of common bricks and as a raw material to provide energy for heating in industrial production of bricks.

2. METHODS AND MATERIALS

The liquid and dried WTS samples used in this study were obtained from Ggaba III water treatment plant while the clay samples were obtained from Lweza clays Ltd at Kajjansi.

2.1 Quantification of the WT sludge at Ggaba III

Liquid WTS at Ggaba III is discharged to the drying beds or wetlands through the nine groups of pipes from each of the four clarification tanks as illustrated in Figure 1. The sludge is discharged to the beds along with the cleaning water, from the cleaning operation of the clarifiers, which is done once a week according to the plant operators. In order to assess the sludge production rate, the time for the discharge of each clarifier was measured.

Up to 100 mL of liquid WTS samples was collected every 10 minutes for 60 minutes for the analysis of Total Suspended Solids (TSS) in triplicates using the procedures in Sandec Protocol Section D, SOP 014. Based on the discharge rate and TSS, the sludge production rate was computed as follows:

$$\text{Sludge production rate} = \text{Discharge}\left(\frac{m^3}{day}\right) * \frac{Average\ TSS\ (\frac{mg}{l})}{1{,}000{,}000} \quad (1)$$

2.2 Sampling

Composite samples were collected from sludge drying beds, illustrated in Figure 2. The top surface of a bed was cleared and a trowel inserted 2cm deep from the surface to obtain grab samples. The grab samples obtained were mixed in a plastic container to get composite samples, placed into a cool box and stored under controlled conditions for the subsequent characterization and experimental campaign.

2.3 Sample preparation

Clay samples, existing in the form of boulders were spread on a tarpaulin laid on a hard flat surface to dry in the sun for two days in order to attain moisture content of about 20%. The samples were then crushed using a tamping rod and sieved through a 425µm sieve in order to remove the coarse fraction. The WTS sample was air dried for 24 hours. The dried samples were pulverized using a ball mill crusher and sieved through a 425µm sieve.

2.4 Laboratory analysis on sludge and clay

Total Solids (TS), Total Volatile Solids (TVS) and ash content measurements were performed in triplicates according to the procedures in Sandec, Protocol Section D, SOP 013.

The higher heating value of air dried dewatered WTS samples was determined using a Bomb Calorimeter according to the procedure in Parr Manual (1948). The results were compared with those attained by using a Bomb Calorimeter IKA C1® according to DIN 51900, ISO 1928 standards without acid correction, using for calibration the pelletized benzoic acid as standard substance to determine the C-value of the calorimeter.

Dewatered WTS and clay samples were also tested for their moisture content, in triplicates according to British Standard (BS) 1377: Part 2: 1990 clause 3, elemental composition, by means of Xray fluorescence (Spectro Xepos), bulk density and particle size distribution (BS 1377: Part 2: 1990 Clause 9).

In order to determine the Plastic Limit of the different sludge and clay mixtures, the Atterberg method was applied in accordance to BS 1377: Part 2: 1990. Along with the plastic limit, liquid limit and linear shrinkage were measured.

2.5 Bricks production and characterization

Different mixtures containing sludge and clays were prepared in batch by varying the sludge dosage according to the following sequence: 0%, 5%, 10%, 15%, 20%, and 30% by weight. Thereafter, the mixture was spread on a tarpaulin laid on a hard flat surface, and mixed using spades until a uniform color was achieved. Water was added while mixing until the optimum moisture content of 13.2% was achieved and samples kept for at least three days to reduce air voids.

The commonly used hand moulding method (Nyakairu et al., 2002) for the preparation of local bricks was used. The raw material was placed into a water-lubricated wooden mould and then compacted by hands. The mixture in excess was scrapped off using a flat piece of wood to level the surface. Bricks, prepared from each of the above-described mixtures, were produced in accordance to BS 3921:1985 with a nominal size of 215 x 102.5x 65mm. A total of 200 bricks were produced and air dried for seven days prior to firing.

Firing of the bricks was done in two batches. The first

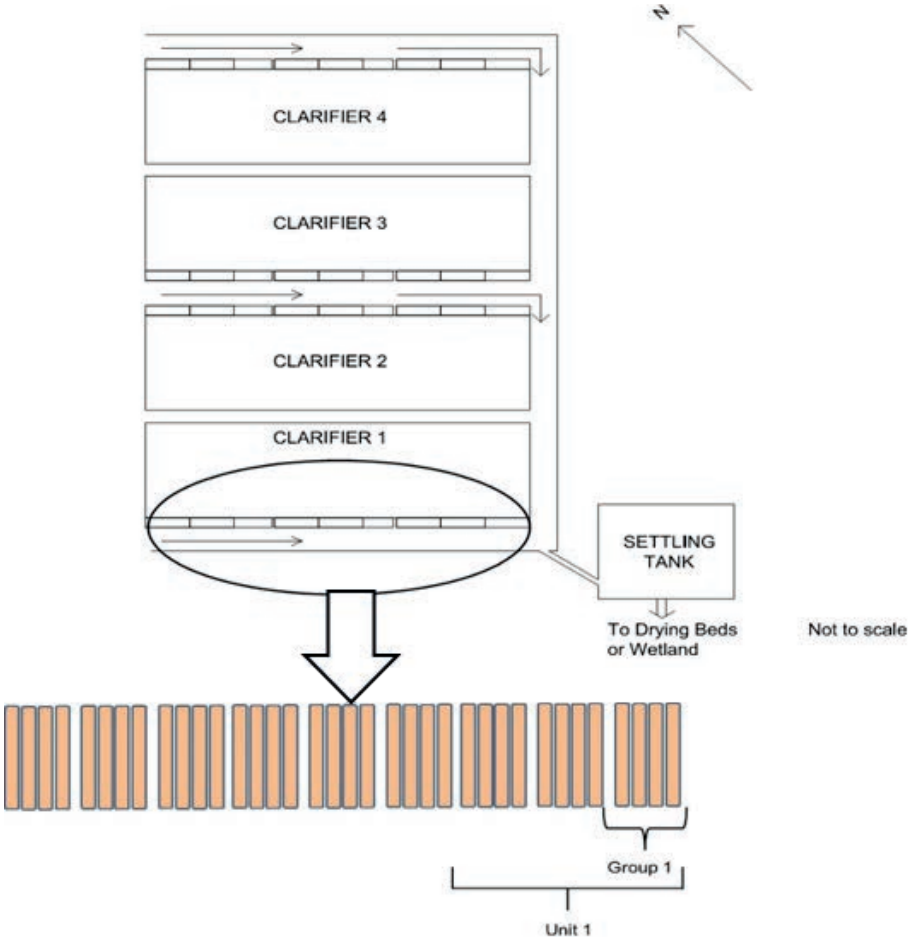

FIGURE 1: Illustration of discharge valves from one clarifier.

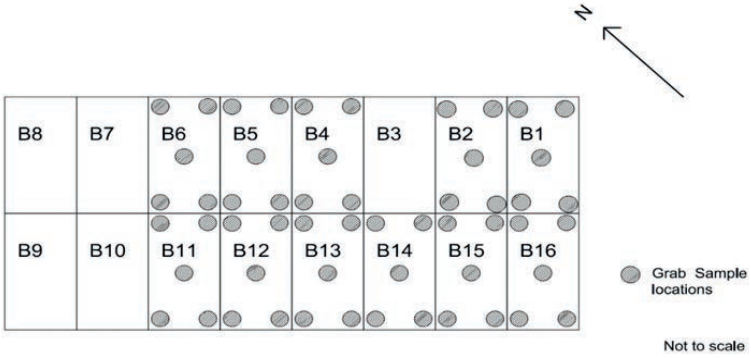

FIGURE 2: Schematic of sludge sampling points in the drying beds.

batch was fired in an electric furnace produced by Kilns and Furnaces Limited (Type: FL500, SN: FL 94 4417) at the Margret Trowell School of Industrial and Fine Arts Makerere University at temperatures equal to: 850, 900, 950, 1000 and 1050˚C for 6hrs. The second batch was fired in a furnace at Lweza Clays Ltd for 5 days. This was done in order to simulate the local conditions.

Tests were carried out on the fired bricks, as required by the Ugandan Standards (US), to establish their quality, including compressive strength measurement (BS 3921: 1985, BS EN 772-1:2011 and US 102: 1995), water absorption (BS 3921: 1985, BS EN 772-1:2011 and US 102: 1995), shrinkage, bulk density (BS 1881) and weight loss on ignition (BS 1377: Part 3: 1990).

3. RESULTS AND DISCUSSION

3.1 Sludge production rate at Ggaba III

For the four clarifiers, each discharge cycle was estimated to be equal to 9.9 minutes. The settled sludge discharged from the clarifiers at a rate of 0.56 m^3/min with an average TSS content of 737.817 mg/L, corresponded to

approximately 2,140 metric tons TSS/year. This production rate cannot be handled by the capacity of the drying beds currently available at the plant, and as such some of it is being disposed in the wetland.

3.2 Physical properties of sludge and clay

Table 1 shows the average total solid, moisture and ash content of sludge and clay, along with the bulk density. The moisture content of dewatered sludge and clay was equal to 13.2%±1.2% and 16.7%±3.2%, respectively. The ash content of sludge was obtained as 79.7%. The bulk densities of WT sludge and clay were 1.2920 g/cm^3 and 1.9840 g/cm^3 respectively at the end.

3.3 Elemental composition

The elemental content of the sludge and clay are shown in Figure 3. From Figure 3, both the WTS and clay have similar composition in terms of macro-elements, including Al, Si, Fe, S, and P. WTS has higher aluminum and phosphorous content than clay, with an Al content of 119500 mg/kg for sludge and 95560 mg/kg for clay, and a phosphorous content of 5093 mg/kg for sludge and 509.3 mg/kg for clay. Conversely, clay showed much higher silicon content of 40460 mg/kg than WTS with a silicon content of 268400 mg/kg. Toxic metals, including Arsenic, Mercury, Cadmium

TABLE 1: Physical properties and element content of sludge and clay.

Property	Unit	WT sludge	Clay
Total solids	wt%	74.5	83.3
Moisture content	wt%	13.2±1.2	16.7±3.2
Average ash content	wt%	79.7	-
Bulk density	g/cm^3	1.2920	1.9840

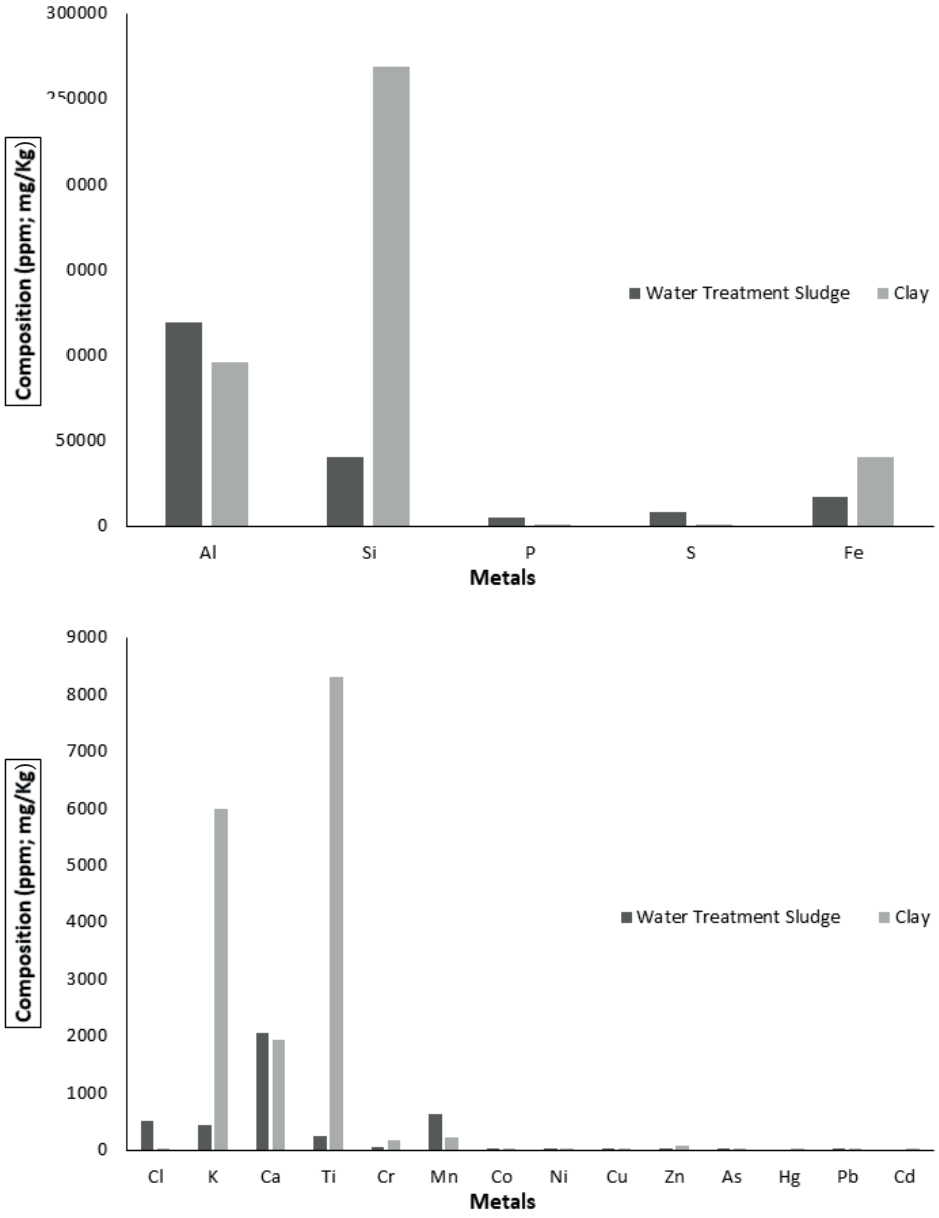

FIGURE 3: Elemental composition of sludge and clay.

and Lead were present in trace amounts.

3.4 Higher Heating Value

From the test carried out at Makerere University, the average higher heating value of the WTS obtained was 8.04 MJ/kg TS at an average moisture content of 13.2%. A higher heating value of 8.8415 MJ/kg TS was obtained from the test carried out at Paul Scherrer Institute (PSI) in Zurich Switzerland. The average higher heating value of 8.44 MJ/kg TS obtained in this study was compared with that of selected agricultural residues that are used as a source of renewal biomass energy as shown in the Figure 4.

The higher heating value of the WTS obtained in this study is lower compared to that of agricultural wastes usually used as biomass fuels. On the other hand, this value is comparable to the 8.0–23 MJ/kg observed from other biosolids (Spinosa and Vesilind, 2001; Mödinger and Mayr, 2006; Skjeggerud et al., 2009).

3.5 Particle size distributions

Figure 5 shows the particle size distributions of the WTS and clay respectively. Accordingly, the WTS and clay can be classified as fine sand, since maximum passing sieve No. 200 was 20%, which was less than 35% for silty soils, and clayey soil respectively basing on their plasticity and percentages of particles passing No. 200 sieve according to AASHTO classifications.

3.6 Atterberg limits

Atterberg Limits (BS 1377: Part 2, Clause 5:1990) testing was conducted to determine the Liquid and Plastic Limit (LL and PL, respectively) for each mixture. Atterberg limits are widely used in engineering to determine the plastic properties of clay materials (White, 1949). Plasticity enables the soil to undergo unrecoverable deformation without cracking or crumbling. It results from the presence of a significant content of clay mineral particles (or organic material) in the soil. The void space between such particles is generally very small in size with the result that water is held at negative pressure by capillary tension. This produces a

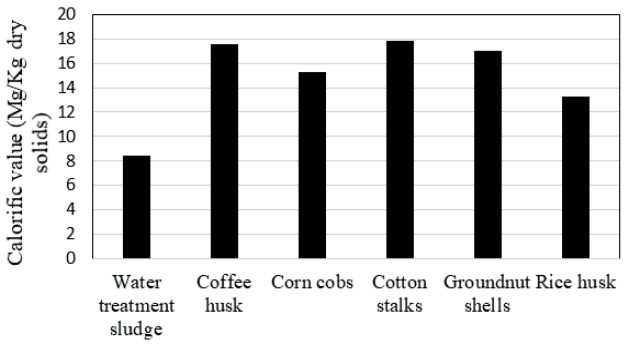

FIGURE 4: Calorific value of sludge and common wastes (Kumar et al, 2001).

degree of cohesion between the particles, allowing the soil to be deformed or moulded. The upper and lower limits of the range of water content over which the soil exhibits plastic behaviour are defined as the liquid limit and the plastic limit, respectively (Craig, 2004).

The Liquid limit increased with increasing sludge addition to the mixture, as illustrated in Table 2. The plasticity index decreased with increase in sludge dosage implying a decrease in the water content range within which the mixtures exhibit plastic behavior. The Atterberg's tests showed that the plasticity index (PI) is inversely proportional to the sludge content of the brick.

3.7 Compressive Strength

The compressive strength test is the most important test for assuring the engineering quality of a building material (Weng et al, 2003). From the results shown in Figure 6, the compressive strength of the produced bricks is greatly dependent on the amount of sludge in the mixture and the firing temperature. The general decrease in strength with sludge addition was also noted with bricks fired at Lweza Clays Ltd. An increase in strength with firing temperature was observed for each mixture tested.

Bricks having sludge addition of 0% and 5% and for all the firing temperature had compressive strengths greater than 3 N/mm^2, the minimum compressive strength specified by

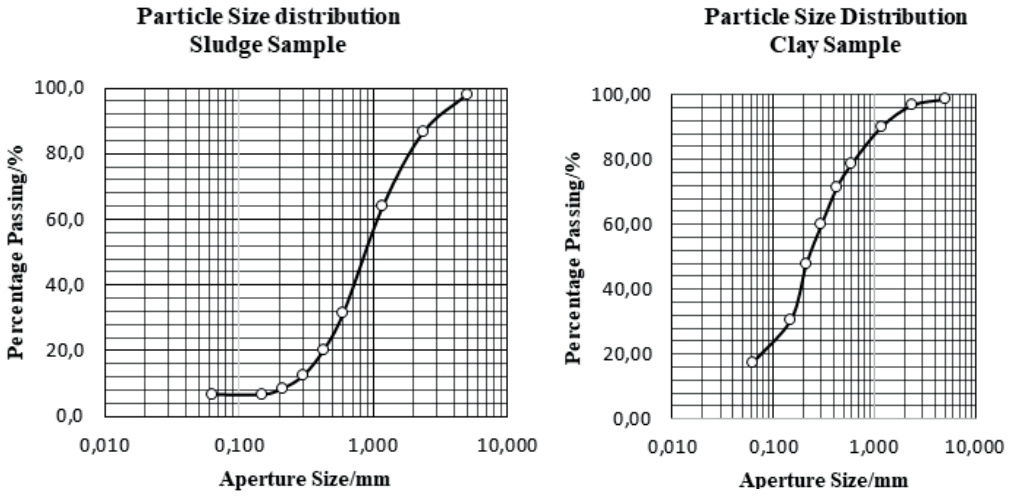

FIGURE 5: Particle size distribution of sludge and clay.

TABLE 2: Liquid Limit, plastic Limit and Plasticity Index of different clay and sludge mixtures.

Specimen	0%	5%	10%	20%	30%	50%
Sludge (% weight)	0	5	10	20	30	50
Clay (% weight)	100	95	90	80	70	50
LL, %	42.0	42.4	42.8	43.5	44.4	45.3
PL, %	20	23	25	27.7	31	34.5
PI	22	19.4	17.8	15.8	13.4	10.8

the Ugandan Standard specification for burnt clay bricks, US 102: 1995. Bricks with 10% sludge addition fired for all the temperatures above 980^{0}C met the compressive strength requirement of 3 N/mm^2 and were classified as common bricks.

3.8 Water Absorption

Water absorption is one of the key parameters that affects the durability of brick. The less water infiltrates into brick, the more durability of the brick. Thus, the internal structure of the brick must be compact enough to avoid the intrusion of water. The water absorption results obtained for the bricks with different sludge content were plotted against the firing temperatures in the figure below. From Figure 7, the water absorption of the bricks increased with increasing sludge dosages and decreased

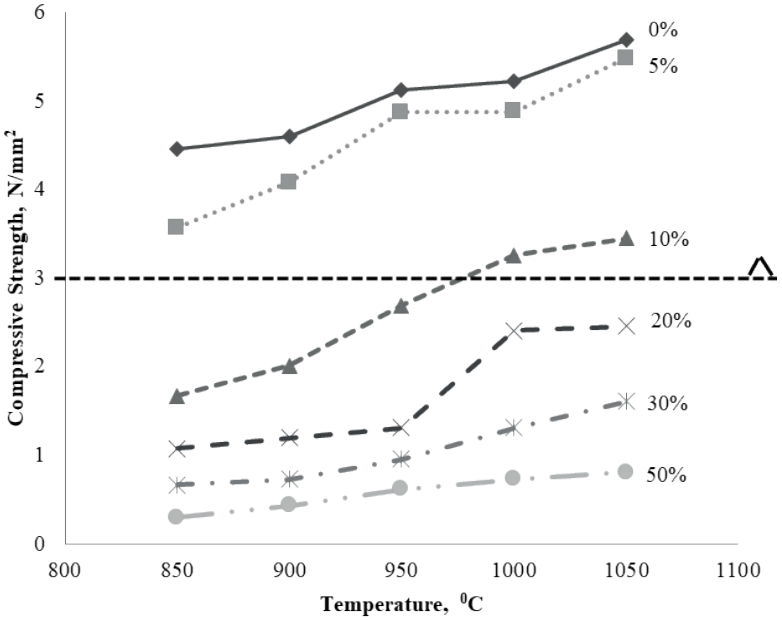

FIGURE 6: The compressive strength of bricks.

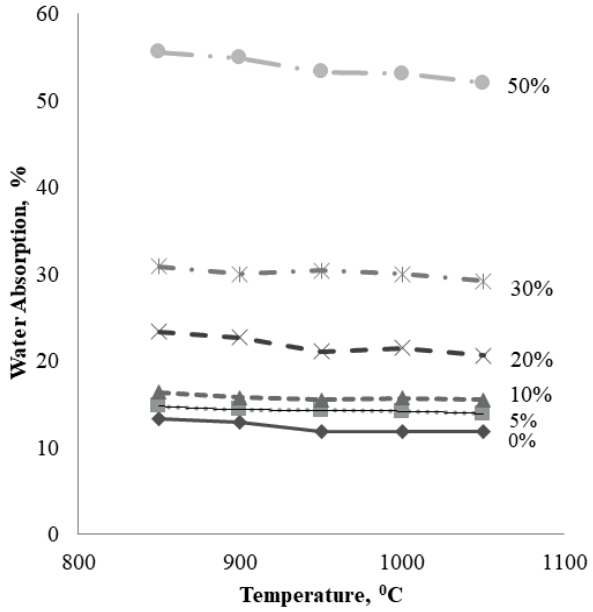

FIGURE 7: Water absorption of bricks.

with increasing firing temperature. The latter behavior may be explained hypothesizing that at the highest firing temperatures the degree of crystallization is enhanced, thus causing a reduction of pores size in the brick. Conversely, the increase of sludge content may be able to reduce the adhesiveness of the mixture, so that the internal pore size of the brick increases. All the bricks had higher water absorption in comparison to engineering bricks (less than 6.3%), industrial bricks (6.3%) and facing bricks (7%) according to the Uganda Standard (US) 102: 1995 standard. The standard did not specify limits for common bricks. According to Chiang et al. (2009), lightweight bricks with relatively high water absorption have been applied widely in the inner walls of green buildings, hence the water absorption of lightweight bricks is an insignificant factor in considering their application.

3.9 Shrinkage of bricks

There was no direct relation between the percentage of sludge content and shrinkage of the brick. A similar relationship was reported by Jordán et al. (2005) while studying the effect of substitution of clay for sewage sludge in different proportions on the technological properties of a ceramic material.

3.10 Bulk density of fired bricks

The bulk density as illustrated in Figure 8 decreases with increasing sludge dosages, and increases with increasing firing temperature. The bulk density of the bricks made of clay was between 1.8 and 1.814g/cm^3, in the range of 1.8-2.0 g/cm^2 for bricks made of clay (Weng et al., 2003) at temperatures between 960^{0}C to 1000^{0}C. At higher sludge additions, the bricks were lighter.

The decrease in bulk density of the fired bricks with increasing sludge dosage may be related to volatile organic content of the sludge (Babatunde and Zhao, 2007). As the combustible fraction is lost upon firing, pores are created within the bricks (Chiang et al., 2009) hence providing a lighter weight and good thermal insulation properties for green building applications. Weng et al. (2003) attributed the decrease in bulk density to the increasing amounts of water required for mixing at higher sludge additions, which leads to larger pore sizes on firing.

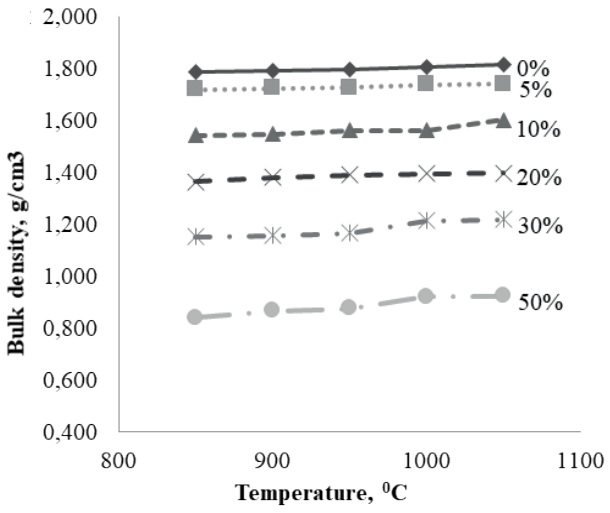

FIGURE 8: Bulk density of fired bricks.

3.11 Weight loss on ignition of bricks

The weight loss on ignition of bricks during the firing process is mostly due to the organic compounds and the inorganic Calcium carbonate, $CaCO_3$. Figure 9 shows the results of weight loss on ignition of bricks. It shows that increasing the sludge proportion and temperature resulted in increase in brick weight loss on ignition. The Ugandan standard did not give any comparison of the weight loss on ignition. When compared to available standards as Chinese National Standards (CNS, 1999), the weight loss criterion for a normal clay brick is 15%.

Bricks 0% and 5% sludge addition fired at all the firing temperatures, met the criteria according to CNS, 1999. However, upon the addition of sludge in the mixture, the loss of weight apparently increased as a result of organic matter loss from sludge. Furthermore, the brick weight loss on ignition also depends on the inorganic substances in both laterite and sludge being burnt-off during the firing process. The relationship between the compressive strength and weight loss on ignition at various temperature series is presented in Figure 10. From this figure, the compressive strength of the produced bricks of the sludge/clay mixture decreases with increase in weight loss on ignition. The relationship between compressive strength and bulk density at various temperatures is shown in Figure 11. According to Figure 11, the compressive strength is directly proportional to the bulk densities of the bricks produced from the sludge/clay mixture but is inversely proportional to the percentage of sludge additions.

4. CONCLUSIONS

The aim of this study was to determine the potential of water treatment sludge from Ggaba III as a building material and as a raw material for energy. The following conclusions are supported by the results of this study:

- The rate of production of the WTS was 2,140 tons TSS/year and this rate of sludge production cannot be managed by using the available sludge drying beds. Recycling this WTS by using it as substitute for clay in the production of common bricks is a possibility to reduce treatment and disposal costs. The rate of production of the WTS will also ensure that the business of using it with clay to produce bricks is reliably available in quantity.
- From the comparison of the elemental compositions of the WTS with clay, major elements appeared to be similar and hence the two materials were compatible when mixed together. The content of critical elements (Arsenic, Mercury, Cadmium and Lead) was low and hence the use of sludge is expected to pose no danger to the environment.
- The higher heating value of the WTS was obtained as 8.44 MJ/Kg TSS. This value is low although comparable to the 8.0-23 MJ/kg observed with other biosolids

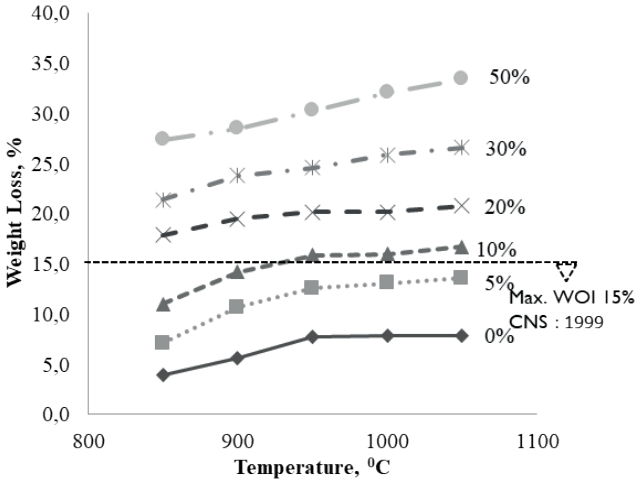

FIGURE 9: Weight loss on ignition of bricks.

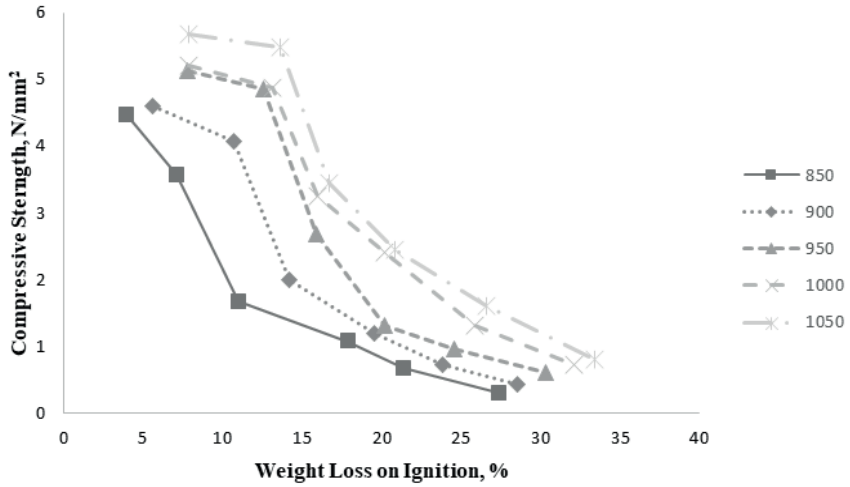

FIGURE 10: A graph relating the compressive strength to weight loss on ignition at various temperature series. The data points of the graphs are arranged with increasing sludge additions.

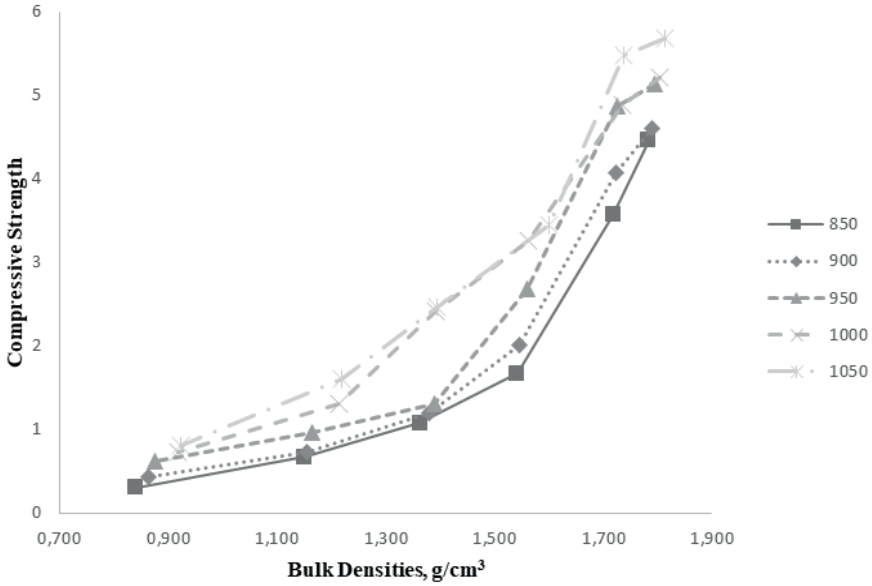

FIGURE 11: A graph relating the compressive strength to bulk densities at various temperatures. The data points of the graphs are arranged with decreasing sludge additions.

obtained from previously published literature.

- The compressive strength of the bricks decreased with increasing the sludge content and increased with the firing temperature at each tested sludge content; the water absorption increased with increasing sludge content of the bricks and decreased with increasing the firing temperature at each sludge addition; the weight loss on ignition increased with increasing both the sludge addition and the firing temperature.
- At temperatures of 980˚C and above, the optimal sludge dosage to meet the compressive strength requirement of 3N/mm^2 for common bricks (according to US 102: 1995) appeared to be 10%; at 5% sludge dosage, the requirements of attaining a water absorption value less than 15% (according to US 849: 2011) was met. The major factors affecting the engineering quality of the bricks were found to be the proportion of the sludge to be added to the clay and the firing temperature.
- Consequently, the WTS is more suitable for use in the production of common bricks than using it as an energy source due to its low value of the higher heating value in comparison with other readily available wastes.

ACKNOWLEDGEMENTS

We thank the staff of the Mineral Dressing Laboratory, Ministry of Energy and Mineral Development, Entebbe who assisted with the equipment to pulverize the clay samples. We thank the staff of the Animal Science Research Laboratory at the College of Agricultural and Environmental Sciences at Makerere University for providing the Bomb Calorimeter, which was used for the determination of the higher heating value of the samples.

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NOVEL OXYGEN-STEAM GASIFICATION PROCESS FOR HIGH QUALITY GAS FROM BIOMASS

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Article Info:
Received:
13 November 2018
Revised:
3 April 2019
Accepted:
8 April 2019
Available online:
19 May 2019

Keywords:
Biomass gasification
Oxygen reforming
Gas engine

ABSTRACT

We present the design of a novel oxygen-steam-air updraft gasification plant and technological process capable of delivering high quality producer gas. Gas with calorific value of 8 MJ/m^3 is cleaned with an absorption type gas purification unit and then coupled with a piston engine for electricity production with 31% efficiency. We present the results of test runs of the installation with various combinations of gasification agents and plasma torch influence on gas quality and tar removal. We have used a classical hot plasma torch for tar cracking, but it creates a problem with proper gas mixing with very high temperature flame requiring special design of the mixing chamber. In the future development we plan to install a cold plasma reactor for more selective heavy tar reforming prior to gas entering purification unit.

1. INTRODUCTION

Gasification systems can be used for effective energy production in a variety of ways (Arena et al. 2010, Bang-Møller et al. 2010, Dudyński 2018). The conventional solution of generating energy from locally produced biomass waste, wood pellets or chips utilizes steam generation process, which is then used for integrated heat and/or electricity production (Dudyński et al. 2012, Dudyński 2018). Analogous solution has found widespread use in cogeneration installations (Kirsanovs et al. 2017), but the electric energy effectiveness of simple turbines is usually far below 20% for small systems. Improvements are possible with the application of a gas engine, but such solutions are still under development. Systems now in operation – mainly downdraft type or combining wood pyrolysis with gasification of chars – are quite complicated and costly in maintaining (Farzad et. al. 2016). All of those need special electric energy tariffs to operate profitably. More advanced systems, using micro turbines or fuel cells, while promising are currently in research stage, although the technology has been successfully demonstrated in several regions globally, including systems operating in Europe. (Santarelli M. et al. 2017), (Brunaccini G. et al. 2017), (Kupecki J. et al. 2017), (Kupecki J. et al. 2016). Fuel cells-based systems are therefore at the level of subsidized early market penetration and are still under development. Commercial small scale unsubsidized operating systems produce mainly heat or steam used in host production plants (Dudyński 2018) and the market breakthrough for electricity producing units is dependent on significant reduction of construction and operating costs.

The fixed bed, updraft gasification systems are more robust, reliable and easy to scale up to higher capacities when compared to other gasification systems but require effective gas cleaning units and improvements of syngas quality to achieve successful coupling with a piston engine. One of the most promising methods to achieve this objective is to replace the air as a gasification agent with oxygen enriched air mixed with steam to enhance the calorific value of the gas and limit the tar levels (Liu et al. 2018). We developed a gasification fixed bed 2MW unit which can operate with hot air, hot air – steam combination and 40% oxygen air – steam – hot air combination as gasification agents. We use heat from producer gas cooling to heat up the primary and secondary air to temperatures above 250°C. The primary air can be mixed with 186°C steam to form controllable gasifying gases we inject into the chamber in the bottom of the installation. Combination of these two gasses allows us to operate in two distinct modes.

Low bed height – low steam content in primary air which can be applied for wet low calorific fuels. The producer gas has a temperature above 750°C at the outlet, LHV 3,5MJ/m^3 and less than 5g/m^3 of tars.

High bed height - high steam air ratio in primary air, effective for dry, high calorific fuels. The gas temperatures in the gas chamber is below 500°C, the LHV above 5MJ/m^3 and tar content above 10 g/m^3.

For dry wood, we can enhance the effect of mixed gasifying gases by adding air with 40% oxygen content to im-

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Detritus / Volume 06 - 2019 / pages 68-76
https://doi.org/10.31025/2611-4135/2019.13814

prove the producer gas quality. Details of the design and results of syngas generation for dry wood chips and different composition choices of the gasification agent are presented below.

2. THE GASIFICATION UNIT

We have developed a new, oxygen-steam-air driven biomass gasification system and process capable of producing syngas with calorific value up to 8 MJ/m^3 on a tar free basis. This device is an improvement of the biomass gasification units successfully used in many industrial plants in Poland, intended for energy production using waste from technological processes as a fuel (Kwiatkowski et al. 2013, Dudyński 2018). The operating scheme of the system is presented on Figure 1.

The fuel is fed into the gasifying chamber at the upper part of the device through two separate dosing systems located symmetrically on both sides of the unit to secure more uniform level of bed formation. The chamber is 2.5 m in width a 4.5m in height. The bed height is 3,5 m while operating in high bed mode in order to accommodate dry wood chips feed, separated graphically on Figure 1 into ash, gasifying, pyrolysis and drying zones.

At the bottom we placed a sophisticated ash removal system coupled with steam and air injection in the form of four rotating conical grates each equipped with two ploughshares moving ash to the receiving auger presented on Figure 2.

Conical grates operate in the ash and carbon layer at the bottom of the gasifier where the final carbon burn out takes place. A combination of hot air-steam-oxygen rich air is fed into the gasifier through each of the four rotating elements made of cast iron with very high chrome content in order to withstand the temperatures of the process. The

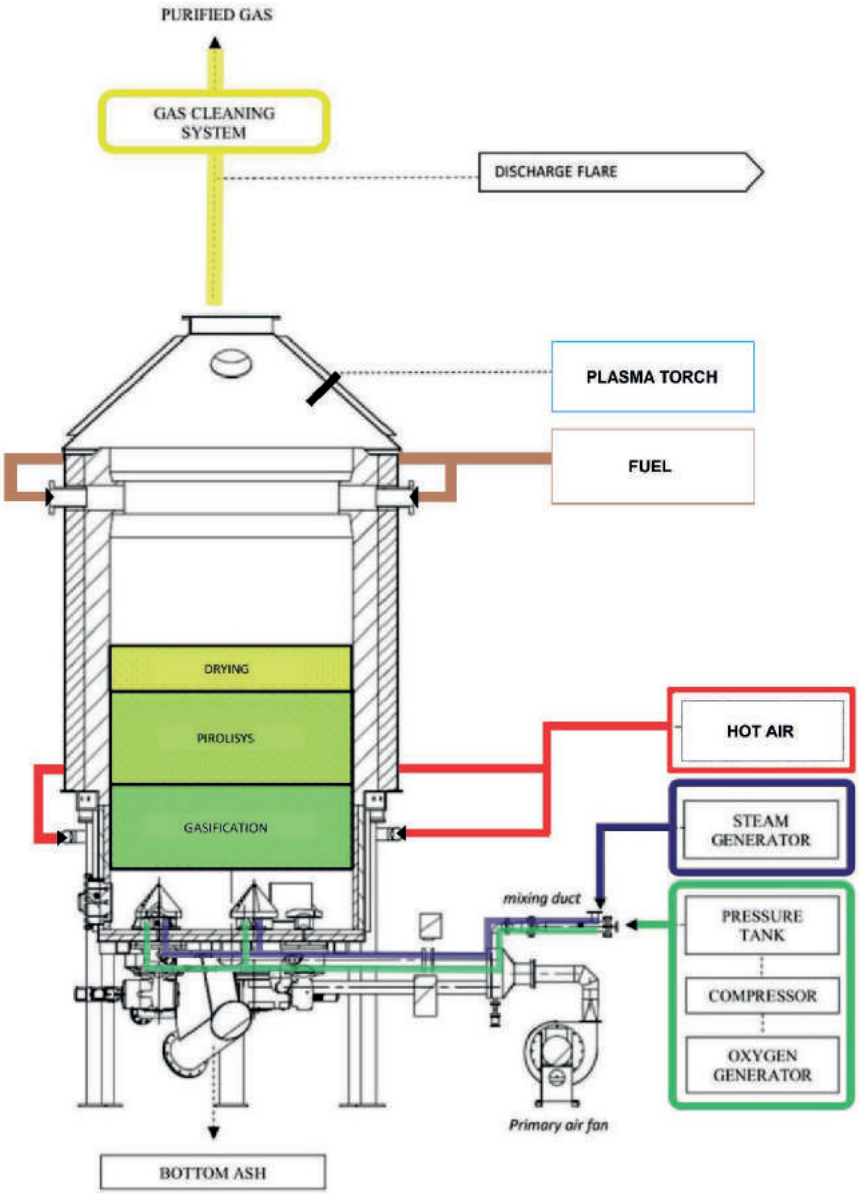

FIGURE 1: Operating scheme of the biomass gasifying unit.

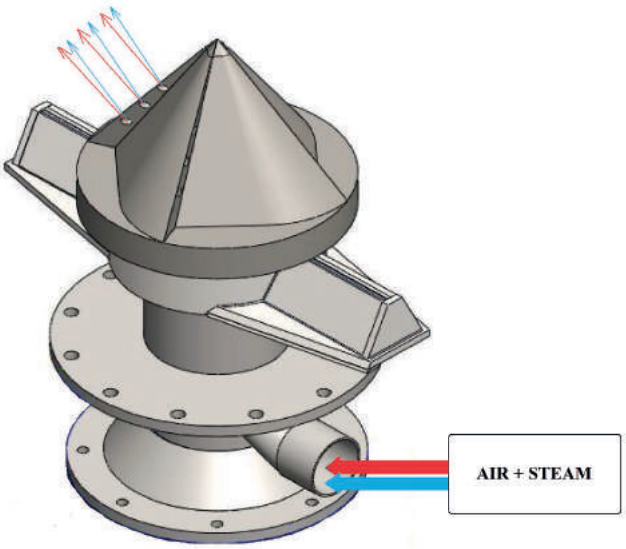

FIGURE 2: The conical moving element.

slow rotating movement of the grate mixes the materials and pushes the ash into the receiving augur.

The heat removed from the producer gas by the gas cleaning system is used to heat up the primary and secondary air to 250°C. The 186°C overheated steam with 4bar pressure is produced in electrically heated steam generator at a rate up to 100kg/h.

The oxygen rich air, containing up to 40% of oxygen and flow speed standing at 100 is produced in a separate unit utilizing the molecular sieves method. These air streams are mixed with steam prior to being applied in gasification process. The role of steam is twofold. First it performs as an oxygen dispersive medium preventing occurrences of high temperature spots at the bottom of the gasifier, where the char and gas burning processes are the most intense. Application of 40% of oxygen air can cause temperatures to locally reach 1800°C causing serious damage to the structure and equipment, with steam and additional air lowering the oxygen content to a maximum of 25% of gas volume, the temperatures in the bottom part of the gasifying chamber are kept below 1200°C ensuring that the operation is smooth and safe. The hot gases move up in the chamber and in the upper part of the gasification unit the overheated steam and CO_2 can react with fixed carbon and tars to produce CO and H_2 in a water shift reaction, efficient in high temperatures:

$$C + H_2O \rightarrow CO + H_2 + 131.28\ \text{kJ/kmol (endothermic)}$$
$$CO + H_2O \rightarrow CO_2 + H_2 - 41.15\ \text{kJ/kmol (exothermic)}$$
$$C + CO_2 \rightarrow 2CO + 172\text{kJ/kmol (endothermic)}$$

improving significantly the producer gas quality and generally lowering temperatures in the carbon burning zone. We constructed a unit allowing for various combinations of gasification gases delivered via a multilayer and multipoint injection system. At the bottom of the gasifier four rotating cones equipped with multiple inlets serve as main injection ports. The oxygen rich air mixed with steam is delivered into the bottom part of the unit through these cones. Approximately 50% of the oxygen necessary for the gasification process is provided through these four injection ports and most of the necessary hot CO_2 and H_2O, is produced in the bottom area of the reactor. The remaining air necessary for the process is injected by 32 nozzles located above the cones, close to the bottom of the pyrolysis zone. The delivered air reacts with the hot pyrolytic carbon producing CO – rich gases and extending the high temperature zone in the gasifier - intensifying the process of wood drying and carbonizing. Such construction guarantees a more uniform distribution of gases and wood chips in the chamber, and therefore improves the mixing of carbonized material with gasifying agents. This enhances the effectiveness of gas production and unification of the temperature's distribution in the gasification process. The control unit allows us to continuously change the parameters and the composition of gasifying gases leading to better control of the producer gas parameters.

Gas leaving the gasification unit still contains high amounts of carbon dust and heavy hydrocarbons as shown in Table 1, which must be removed before gas is fed into the engine.

With such high levels of tars and carbon dust the produced gas requires a very efficient purification system. There are various methods of gas cleaning and virtually every wood gasification unit developed their own unique technology (Boerrgter et al. 2004), (Bocci et al. 2010). In our case we designed an absorptive system presented on Figure 3 coupled with a 20kW plasma torch located at the exit of hot gases from the gasification chamber where an upper part of the chamber was used for syngas mixing with very hot plasma flame. This allows for an analysis of the influence of plasma on the composition of producer gas and heavy tars content.

The gas cleaning system consists of a cyclone integrated with the gasifier outlet (not shown on the drawings), with ceramic lining capable of withstanding temperatures of up to 1200°C, which removes part of carbon dust and as we see on Figure 3 an absorptive unit composed of an air-gas exchanger (cooler 1), water scrubber (3) for dust and impurities removal, two water operated coolers (2) able to lower the syngas temperature to 60°C, an oil scrubber (4) and an active carbon filter (5). On the Figure 4. we present the operating scheme of the absorptive gas cleaning unit where the technology of the process and the mass flow can be seen.

The syngas cleaning takes place by precipitation and removal of tars and heavy hydrocarbons contained therein by adequate cooling of the gas in several stages and absorption of light impurities in the syngas by directing the gas flow through two absorption devices (scrubbers). The first scrubber uses water and the other fuel oil as absorbent. Both scrubbers are equipped with demisters placed immediately before the outlet, which keeps the scrubbing liquids inside the apparatus. Up to 800 m^3/h of gas, at temperatures up to 750°C, resulting from the gasification of wood is transported to the first exchanger (cooler 1) in a counter-current system, where the gas will be cooled to temperatures in the range of 105-110°C and the concurrent air heated up to 250°C.

TABLE 1: Properties of the producer gas.

Substance	Run 1 [mg/Nm³]	Run 2 [mg/Nm³]	Run 3 [mg/Nm³]
Dust	1700	2120	1860
Benzene	73.6	147.3	117.3
Toluene	6050	12700	12810
Xylene	210	450	440
Sum of BTX *	8380	18040	16230
Naphtalene	1.83	0.20	0.21
Acenaphthylene	0.20	0.86	0.77
Acenaphthene	0.05	0.18	0.17
Flurene	0.42	1.26	1.15
Phenanthrene	2.63	7.24	6.43
Anthracene	0.69	1.93	1.76
Fluoranthene	1.22	4.09	4.86
Pyrene	1.19	4.27	5.35
Benzo [a] anthracene	0.33	0.92	1.24
Chrysene	0.28	0.83	1.10
Benzo[b]fluoranthene	<0.01	0.80	1.66
Benzo(j)fluoranten	0.31	0.80	1.61
Benzo(k)fluoranten	0.11	0.31	0.48
Benzo[a]pyrene	0.26	0.75	1.21
Indeno[1,2,3-cd]pyrene	0.13	0.32	0.51
Benzo[ghi]perylene	0.18	0.50	0.77
Sum of polycyclic aromatic hydrocarbon compounds	9.82	25.25	29.27

** Including polycyclic aromatic hydrocarbon compounds*

In the first stage of cooling, heavy tar fractions precipitate and flow down the device from where they can be collected.

Then the gas flows to the water scrubber, where it is cleaned of larger impurities, dust and water-soluble elements. After passing through the water scrubber, the gas moves to the fan and then is pressed to a two-stage heat exchanger system where in the second and third levels of cooling, the temperature of the gas drops below the precipitation point of tar and light hydrocarbons which flow with water down the walls of the exchangers into the lower parts, where a discharge spigot enables the liquefied contaminants to be collected. An important factor determining the possibility of precipitation of tars, preventing the formation of carbon deposits and clogging of exchangers is to maintain appropriate temperatures at the inlet and outlet of each cooler. The gas will then be fed to the absorption column (oil scrubber), where the residual tars will be washed away by the oil. After final cleaning in the active carbon candle filter the gas can be transported to the engine or directed to the combustion chamber.

The oil flowing through the scrubber is pumped to the clarifier and then to a candle filter unit. The purified and cooled oil is then returned to the absorption column (oil scrubber).

The water flowing through the scrubber is pumped to a clarifier and then recycled to the installation.

3. THE RESULTS

The final plant integrates the gasification, purification, oxygen and steam generation units with piston engine, can generate 500kW of electricity and use the excess heat for wood chips drying.

The system can operate on wood chips or pellets, with both air-steam mixtures and oxygen enriched gasification gases. We tested the process with dry wood chips with 20-25% humidity and average diameter of 10-30 . We have compared the producer gas quality during simple air gasification (Run 1) with mixture of air and steam (Run 2) and finally with steam-oxygen-air composition (Run 3). The results are summarized in Table 2.

The amount of wood used in each test was dependent on the effectiveness of the gasification process as the level of material in the gasifying chamber was kept constant. In all tests the amount of oxygen in gasification gases was also kept constant and equal to 120 kg/h with different combination of air, steam and oxygen enriched air making the results directly comparable. We observe the differences in thermal output of each run showing the dynamics of the process measured in effective amount of wood gasified during each test.

We have introduced changes of gasification gases during consecutively executed runs as presented on Figure 4. This allows for a direct comparison of the effects of dif-

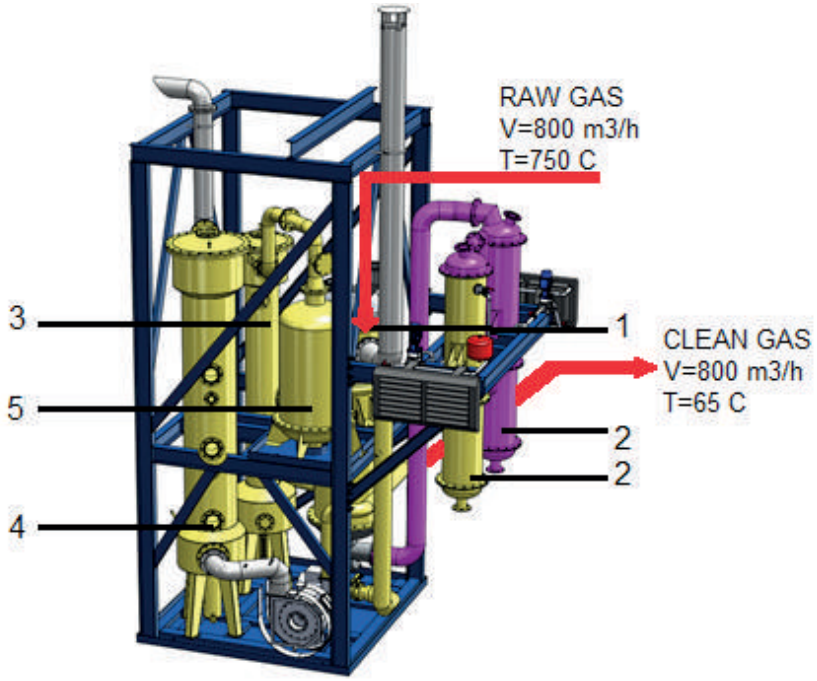

FIGURE 3: 3D scheme of gas purification unit. 1. First cooler, 2. Water coolers, 3. Water scrubber, 4. Oil scrubber, 5. Carbon filter.

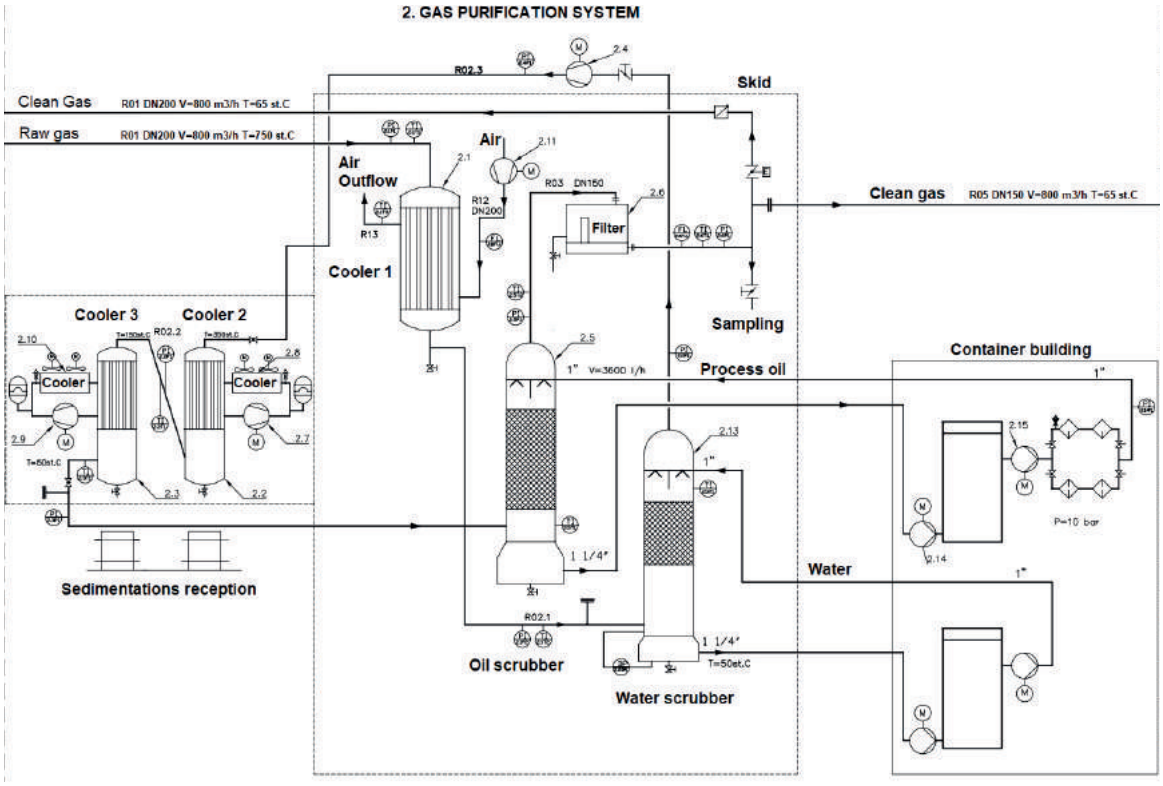

FIGURE 4: Operating scheme of a gas purification unit.

ferent factors on the quality of obtained producer gas and the stability of the process. We have measured the tar contents of the gas leaving the gasifier and the composition of tars collected in various stages of the purification process during each run. On Figure 5 we present the gas composition and LHV of the gas leaving the purification unit.

The results clearly indicate that the application of various combination of oxygen, steam and air composition significantly influences the properties of the producer gas leading to significant grow of calorific value but not necessary lowering the amount of tar and light hydrocarbons in the raw gases as is clearly seen in the Table 1. The compo-

TABLE 2: Parameters of the process and properties of the producer gas after purification.

	Run 1	Run 2	Run 3
Parameter			
Thermal output [MW]	1.35	1	1.8
Fuel [kgh^{-1}]	300	225	380
Air flow [m^3h^{-1}]	600	500	400
Oxygen content [%]	21	21	25.75
Steam flow [kgh^{-1}]	0	20	20
Syngas Parameters After Purification			
CO [%]	29.30	26.47	34.05
H_2 [%]	10.75	8.35	22.30
CH_4 [%]	2.57	1.80	4.04
CO_2 [%]	8.42	7.56	16.39
Syngas LHV [MJm^{-3}]	5.45062	4.58903	7.76995

sition of tar in water collected at the third cooler is presented in Table 3.

These results are representative for all the condensates collected in all three coolers where we find mainly water and the hydrocarbons content does not exceed 0.5% of the collected water.

Finally, we chose the oxygen, air and steam composition for the long – term tests including integration of the gasification unit with a piston engine for electricity generation. The time stability of the clean gas composition generated during a test is presented on Figure 6.

The gas leaving the purification system is relatively almost tar free as shown on Table 4, with both heavy and lights hydrocarbons components virtually absent.

This shows that the absorptive systems are too reactive – removing not only heavy but also light hydrocarbons as well and unnecessarily lowering the calorific value of the gas, while generating more waste water and oil, then necessary for the process of production of fuel-grade syngas. More selective systems, eliminating heavy hydrocarbons only, can be less cumbersome and more effective for this purpose and shall be looked for.

We observe that the plasma torch seems to lower the amount of heavy tars transforming them into light hydrocarbons, which is promising, but a 20KW torch is not enough to replace the purification unit for 800 m^3/h volume of producer gas and a much powerful unit will be tested but it shall require a significant amount of produced power to

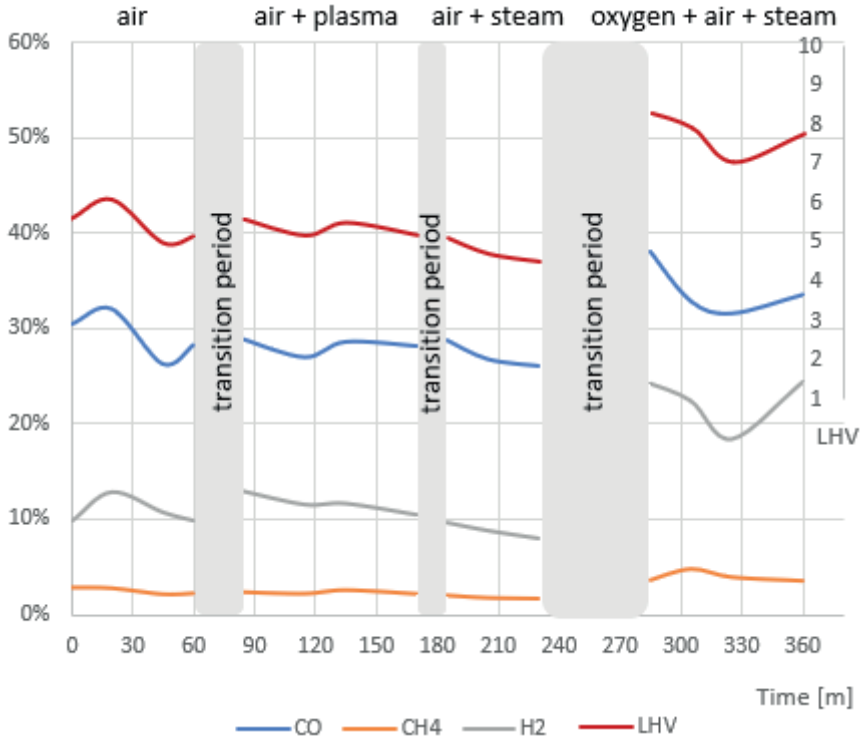

FIGURE 5: The composition and LHV of producer gas.

TABLE 3: Hydrocarbon concentration in condensed water from cooler III.

Compound	Retention time	Concentration [%]	Crystalization temperature [°C]
Acetaldehyde	5.255	0.022	-123
Butanal	9.227	0.002	-97
Toluene	14.501	0.006	-95
Acentone	6.68	0.026	-94
2-Butanone,3-methyl-	11.323	0.001	-92
2-Propenal	6.575	0.001	-88
2-Butanone	9.353	0.009	-86
2-Pentanone	12.134	0.002	-86
Methacrolein	8.603	0.003	-81
Cyclopentanone,2-methyl-	16.141	0.002	-75
Cyclopentanone	14.972	0.002	-58
3-Buten-2-one,3-methyl	11.747	0.005	-54
2,3-Pentanedione	12.312	0.002	-52
Indene	19.98	0.004	-25
1,3,5,7-Cyclooctatetraene	17.262	0.002	-4
2,3-Butanedione	9.084	0.015	-3
2-Cyclopentene-1-one, 2-methyl-	17.466	0.002	2
Benzene	11.519	0.034	5.5
Creosol	21.776	0.005	5.5
Phenol, 2-methoxy-	20.445	0.003	28
Phenol, 2-methyl-	19.799	0.004	34
Phenol, 2-methyl-	20.083	0.002	34
Phenol	18.643	0.002	40.5
Phenol, 2,6-dimethyl-	20.691	0.001	47
Phenol, 2,6-dimethyl-	21.125	0.003	47
4-Methylphthalic anhydride	19.175	0.001	91
Azulene	22.028	0.006	99

TABLE 4: Hydrocarbon concentration in clean producer gas.

Name	Retention time	Concentration [%]	Concentration [mg/Nm^3]	Temperature of crystallization [°C]	Boiling temperature [°C]	Molar mass [g/mol]
2-Propanol, 1-Methoxy	9.709	0.001	11	-96.00	120.00	90.12
Toluene	12.283	0.010	110	-95.00	110.6	92.14
Formic acid, 10methylethyl ester	7.290	0.003	33	-93.00	68.00	88.11
2-Butanone	7.712	0.002	22	86.00	79.6	72.11
Isopropyl acetate	9.240	0.010	110	-73.00	89.00	102.13
Nonane	15.580	0.001	11	-53.00	150.4	128.26
Benzene	9.268	0.035	385	5.5	80.1	78.11
Propanoic acid, 1-methylethyl ester	11.692	0.001	11	-88.00	108.00	116.16
5-Norbornane-2-carboxaldehyde	18.484	0.005	55	-	67 (12 mm Hg)	122.16
Bicyclo[3.2.0]hept-2-ene, 2-methyl	18.885	0.001	11		158.00	108.10
2-Acetyl-5-norbornene	20.125	0.003	33		191.00	136.19
SUM	-	0.074	803	-	-	-

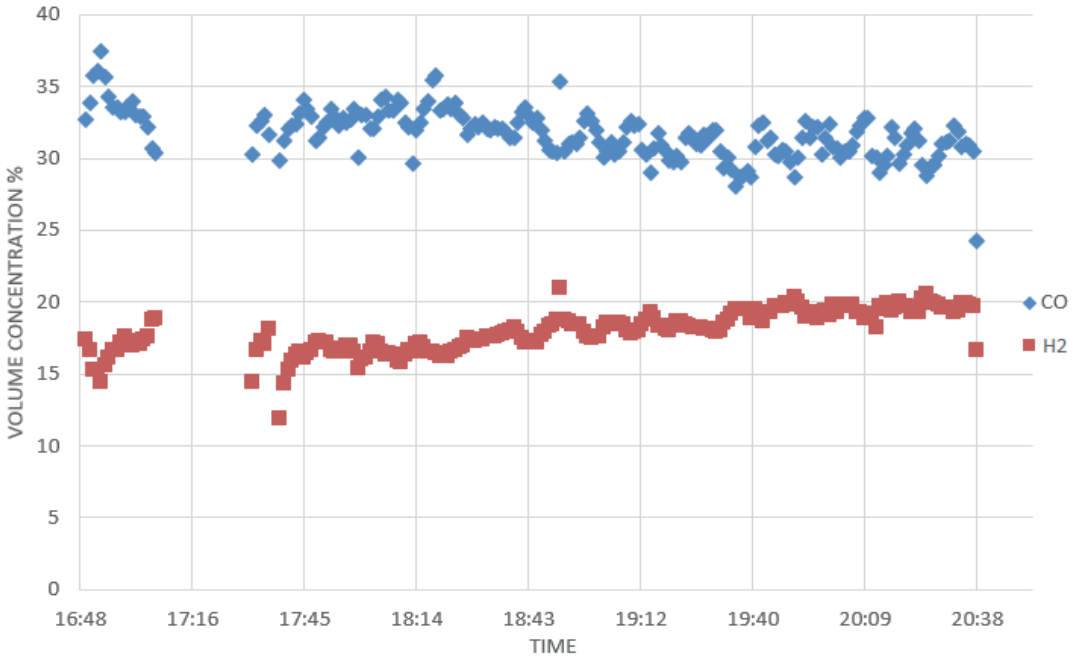

FIGURE 6: Composition of gas leaving the purification unit.

be spent on gas purification lowering the effective efficiency of the system.

The gas leaving the unit is wet, thus it is necessary to heat it up to 70°C with heat exchanger in order to successfully feed it to the engine. The whole unit with necessary adjustments in the engine control system, introduced to account for the fluctuations of the gas quality, worked satisfactory and achieved 31% of overall electric efficiency. We are currently working to reuse the spent water from the scrubber for steam generation and utilize the tars separated in coolers or used scrubber oil as a source of heat in the steam production unit.

4. CONCLUSIONS

There are many steam-oxygen gasification schemes recently tested (Kurkela et al. 2016, Broer et al. 2015) on laboratory or small-scale units indicating the potential of this method to improve the technology of biomass gasification (Baláš et al. 2016). We used the robust fixed bed, updraft gasifier for conducting such tests on the moderate 1.5 MW scale. Application of the oxygen-steam-air combination improve significantly the LHV of the producer gas to the levels comparable with downdraft gasifiers (Wei et al. 2009) cited the gas calorific value of gas for downdraft system as 6.7 MJ/m^3 and 14mg/m^3 of tar while (Omar at al.2018) with application of the hot air and steam gasification were able to reach 8MJ/m^3 and 1.8g/m^3 of tar for small scale 24kW apparatus. We were able to achieve stable production of equally high quality, tar-free, 8 MJ/m^3 producer gas which was used in an engine for uninterrupted electric energy generation with overall 31% efficiency. These results prove that such technology can be commercially applied for effective heat and electricity generation. Excess of heat is used for drying wood chips while waste oil, water and tars are to be used internally in the process.

ACKNOWLEDGEMENTS

This research was partially supported with ERA Bioenergy Advanced Biomass CCHP on gasification, SOFC and cooling machines ERA grant.

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detritus | Multidisciplinary Journal for Waste Resources & Residues

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THE ACTUAL IMPACT OF WASTE-TO-ENERGY PLANT EMISSIONS ON AIR QUALITY: A CASE STUDY FROM NORTHERN ITALY

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Article Info:
Received:
29 November 2018
Revised:
5 April 2019
Accepted:
29 April 2019
Available online:
24 May 2019

Keywords:
Actual WTE emissions
Traffic emissions
Ambient air quality
Environmental impact assessment

ABSTRACT

In recent decades there has been an intense debate about the impact of waste-to-energy (WTE) plant emissions on air quality, and therefore on public health. Currently available data from emissions inventories show the negligible impact of waste incineration on air quality. A number of impact assessment studies are currently available too. A few of them are site-specific, but none of them makes a direct comparison between the local impact of the emissions from a WTE plant and emissions from other "common" sources (such as vehicles and domestic heating) perceived as less potentially hazardous in the public opinion. This paper examines the impact on air quality of actual emissions from a WTE plant in the municipality of Desio (located to the North of Milan in the Lombardy region) using CALPUFF atmospheric dispersion model. Continuous emission monitoring data were used to measure the plant's actual emissions of PM10, NOx, cadmium, and dioxins (PCDD/F) as inputs for model simulations. For comparison purposes, the impact of traffic emissions along the main roads in Desio was also simulated. The results of the model show that the WTE plant has a marginal impact on local air quality compared with pollution from vehicular traffic. The contribution of the plant's actual emissions to ambient pollution concentration levels in the urban area of Desio is between two (NOx and cadmium) to four (PM10 and PCDD/F) orders of magnitude smaller than the contribution from road traffic emissions.

1. INTRODUCTION

Despite ever stricter limits on atmospheric emissions from waste incineration plants and technological improvements in flue gas treatment (Passarini et al., 2014), there is still considerable public concern regarding the potential adverse health effects of waste incineration. Significant positive relationships with broad groups of congenital anomalies in populations living near waste incinerators have been reported in a number of epidemiological studies. However, the results from these studies remain inconclusive due to the limitations of exposure assessment, possible confounding risk factors, and lack of statistical power (Ashworth et al., 2014).

Waste incinerators and state-of-the-art waste-to-energy (WTE) facilities frequently face strong opposition from local communities, making the location of new plants an ongoing concern (Achillas et al., 2011; Ren et al., 2015; Baxter et al., 2016; Song et al., 2017). Health risk assessment studies for new plants are usually based on both maximum plant throughput (i.e. the maximum volume of waste the plant is authorised to burn) and maximum stack concentrations (i.e. the concentration limit set by regulations for pollutants contained in flue gas emissions). Thus, these studies are based on the maximum mass flow rate of pollutants and provide upper-bound estimates of the impact of the WTE plants on air quality. In practice, plant throughput is typically close to, but still below, the authorised value, while actual pollutant concentrations in flue gas are well below the limit values, sometimes by several orders of magnitude. Given such a prudent approach, health risk assessment studies indicate acceptable incremental risk for the exposed population. Nevertheless, risk perception among the public is biased by a number of factors (Ren et al., 2015), primarily still-low levels of environmental education (i.e. risk awareness and knowledge). Improved communication with regard to risk levels and public engagement (Lidskog and Sundqvist, 2004; Chung and Kim, 2009) may reduce undue risk perception (Petts, 1992). Additionally, informed public opinion of projected benefits and risks, together with increased trust in local government, monitoring bodies and plant operators can further foster public acceptance of potentially hazardous facilities (Mah et al., 2014; Liu et al., 2018).

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Detritus / Volume 06 - 2019 / pages 77-84
https://doi.org/10.31025/2611-4135/2019.13817

Public perception of a plant's impact on air quality can be profitably shaped by studies comparing the contribution of WTE plants to local air pollution with that of other common sources such as traffic, domestic heating or biomass burning, which are publicly perceived as less detrimental to human health yet which may contribute significantly to ambient concentration levels locally. However, such studies are extremely rare in the literature, where comparative studies are usually limited to comparisons of emission inventory data (thus ignoring flue gases released into the atmosphere) or to comparisons with alternative solutions to WTE plants, mostly based on the life-cycle assessment approach (Evangelisti et al., 2015; Dong et al., 2018).

In order to bridge this knowledge gap, this paper takes the WTE plant operated by Brianza Energia Ambiente SpA in the city of Desio (Northern Italy, Lombardy region) as a case study. It has three main aims:

- to assess the plant's impact on local air quality based on actual emission data, in order to examine its impact in more detail and make the findings public knowledge for the benefit of the local population;
- to assess the impact on emissions and local air quality of the most recent plant retrofit, which was carried out in 2016 and involved: i) a 40% increase in incineration capacity (up to a throughput of about 20 tons per hour); ii) the installation of a new steam turbine; iii) the revamping of the flue gas treatment system, specifically with the installation of a new SCR (selective catalytic removal) unit for NOx emission control;
- to compare the impact of the plant's stack emissions on air quality with that of ground-level road traffic emissions from the main roads in the municipality of Desio. Traffic emissions from minor roads and the general urban road network were not considered.

The study considered two criteria pollutants, PM10 and NOx, concentrations of which are particularly high in Northern Italy and frequently result in current air quality standards being breached, and two toxic pollutants, cadmium (Cd) and dioxins and furans (PCDD/F) typically emitted by waste incineration plants. In order to evaluate the benefits of the 2016 plant retrofit, emission data from the 2015 (prior to the retrofit) and 2017 (following the retrofit) were compared and separately used for air quality model simulations in two different scenarios.

2. MATERIALS AND METHODS

2.1 Study area

The case study presented in this work is based on the municipality of Desio, a town with a population of 40,000 located in a flat area about 15 kilometres north of Milan in the Lombardy region in Northern Italy. As in most of Northern Italy, air quality is an issue of great concern in Desio, as air quality standards are frequently not respected (Directive 2008/50/EC). As the Lombardy Region's air quality monitoring network has no stations in Desio, the only air quality data available come from regional-scale modelling provided by the Lombardy Regional Environmental Protection Agency (ARPA Lombardia). For 2016, the base year considered in this case study, the annual average concentration estimated for PM10 (31.6 $\mu g/m^3$) was in compliance with the 40 $\mu g/m^3$ EU limit. However, the daily average concentration of PM10 exceeded the 50 $\mu g/m^3$ limit 60 times during the year, on significantly more occasions than the permissible 35 times in a year. The annual average concentration of NO_2 (46.4 $\mu g/m^3$) exceeded the 40 $\mu g/m^3$ EU limit, although the hourly average concentration never exceeded the 200 $\mu g/m^3$ concentration limit.

Emission inventory data for Desio (ARPA Lombardia, 2018) are summarised in Table 1 for the pollutants considered in this study, except for PCDD/F, which is not listed in the Lombardy region emission inventory. Among anthropogenic activities, the sector of road transport was the largest source of emissions, accounting for 41.3% of PM10, 49.9% of NOx and 24.6% of Cd emissions. Non-industrial combustion plants (i.e. commercial and residential heating) were responsible for 24.9% of PM10 emissions, 9.3% of NOx emissions and 18.6% of Cd emissions. The corresponding shares for the waste treatment and disposal sector were 6.9% for PM10, 36.8% for NOx, and 16.8% for Cd.

2.2 Air quality model

The study was conducted using the CALPUFF model (www.src.com), a tri-dimensional air quality model particularly suitable for complex meteorological conditions with light winds and calms, which frequently occur in Northern Italy's Po Valley plain. CALPUFF calculates the time series of 1-hour ground-level concentrations at each grid node of the computational domain. The raw data were then processed by the CALPOST module in order to obtain summary statistical data (e.g. average annual concentration, hourly maximum/daily average/percentile values) for graphical representation through maps showing isoconcentration contour lines. The computational domain was a 10 by 10 kilometre square grid around the WTE plant with 100-metre grid step, giving a total of 10,201 grid points.

Meteorological data were supplied by the Regional Environmental Protection Agency and from the LAMA (Limited Area Meteorological Analysis) database, which was generated using the COSMO meteorological model and GTS data. COSMO is the Italian model for short-term weather

TABLE 1: Emission inventory data for the municipality of Desio (Arpa Lombardia, 2018 - reference year 2014).

Source category	PM10 (t)	Cd (kg)	NOx (t)
Non-industrial combustion plants	7.675	0.168	31.5
Industrial combustion plants	2.220	0.037	10.3
Industrial processes without combustion	0.245	0.003	0.0
Use of solvents and other products	2.910	0.004	0.0
Road Transport	12.908	0.222	168.1
Other mobile sources and machinery	0.162	0.001	2.8
Waste treatment and disposal	2.166	0.152	124.1
Agriculture	0.003	0.000	0.1
Other sources and sinks (nature)	2.944	0.317	0.1

forecasts. In order to make it easier to assess the impact of the plant retrofitting operation on air quality, the same meteorological data for 2016 were used for both 2015 and 2017. The ground-level (10-metre height) wind rose for the plant site in the 2016 calendar year is shown in Figure 1. Winds were mostly northerly, blowing from sectors between the North-West and North-East, but also along the East-West axis. Conversely, southerly winds, coming from the directions between South East and South West, were markedly less frequent.

2.3 Emission data

2.3.1 WTE plant

A waste incineration plant began operating on the outskirts of the municipality of Desio in 1976. Since then, the plant has been retrofitted several times, most recently in 2016, in order to meet emission standards issued in recent years. Currently, the Desio waste incineration plant is a WTE plant with two combustion lines, each with a maximum power output of 21 MWt, and operates according to a Combined Heat and Power (CHP) energy recovery scheme. The two combustion chambers are equipped with moving grates and integrated boilers for a total steam production of 45 t hr^{-1}. Power production relies on a turbine with a maximum power output of 8.25 MW. Heat is supplied to a 40-km district heating network, serving Desio and three other municipalities nearby. The maximum thermal power output to the district heating network is 30 MWt. The plant is mainly fed with municipal solid waste (77%) together with some commercial and industrial waste (16%), medical waste (4%) and sludge (3%). On average, the lower heating value of the waste is about 12 MJ t^{-1}.

Atmospheric emission control is performed through a flue gas dry treatment line. Dust emissions are controlled by means of an electrostatic precipitator and a baghouse filtration unit. Acidic gases are controlled through a double-alkali injection system (one injection point in the post-combustion chamber and one before the baghouse); dioxins and other organic pollutants are treated by means of activated carbon injection before the baghouse filter. NOx emissions are controlled via a two-stage SNCR/SCR system, with the final SCR system introduced in 2016 retrofit. The plant retrofit did not affect the stack of the plant: treated flue gas is released to the atmosphere through a 47-m tall stack.

For this study, hourly data for flue gas temperature and speed, dust (considered as PM10) and NOx (as NO_2) concentrations were taken from the continuous emission monitoring (CEM) system database for the 2015 and 2017 calendar years. Thus, model simulations for PM10 and NOx were based on their actual mass flow rates. For Cd and PCDD/F, which were not continuously monitored, mass flow rates were determined based on the actual flue gas flow rate from CEM data and on fixed concentration data representative of the plant emissions. In particular, concentration data for Cd were taken from the datasets of the discontinuous sampling performed every year: namely, following a precautionary approach, the highest of the three concentration values available for each year was used (0.193 μg m^{-3} in 2015 and 0.323 μg m^{-3} in 2017, respectively). Concentration data for PCDD/F (as equivalent toxicity – I-TEQ) were taken as the annual sets of 12 monthly-averaged concentrations derived from the analysis of the integrated samples collected by the PCDD/F continuous sampling system (Table 2). Therefore, the time series of

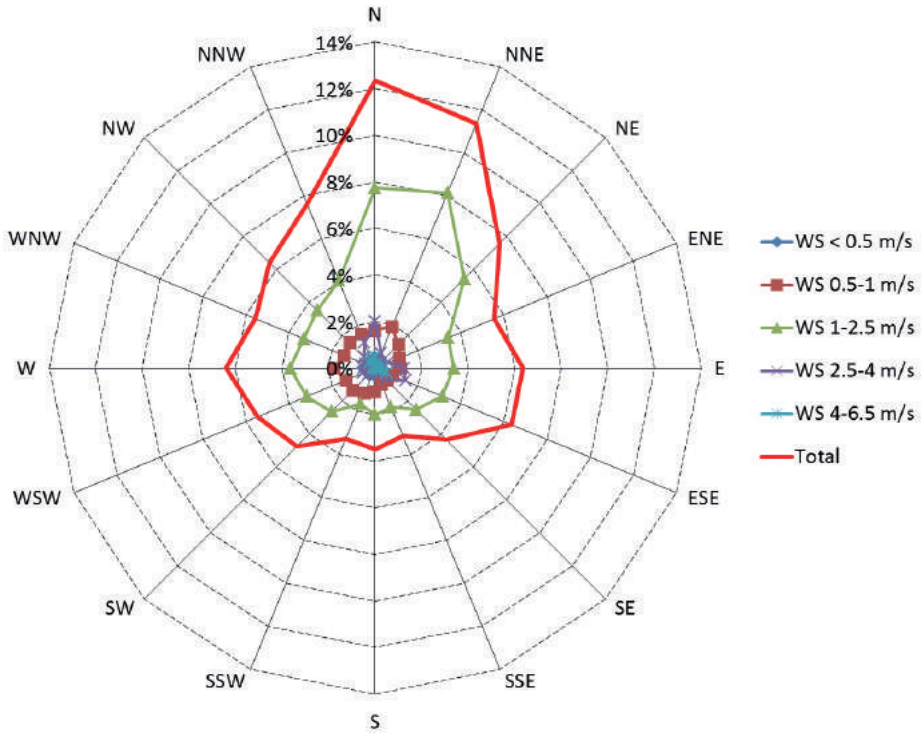

FIGURE 1: Ground-level wind rose for calendar year 2016.

TABLE 2: Monthly averaged PCDD/F concentration (pgTEQ m^{-3}, normal conditions, dry gas 11% O2) from PCDD/F continuous sampling system.

Month	Year 2015	Year 2017
Jan	1.10	0. 41
Feb	0. 61	0. 26
Mar	0. 71	0. 48
Apr	1.00	0.74
May	1.06	0.76
Jun	0. 43	0.04
Jul	1.25	0.71
Aug	0. 72	0.67
Sep	1.71	0.41
Oct	1.47	1.35
Nov	1.15	0.71
Dec	2.54	0.61(*)

(*) Missing December 2017 data the average concentration of the previous months was used

PCDD/F mass flow rates accounted for both flue gas variability (at hourly resolution) and concentration variability (at monthly resolution).

CEM data for 2017 show that the plant retrofit resulted in both a higher flue gas stack temperature (162°C vs. 144°C) and speed (11.4 m s^{-1} vs. 9.4 m s^{-1}) than in 2015, thus leading to a stronger vertical momentum and buoyancy, both favouring the plume rise and the dispersion of the pollutants into the atmosphere.

In both scenarios pollutant concentrations and mass flow rates were always lower than regulatory limits and authorised mass flow rates (Table 3). PCDD/F and Cd maximum hourly flow rates were well below emission limits (two orders of magnitude lower for PCDD/F and even lower for Cd), five times below emission limits in 2015 and two orders of magnitude lower in 2017 for PM10, 58% of the authorised flow rate in 2015 and 68% in 2017 for NOx. However, as NOx and PCDD/F average flow rates were lower in 2017 than in 2015, Cd and PM10 flow rates were higher in 2017 than in 2015.

2.3.2 Road traffic

Hourly emissions from traffic on the main roads around the plant were estimated, based on the length of the road segments, on traffic volumes on any segment, and on emission factors for each vehicle type. Traffic flow rates along the road segments were evaluated by means of a dedicated study which assessed hourly traffic volume broken down into three main classes of vehicles (cars, vans and trucks) during a workday rush hour. The study was based on both transport supply system data (i.e. the structure of the road network) and mobility demand data (i.e. an origin-destination trip matrix), together with traffic flow data for the Lombardy region and for the municipality of Desio.

Emission factors were taken from the Lombardy Region Atmospheric Emission Inventory (INEMAR - ARPA Lombardia, 2018), the Italian traffic emissions factors database (ISPRA, 2017) and the EMEP/EEA air pollutant emission inventory guidebook 2016 (EMEP, 2016). Specifically, emission factors for PM10 were taken from the INEMAR inventory, those for NO_2 and Cd from the ISPRA database, and those for PCDD/F from the EEA guidebook. The INEMAR and ISPRA databases directly provide average emission factors for the three vehicle classes: estimates for these values were based data on the composition of the regional and national vehicle fleet, mileage and average speed for urban and extra-urban driving cycles. Emission factors for PCDD/F provided by the EEA Guidebook were combined with data on the vehicle fleet in the Lombardy region in order to estimate average emission factors for the three classes of vehicles considered in this research.

The estimated daily emission pattern was applied for the whole year, without any seasonal or weekly variation. Although this approach may overestimate the contribution of main road traffic, it partially compensates for emissions from traffic on minor and local roads not considered in the calculations. In practice, a comparison between the traffic emission calculations used for this study and the emission inventory data for road traffic in the municipality of Desio in 2014 indicates that the current study underestimates emissions by 30%.

3. RESULTS AND DISCUSSION

3.1 WTE plant contribution

Isoconcentration maps for the estimated contribution of emissions from the WTE plant to annual average concentrations of NO_2 are presented in Figures 2 and 3 for the 2015 and 2017 scenarios, respectively. The spatial distribution of the concentration levels was similar for all pollutants in both years, as the same meteorological data were used as inputs. Differences in concentration levels are due

TABLE 3: Comparison between actual (Scenario 2015 and Scenario 2017) and authorized hourly flow rates of PM10, NOx, Cd, PCDD/F from WTE plant.

Parameter	PM10 (g h^{-1})		NOx (kg h^{-1})		Cd (mg h^{-1})		PCDD/F (ng_{TEQ} h^{-1})	
	2015	2017	2015	2017	2015	2017	2015	2017
Average	15.5	20.9	7.38	4.00	11.4	22.2	69.1	39.9
Median	10.8	21.6	7.38	4.00	11.6	23.3	64.8	40.7
Minimum	0.4	1.4	0.07	0.22	0.6	4.5	3.2	1.8
Maximum	205.6	72.7	12.82	14.54	15.4	29.4	202.7	110.2
Max Authorized	1,100		22		5,500		11,000	

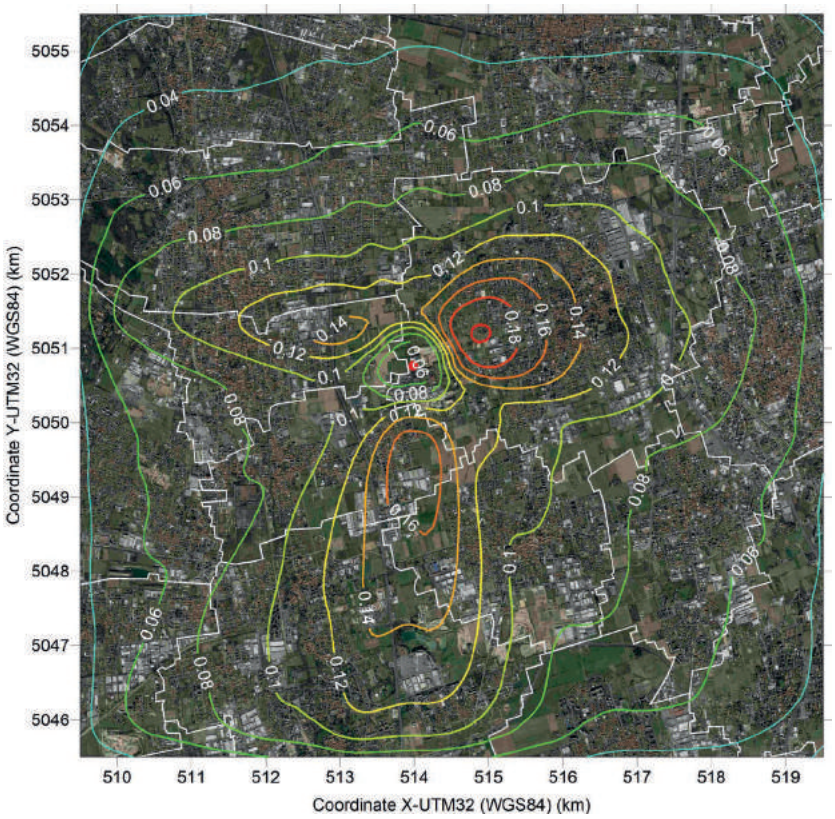

FIGURE 2: Emission scenario 2015: contribution of WTE plant emissions to NO_2 annual average concentration (μg m^{-3}) before of plant retrofit. (red dot: WTE plant; white lines: municipality borders; aerial map source: Google Earth).

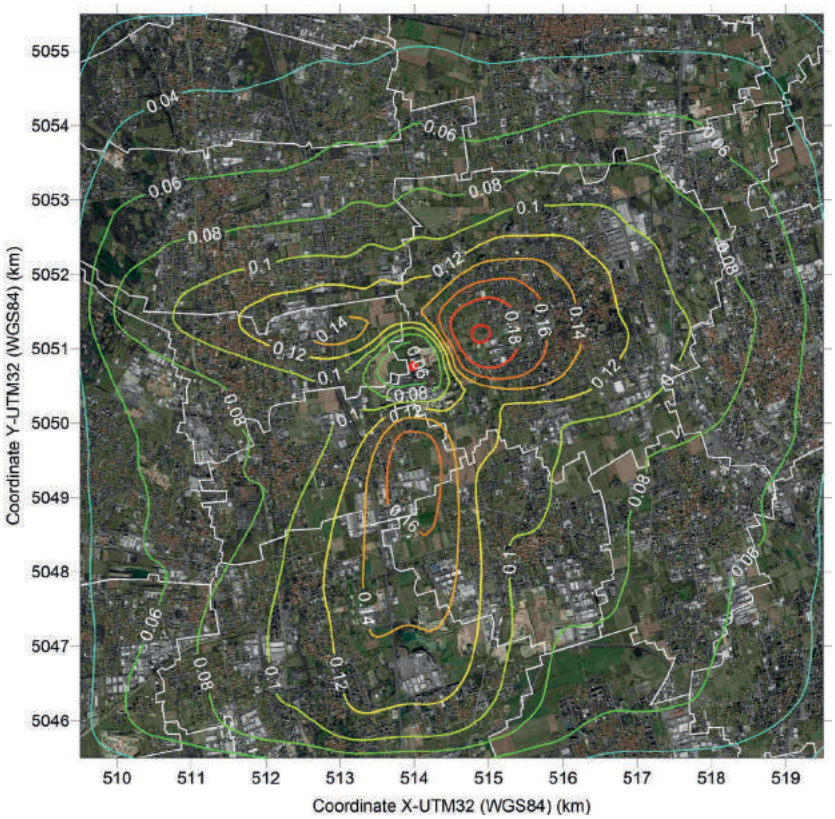

FIGURE 3: Emission scenario 2017: contribution of WTE plant emissions to NO_2 annual average concentration (μg m^{-3}) after plant retrofit. (red dot: WTE plant; white lines: municipality borders; aerial map source: Google Earth).

to the emissions rate only. The distribution pattern shows three lobes, extending North-West, North-East and South of the plant, according to the most frequent wind directions in the area. The area with the highest impact (i.e. where the maximum contribution is estimated) is located about 1 kilometre North-East of the plant, on the outskirts of the Desio urban area. However, emissions from the plant also affect air quality in the centre of the town. Table 4 summarises the values at the maximum fallout point and the annual average concentration of pollutants at ground level within the residential area of Desio before and after the WTE plant retrofit (i.e. the scenarios for 2015 and 2017). The table also compares emissions with air quality regulatory limits (EU Directive 2008/50). Plant emissions were found to have an extremely limited impact on air quality, both before and after retrofitting. After the retrofit, plant emissions accounted for 0.001% of the acceptable limit for PM10, 0.2% of NO_2, 0.05% of Cd and 0.0005% of PCDD/F as set out in German regulations (LAI, 2004). The impact of emissions from the WTE plant were also negligible as far as regulatory short-term concentrations for PM10 and NO_2 are concerned.

3.2 Traffic contribution

An isoconcentration map for the estimated contribution of road traffic to the annual average concentration of NO_2 is presented in Figure 4. As traffic was modelled considering the main roads only and due to its ground-level emission, the resulting spatial distribution has a less regular pattern than the one obtained for the WTE plant and is essentially driven by the layout of the road network under consideration. Maximum pollutant concentrations occurred in proximity to the road axes and decreased sharply with distance from them. In particular, the impact of emissions from the major roads 35 and 36 are clearly visible with their North-South alignment on the left and the right side of the map, bordering the municipality of Desio.

The highest annual average values were in the order of 5-6 μg m^{-3} for PM10, 15-20 μg m^{-3} for NO_2, 0.08-0.1 ng m^{-3} for Cd, and 2-3 fg_{TEQ} m^{-3} for PCDD/F. For the residential area of Desio, away from the main roads, the model results show concentration levels ranging between 2-3 μg m^{-3} for PM10, 6-10 μg m^{-3} for NO_2, 0.02-0.03 ng m^{-3} for Cd, and between 0.5-1 fg_{TEQ} m^{-3} for PCDD/F (Table 5). The highest short-term concentrations were in the 3-6 μg m^{-3} range for 24-hour PM10 and in the 100-150 μg m^{-3} range for 1-hour NO_2, roughly 10% and 50-70% of the corresponding air quality limits (50 μg m^{-3} for PM10, 200 μg m^{-3} for NO_2).

3.3 Comparison between source contributions

A comparison between the estimated annual average concentrations due to the emissions from the WTE plant and from traffic on the main roads crossing Desio municipality shows the much higher role of traffic on local-scale air pollution. This is true both for the contribution estimated at the point of maximum fallout and within the residential area of Desio. The contribution from traffic to any of the pollutants under consideration is at least two orders of magnitude higher than the contribution from the WTE plant in the residential area of Desio (Table 5). Contributions from traffic to PM10 concentrations are four orders of magnitude (10,000 times) higher than those from the WTE plant. For PCDD/F concentrations the corresponding figure is three orders of magnitude (1,000 times) higher, while for NO_2 and Cd concentrations it is two orders of magnitude (100 times) higher. For short-term concentrations, similarly, the impact of traffic is far higher than the impact of the WTE plant in the residential area of Desio: the contribution of traffic to PM10 and NO_2 emissions is about 1,000 times and 40-50 times higher, respectively, than from the WTE plant. Similar considerations hold for the point where the maximum fallout from the WTE plant is expected, although this is located at a considerable distance from the main roads and is thus affected less by traffic emissions.

The results of the study show that road traffic is responsible for about 20% of annual average NO_2 concentrations and 10% of annual average PM10 concentrations in Desio. For Cd levels, an assessment of the role of road traffic emissions cannot rely on specific values for Desio because of the lack of an air quality monitoring station; additionally, regional-scale modelling data provided by ARPA Lombardia do not include Cd. However, annual average Cd data from monitoring stations in Milan in 2017 were in the 0.27-0.33 ng m^{-3} range. Thus we may reasonably assume that road traffic is responsible for roughly 10% of the annual average concentration of Cd, in line with its estimated contribution to PM10 levels.

Putting the plant's estimated contributions to PCDD/F air pollution levels in context is harder as this pollutant is not considered by air quality standards and thus not rou-

TABLE 4: Annual average concentration at the maximum fallout point, concentration range within the residential area of Desio and air quality limits for PM10, NO_2, Cd (EU Directive 2008/50/EC) and PCDD/F (German guidelines on air quality: LAI-Laenderausschuss fur Immissiosschutz).

Scenario	Parameter	PM10 (μg m^{-3})	NO_2 (μg m^{-3})	Cd (ng m^{-3})	PCDD/F (fg_{TEQ} m^{-3})
2015	Maximum value	$5.2 \cdot 10^{-4}$	0.20	$3.3 \cdot 10^{-4}$	$2.3 \cdot 10^{-3}$
	Desio residential area (range)	$3\text{-}4.5 \cdot 10^{-4}$	0.12-0.18	$2\text{-}3 \cdot 10^{-4}$	$1.2\text{-}1.8 \cdot 10^{-3}$
2017	Maximum value	$4.4 \cdot 10^{-4}$	0.08	$4.8 \cdot 10^{-4}$	$8.1 \cdot 10^{-4}$
	Desio residential area (range)	$2\text{-}3.5 \cdot 10^{-4}$	0.05-0.07	$3\text{-}4 \cdot 10^{-4}$	$5\text{-}7 \cdot 10^{-4}$
Air quality limits (annual average value)		40	40	1	150

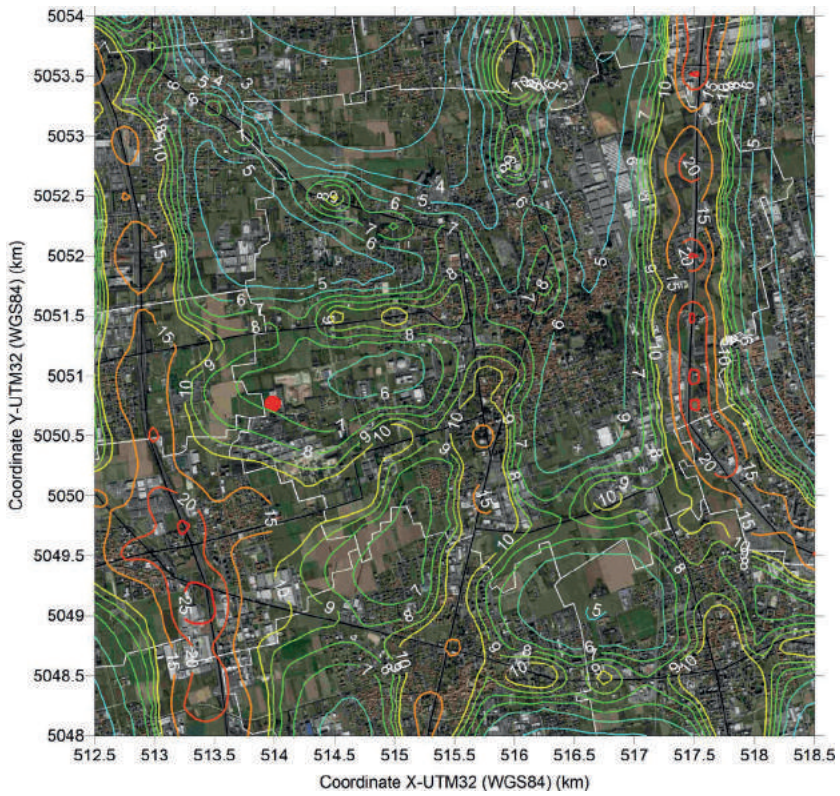

FIGURE 4: Contribution of road traffic emissions to NO2 annual average concentration (μg m^{-3}). (red dot: WTE plant; black lines: main roads considered in this study; white lines: municipality borders; aerial map source: Google Earth).

tinely monitored. Additionally, monitoring campaigns for PCDD/F usually have a limited time-span and do not provide annual average values. Values for PCDD/F concentrations in urban areas vary significantly in the literature, ranging from a few dozen (Turrio-Baldassarri et al., 2005; Schlabach and Tønnessen, 2008) to several hundred fg_{TEQ} m^{-3} (Li et al., 2008). The WHO has estimated average levels of 100 fg_{TEQ} m^{-3} for the urban environment (WHO, 2000). Studies developed in Northern Italy report PCDD/F values in the order of 5-100 fg_{TEQ} m^{-3} (ARPA Veneto, 2016). Such variability is strongly correlated with local factors, mainly related to the presence of industrial plants and the practice of wood burning for household heating. PCDD/F levels in ambient air in proximity to state-of-the-art waste incinerators in Northern Italy have been found to be in the range of 10-125 $fg_{I\text{-}TEQ}$ m^{-3} and between 144 and 337 $fg_{I\text{-}TEQ}$ m^{-3} in proximity to an old plant not equipped with the best available technology (BAT) for the removal of dioxins (Caserini et al., 2004); a more recent study after this plant was retrofitted with BAT reported maximum levels of about 25 $fg_{I\text{-}TEQ}$ m^{-3} as a weekly average for the winter period (ARPA Veneto, 2018). Given this remarkably wide range of values, we can assume that the actual contribution may range from a few percentage points up to 20% of the annual average concentration.

However, all these percentage contributions underestimate the true impact of traffic emissions, as this study only considered the main roads while ignoring the contribution of traffic on the urban minor road network. Taking this latter contribution into account, we may estimate that road traffic is responsible for about 15% of annual average PM10 and Cd concentrations and 30% of NO_2 concentrations.

4. CONCLUSIONS

The results of the model provide evidence of the positive effects of the flue gas treatment retrofit in the Desio WTE plant on NO_x and PCDD/F, emissions of which have been halved. Additionally, the higher temperature and velocity of the effluents following the retrofit have increased the dispersion of pollutants, reducing average annual pollutant concentrations by 60%. Thus the impact on air quality has been reduced, in spite of increased incineration

TABLE 5: Range of estimated contributions from WTE plant and road traffic emissions to the annual average concentrations of PM10, NO_2, Cd, PCDD/F in the residential area of Desio.

Source	PM10 (μg m^{-3})	NO_2 (μg m^{-3})	Cd (ng m^{-3})	PCDD/F (fg_{TEQ} m^{-3})
WTE plant (Scenario 2017)	0.0002-0.00035	0.05-0.07	0.0003-0.0004	0.0005-0.0007
Road traffic	2-3	6-10	0.02-0.03	0.5-1

capacity. Generally speaking, WTE plant emissions have a very low impact on local air quality, both in terms of annual and hourly averaged concentrations and for the main pollutants such as PM10 and NO_2 and for organic micro-contaminants and persistent inorganic toxic pollutants such as dioxins and cadmium.

The extremely low impact of WTE emissions is highlighted by the comparison with the estimated impact of the traffic in the same area, which is likely to be underestimated as the study considers main roads only while ignoring the dispersed, low-speed/high-emission factor, stop-and-go traffic on the network of minor roads. On average, the contribution of traffic in the residential area of Desio is between two and four orders of magnitude higher than the WTE plant impact, both for criteria and toxic pollutants. Considering average annual values, the contribution from traffic is four orders of magnitude higher than that of the WTE plant for PM10, three orders of magnitude higher for dioxins, and two orders of magnitude higher for NO_2 and cadmium. The impact of traffic is even higher when short-term concentration values for PM10 and for NO_2 are considered.

ACKNOWLEDGEMENTS

The research described in this paper was entirely funded by BEA Spa under a research contract with the Civil and Environmental Department (DICA) of the Politecnico di Milano.

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VISUAL INTERPRETATION OF SATELLITE AND AERIAL IMAGES TO IDENTIFY AND STUDY THE EVOLUTION OF INADEQUATE URBAN WASTE DISPOSAL SITES

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Article Info:
Received:
22 January 2019
Revised:
20 May 2019
Accepted:
27 May 2019
Available online:
28 June 2019

Keywords:
Waste disposal sites
Environmental characterization
Historical analysis
Aerial photographs
Remote sensing images

ABSTRACT

One of the main problems Brazil faces in solid waste management is the existence of dumpsites that must be closed. For the shutdown process, an environmental assessment should be performed, which requires knowledge of the history of these sites. This study is often hampered by the lack of past records on the waste disposal activities in these places, especially when they are very old. In this context, the analysis of multitemporal remote sensing images and aerial photographs is an interesting tool for the identification and study of the development of inadequate waste disposal sites. In the present study, these techniques were used to assess former dumps in four municipalities in the state of São Paulo, Brazil. The analysis comprised satellite images and aerial photographs between 1960 and 2010. They enabled the obtainment of information on the progress of activities in the dumps and their surroundings, on how the waste disposal occurred, and on the end of operations and the revegetation process. With geographic information system (GIS) support, information from different data sources was crossed with previous databases, thus allowing the obtainment of greater data structure for a better interpretation of the available information. Thus, the interpretation of multitemporal remote sensing images and aerial photographs allowed a better interpretation of the posterior environmental assessment data through the knowledge of the context in which these were inserted.

1. INTRODUCTION

Proper urban solid waste management represents one of the biggest challenges for Brazilian municipalities. In 2017, the waste generation in Brazil was 78,4 million tons, of which 40,9% was disposed in inadequate sites by more than 60% of Brazilian municipalities (ABRELPE, 2018). This represents approximately 32 million tons of waste disposed in the environment without the necessary structures and control to avoid the contamination of water and soil as well as to protect the surrounding community members' health.

This scenario was even worse from the last century until the first decade of the present century due to the lack of specific regulations towards the topic at the federal level. This means that, in addition to the inadequate sites in operation, there are those used for waste disposal in the past, which still represent possible environmental risks that should be investigated.

There are two classes of inadequate waste disposal sites in Brazil: dumpsites and controlled dumpsites. Dumpsites (or dumps) are the most common, in which waste is simply disposed in surface soil or natural ditches without coverage or soil impermeabilization. The controlled dumpsites (or controlled dumps) have some degree of engineering and management control, such as waste coverage with soil, the construction of ditches or slopes, access control and, less frequently, rainfall, gases and leachate drainage systems.

The Solid Waste National Policy approved in 2010 established that Brazilian municipalities should set goals for the elimination and recovery of dumps and controlled dumps, and the National Solid Waste Plan developed in 2012 set a goal to eliminate all dumps by 2031, promoting the environmentally appropriate final disposal of the waste generated in Brazil.

For the recovery process of these areas, an investigation of soil and water contamination as well as the impacts on human and environmental health must be performed. It is necessary to retrieve the operation history of the dumps,

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https://doi.org/10.31025/2611-4135/2019.13821

obtaining information about the waste distribution and period of disposal, the types of waste disposed, the construction of trenches and slopes, the existence of drainage systems and zones of leachate accumulation, and the use and occupation of the surroundings among others.

However, this investigation is often hampered by the lack of past records in the municipalities and environmental agencies about the inadequate disposal sites, especially when they are very old or have had their operations shut down for a long time. For the dumps that are already closed, the study is also hampered by rapid vegetation growth, making it harder to clearly identify where exactly the waste was disposed. In this context, the utilization of historical aerial photographs and remote sensing images constitutes an interesting tool for the identification and analysis of inadequate waste disposal site evolution, contributing to the process of their correct closure and recovery.

According to the Environmental Agency of São Paulo State (CETESB, 2001), the interpretation of aerial photographs and satellite images for the analysis of waste disposal sites has the advantage of permitting the observation of historical details of their operation, which would not be possible otherwise because of the lack of available documentation. The importance of this analysis is due to the fact that these sites may be used for other activities in the future, presenting risks for human health.

Kuehn & Hoerig (2000) identified the possible uses for remote sensing images and aerial photographs in waste disposal site investigation. The interpretation of these tools permits the study of the chronological development of site operation, including the handling and quantity of dumped waste; the localization of heat sources inside a waste heap; the identification of water seepage at the edges of waste disposal sites; the observation of vegetation succession and vitality; the exploration of the immediate vicinity around a disposal site; the identification of natural and artificial drainage systems; the assessment of surface water conditions; and the identification of previous excavations.

Remote sensing imagery has been largely used by researchers for the identification and mapping of uncontrolled and illegal urban and industrial waste disposal sites. In the 1970s, Garofalo & Wobber (1974) suggested that remote sensing could provide useful solid waste management and planning data. The authors applied aerial photography to estimate waste characteristics and quantities, to study waste disposal site selection and utilization as well as waste collection and transportation, to address environmental impacts of on-site and off-site disposals and to identify waste-generating sources.

Getz et al. (1983) used aerial reconnaissance through a direct aerial search and the photointerpretation of aerial photographs to search for open dumps in the United States. Irvine et al. (1997) made a comparison between historical aerial photography and collected thermal imagery to determine the location of buried industrial waste trenches.

Silvestri & Omri (2008) developed a method for identifying illegal waste disposal sites over large areas using remotely sensed information and a geographic information system (GIS). Glanville and Chang (2015) also mapped illegal disposal through the identification and integration of predictive spatial data in GIS, identifying explanatory variables suitable for predicting its distribution.

More recently, Al-Joburi (2018) used aerial photos, satellite images, digital elevation models and historical land use maps to identify possible subsurface dumpsites in the Kingdom of Bahrain, developing a method to identify, locate and map random dumping sites. Similarly, Massarelli (2018) developed a processing procedure with satellite images applicable to agroecosystems to detect excavation activities and illegal waste disposal in areas that had been heavily disturbed and subjected to continuous change over time.

In Brazil, remote sensing images and aerial photographs have been mainly used to control existing landfills and identify current irregular waste disposal as well as to select suitable areas for the implementation of landfills (Samizava et al., 2008; Coelho, 2017; Carrilho et al., 2018). However, little research has been conducted on the identification and assessment of former dumpsites, as most of them show little evidence of waste disposal or environmental contamination because the sites were usually covered by vegetation in advanced stages of succession.

In his context, as highlighted by Erb et al. (1981), aerial photographs and remote sensing images can be successfully used to inventory waste disposal sites that are presently inactive and have possibly developed other land uses. They are also useful for documenting site boundaries that may have changed through time. The quality of available information as well as the range of dates and scales determine how precisely the history of a site will be reconstructed.

Thus, the objective of the present study was to apply the visual interpretation of satellite images and aerial photographs in the historical analysis of inadequate urban solid waste disposal sites to provide essential information for their investigation and recovery processes and conditions for the elaboration of a robust conceptual site model that reflects the sites' actual environmental conditions.

2. METHODS

The present study assessed former controlled dumpsites in four municipalities in the state of São Paulo, the most populated and richest state in Brazil, which is composed of 645 municipalities. The Urban Solid Waste State Inventory carried out by its environmental agency (CETESB, 2018) shows a generation of approximately 39.890 tons of waste per day. Although 98% of this amount is disposed in adequate sites, such as landfills and composting plants, there are still more than 40 active dumpsites throughout the state.

In addition, as stressed by the environmental agency, the information presented in the inventory refers to the disposal systems currently under operation and does not include the environmental liabilities represented by former and deactivated inadequate waste disposal sites. There are no official data on the number of existing former dumpsites, but it is estimated that almost all the municipalities in the state have once had an active dump in their territories,

totaling more than 600 sites. Most of these sites have not been investigated for contamination issues nor have had an established deactivation and recovery plan.

In the present study, four former controlled dumps in the state of São Paulo were analyzed in the municipalities of Capivari, Santana de Parnaíba, Cananéia and Miracatu (Figure 1). The main characteristics of these areas are presented in Table 1. The interpretation of aerial photographs and remote sensing images were part of the contamination investigation and recovery process of these sites, which was carried out between 2014 and 2018.

The municipality of Capivari has approximately 53 000 inhabitants and is located in the east-central region of the state of São Paulo. During the 1990s, Capivari used three dumpsites for waste disposal, one of which, Anicchino was analyzed in the present study.

The utilization of Anicchino dump for municipal solid waste (MSW) disposal started in 1999 and occurred for approximately three years. This dumpsite, which was later transformed into a controlled dump, is 20.000 m^2 in area and located in an agricultural production area. Waste disposal occurred superficially following the features of the terrain along a natural thalweg with a depth of 6 m.

The municipality of Santana de Parnaíba has almost 110,000 inhabitants and is located in the metropolitan region of São Paulo municipality. From the mid-1990s to 2011, an area called Vila Esperança dump was used for irregular MSW disposal and was 65.000 m^2 in area.

Disposal occurred by the slope method, with slope heights varying between 3 m and 15 m.

The municipality of Cananéia is located in the southwestern region of the state and has a population of approximately 12.600 people. Its former controlled dumpsite, Aroeira dump, is located 20 km from the city center and received Cananéia's MSW from the end of the 1980s until 2009. The waste was disposed in man-made trenches of 2 m depth and piles of 3 m height.

The municipality of Miracatu is located in the southwestern region of São Paulo and has 20.790 inhabitants. From the end of the 1980s until 2006, the municipality used a 35.000 m^2 area for waste disposal, Miracatu dump. A natural slope of approximately 10 m was utilized because the site was limited by a mountain on its highest portion and by a river on its lowest portion.

For the historical study of the sites, the corresponding aerial photographs were taken from the database of the Laboratory of Aerophotogeography and Remote Sensing of the Department of Geography in the University of São Paulo as well as from the archives of the Institute for Technological Research; from a survey carried out in 1972 by the Brazilian Coffee Institute; and from specialized private companies. For the visual interpretation process, digital images were obtained from the scanning of the analogic aerial imagery with a 600 dpi.

False color satellite images were obtained in Google Earth® (Quickbird Pan-sharpened images, true color composition, 0,7 m resolution) and for the historical images the "time" tool was used. As the objective of the study is the evaluation of the historical evolution of dumpsites as well as the obtainment of information about irregular waste disposal, the visual interpretation of digital images was considered sufficient to understand the processes of use and management of these sites. For this reason, false color satellite images were chosen instead of original multispectral data. The characteristics of the materials used in the study are shown in Table 2.

All the images were georeferenced using the "georeferencing" tool in Esri® ArcMap™ 10.2.1.3497, applying landmarks identified along the sites as points of similarity. The visual interpretation of the selected images was made tak-

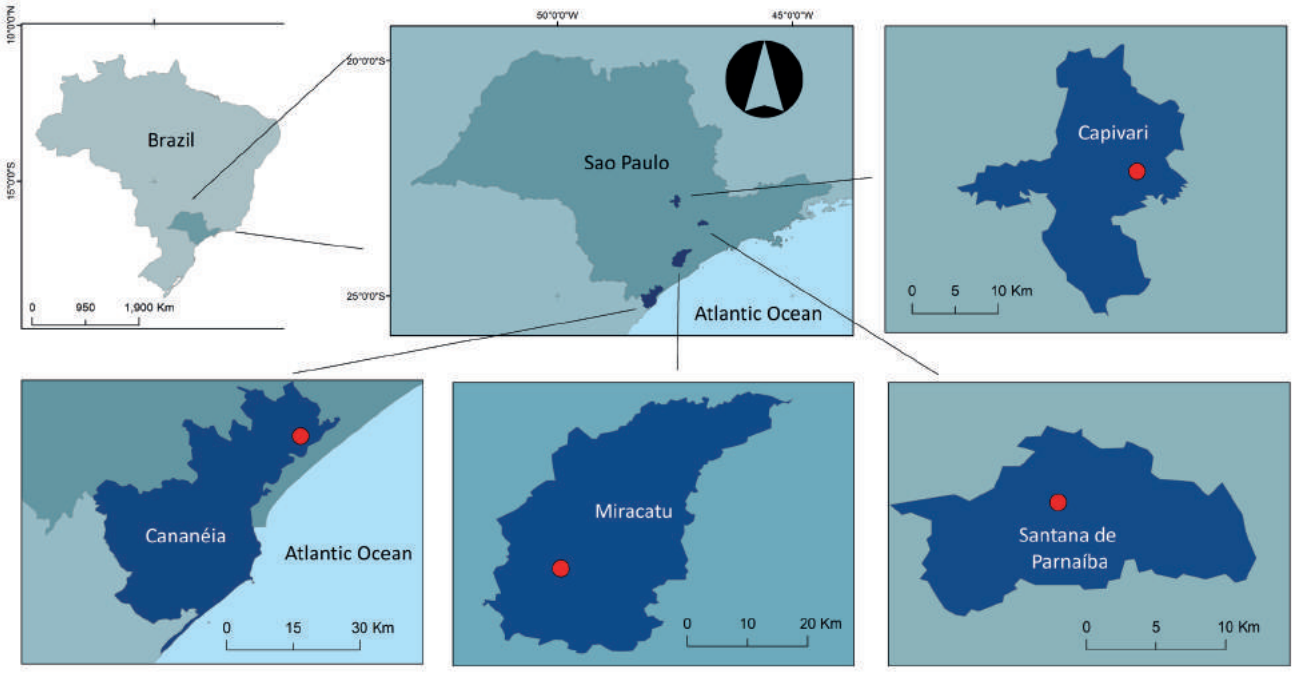

FIGURE 1: Location of the municipalities of Capivari, Santana de Parnaíba, Cananéia and Miracatu in the state of São Paulo, Brazil.

TABLE 1: Main characteristics of the former controlled dumpsites assessed in the study.

Municipality	Area	Duration of use	Waste disposal technique	Type of use of the surroundings
Capivari	20.000 m²	3 years	Slope method	Agricultural
Santana de Parnaíba	65.000 m²	15-20 years	Area method	Urban
Cananéia	50.000 m²	20 years	Combination of area and trench methods	Environmental protection area
Miracatu	35.000 m²	20 years	Slope method	Environmental protection area with agricultural activities

TABLE 2: Characteristics of the imagery used for the historical analysis of the former controlled dumpsites.

Municipality	Former controlled dumpsite	Year	Resolution (m/pixel)	Source	Original image type
Capivari	Anicchino	1972	1.65	Airborne survey (commercial provider)	Analogic
		2014	0.7	Google Earth ("time" tool)	Digital
Santana de Parnaíba	Vila Esperança	1962	1.2	Airborne survey (Brazilian Coffee Institute)	Analogic
		2002	0.7	Google Earth ("time" tool)	Digital
		2008	0.7	Google Earth ("time" tool)	Digital
		2018	0.7	Google Earth	Digital
Cananéia	Aroeira	1972	2.5	Airborne survey (commercial provider)	Analogic
		1981	1.5	Airborne survey (archives from the Institute for Technological Research)	Analogic
		2012	0.7	Google Earth ("time" tool)	Digital
Miracatu	Estrada do Teagem	1991	1.05	Airborne survey (commercial provider)	Analogic
		2006	0.7	Google Earth ("time" tool)	Digital
		2018	0.05	Drone survey	Digital

ing into consideration the following parameters: use and occupation of the sites and surroundings, vegetation cover and geomorphological characteristics. The photointerpretation process for the evaluation and understanding of the sites geomorphologies was performed by the stereoscopic analysis of aerial photographs, using a Zeiss mirror stereoscope.

Moreover, field evaluations were performed to validate the resulting interpretations, especially regarding the delimitation of the dumpsites limits, using known points (such as water bodies) and a Trimble® Differential Global Positioning System (DGPS), model Pro-XRT and OmniSTAR® correction (error of ± 0,25 cm). The opening of some inspection trenches to confirm the distribution of the waste mass was also performed.

The results were subsequently crossed with other information, such as data from geophysical studies (in the cases of Vila Esperança and Miracatu dumps), thus allowing a better understanding of site evolution and current situation.

The geophysical survey in Vila Esperança dump was performed in 2015. The frequency domain electromagnetics method was used to determine the apparent electrical conductivity of the subsoil, as anomalies could indicate the presence of waste and its byproducts in subsurface. For this, 74 pikets were installed with a spacing of approximately 40 m and georeferenced using a Trimble® DGPS.

The electrical conductivity measurements were performed in every 20 m, using the EM-34 equipment from Geonics Limited, with a space of 20 m between the transmission and reception coil, in both horizontal and vertical dipole mode. Thus, data were obtained by sampling a subsurface volume at two depths: from 0 m to 15 m (in the horizontal dipole model) and from 0 m to 30 m (in the vertical dipole mode).

The results were exported to Golden Software Surfer® 12 for interpolation using the "Griding data" tool (gridding method: kriging), generating contour maps of the electrical conductivity values measured in the field. The results were then incorporated into the site topographic plant provided by the municipality of Santana de Parnaíba.

For the generation of the contour maps, the apparent electrical conductivity was normalized using an established background, according to the following equation, in which $\sigma_{(x,y)}$ is the conductivity measure at the point and σ_{bg} is the stipulated background value:

$$db = 20 \cdot \log_{10} \sigma_{(x,y)}/\sigma_{bg} \quad (1)$$

In the case of the shallower investigation (0 m to 15 m), the measurements were normalized using a background of 15 mS/m corresponding to the natural values of the ground. For the deeper investigation (0 m a 30 m), the established background was 10 mS/m.

For the geophysical survey in Miracatu dump, the selected method was the electric resistivity, using the dipole-dipole array. The results are expressed in values of electrical resistivity of the studied ground volume. Low electrical resistivity values can indicate the presence of waste and/or its byproducts in the subsurface. The survey was conducted in 2018.

The electric resistivity method employs an artificial

source of direct or low frequency current (with intensity I) introduced into the soil by a pair of electrodes. The potential difference (V) that is established in response to this current injection can be measured by two other electrodes located nearby. Thus, the value of apparent electrical resistivity (ρ_a) of the investigated volume is measured by:

$$\rho_a = K \cdot V / I \quad (2)$$

The geometric factor K, which presents distance dimensions, is defined by the arrangement of the four electrodes. In the dipole-dipole array the space between electrodes was of 5 m and 5 depth levels were investigated. For the investigation 8 lines were defined, 4 longitudinal (L1 = 230 m; L2 = 290 m; L3 = 290 m and L4 = 300 m) and 4 transverse (T1 = 105 m; T2 = 90 m; T3 = 75 m and T4 = 70 m).

The data were interpolated in Golden Software Surfer® 12 using the "kriging" method, generating isovalues maps of apparent electrical resistivity of the surface to a depth of 5 m. The results were incorporated into the site topographic plant provided by the municipality of Miracatu.

For Miracatu dump, the 2018 image was obtained by high resolution photogrammetric survey using a drone Dji®, model Phantom 3, with a flight height of 120 m and acquisition resolution of 0.05 m. The activities comprised the high resolution photographic record with subsequent image processing, obtaining the orthophotos and digital surface model. The image processing was done in PrecisionMapper®, using 103 photos to generate an orthophoto. This allowed the visualization of the site current situation in detail.

3. RESULTS AND DISCUSSION

The historical interpretation of aerial photographs and satellite images enabled the obtainment of several types of information about the use and operation of former dumpsites. In general, four main results were identified: the delimitation of waste distribution; the identification of the waste disposal method; the evaluation of vegetation development and reconstitution; and the assessment of historical use and occupation of the surroundings. These results will be detailed in the following sections.

3.1 Anicchino dump

The historical images of Anicchino dump allowed the delimitation of the waste distribution during its period of operation. Overlapping the obtained images and evaluating the dumpsite evolution, the most significant contributions were obtained by comparing the disposal limit information obtained in the field and delimited in the 2014 image with the geomorphological interpretations based on the 1972 image (Figures 2B and 2A, respectively).

In the 1972 image (Figure 2A), it was possible to identify the delimitation of a natural ditch (yellow line), which resulted from a great erosive process, with an elevation difference between the thalweg base and the top of the terrain. At the base of the thalweg, the delimited drainage and a water spring were also identified (blue line).

Taking advantage of the geomorphological situation of the site, waste was disposed inside the thalweg, pushed to the bottom of the valley. This practice led to the filling of the thalweg, extinguishing the natural relief differences, which cannot currently be perceived at the site. In this way, the interpretations lead to the conclusion that the waste disposal process was carried out through dumping and scattering, taking advantage of the site's natural characteristics.

However, during field observations in the area delimitated for Anicchino dump, large amounts of vegetation cover that developed on the waste mass were identified (Figure

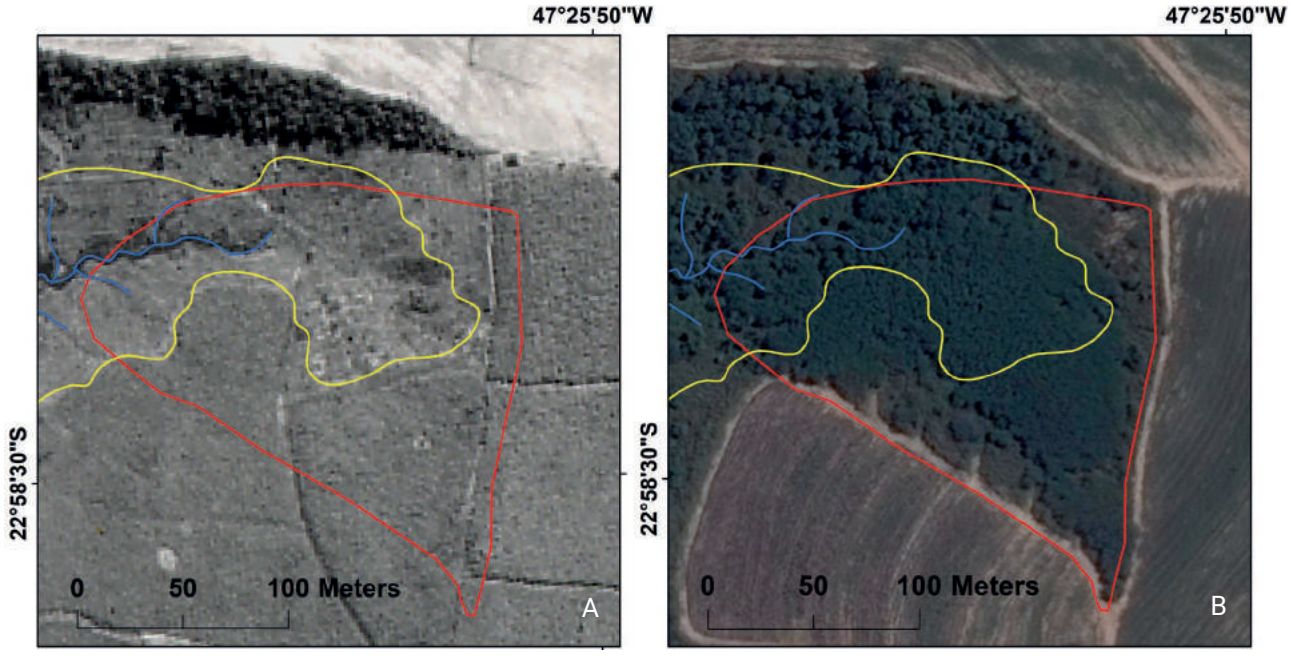

FIGURE 2: Analysis of Anicchino's 1972 aerial photograph (A) and the 2014 satellite image (B).

3). The vegetation was primarily composed of Leucaena leucocephala with trees approximately 10 m in height. The surroundings were occupied by sugarcane plantations (Saccharum ssp), which are very common in that region.

Kuehn & Hoerig (2000) classify anomalies in vegetation as indicative of the presence of anthropogenic components in a landscape, and they can be easily identified through the study of aerial photographs and remote sensing images. The authors consider vegetation as a good indicator of the presence of hazardous substances in soil, which can cause alterations in its morphological characteristics.

Regarding Anicchino dump, the presence of waste in the subsurface was indicated by the natural development of an exotic species generally used in forest recovery processes because of its high dispersion capacity. In this way, the high available organic matter content favored the growth of this leguminous species, which is capable of forming dense vegetated masses and advancing into adjacent areas (Costa & Durigan, 2010).

Considering this information, when analyzing the 2014 image (Figure 3B), updated limits of waste disposal along the site were generated (red line) in addition to the one identified in the 1972 image. In this case, as in the study developed by Erb et al. (1981), little evidence of thalweg filling can be found in the site currently or in the most recent images. Alternatively, if only the 1972 image was considered, the extension of the waste mass, as indicated by the development of a specific type of vegetation, would not be fully understood.

In this way, the sole analysis of the site's most recent satellite image as well as field inspections would not permit the real delimitation of the waste distribution, which was obtained by the interpretation of historical imagery together with the geomorphological characteristics of the site. In addition, the field inspections and analysis of more recent images permitted updating the site limits based on the correlation between the waste distribution and vegetation cover.

3.2 Vila Esperança dump

Analyzing the historical imagery between 1962 and 2018, it was possible to observe the evolution of Vila Esperança dump. In 1962 (Figure 4A), the site (perimeter defined by the red line) was not yet under operation, as it was partially covered by a secondary forest. In 2002 (Figure 4B), it is possible to observe soil movements and the presence of waste disposed in the surface soil without coverage.

In the 2008 image (Figure 4C), the excavation of the borrow pit is evident (northern portion of the site) as well as the construction of slopes and the tanks for leachate storage. It should be noted that in 2008, it is no longer possible to identify the surface waste present in the 2002 image, which was already covered in this stage of the dump operation. Both in 2002 and 2008, it is possible to observe the presence and growth of irregular occupations in the surroundings of Vila Esperança dump, which was principally carried out by waste pickers that collected recyclables and used the materials as their means of subsistence.

Finally, in 2018 (Figure 4D), approximately seven years after the end of the dump operation, the removal of the irregular occupations is observed as well as the high level of vegetation recovery on and around the waste mass. During the field inspections, it was not possible to identify clear evidence of waste disposal in the surface soil, which was

FIGURE 3: Vegetation cover identified on the waste mass.

identified in the 2002 image, and only its distribution on the slopes was evident.

Geophysical techniques are also important tools in the study of former inadequate waste disposal sites as they allow the estimation of waste distribution from the results of conductivity or resistivity anomalies. As a noninvasive technique, geophysical surveys reduce the need to open trenches to confirm waste distribution, which may affect the current stability condition of a site.

For this site, geophysical studies were carried out using the frequency domain electromagnetics method with the results obtained in electrical conductivity scales measured in the subsoil for depths of 0 to 15 m (Figure 5, left) and 15 to 30 m (Figure 5, right). The identified conductivity anomalies can be correlated with the presence of waste and/or leachate.

At depths of 0 to 15 m, the identified conductivity anomalies were restricted mainly to the slope base, where the main mass of waste was concentrated. However, at depths of 15 to 30 m, several electrical anomalies were detected outside the waste mass. Although some of them followed the water flow direction, indicating the movement of leachate through the dump, others were contrary to the flow direction, and there was no clear explanation for this. When combining the geophysical data with the topography, the dump adequation project and the satellite image of 2008,

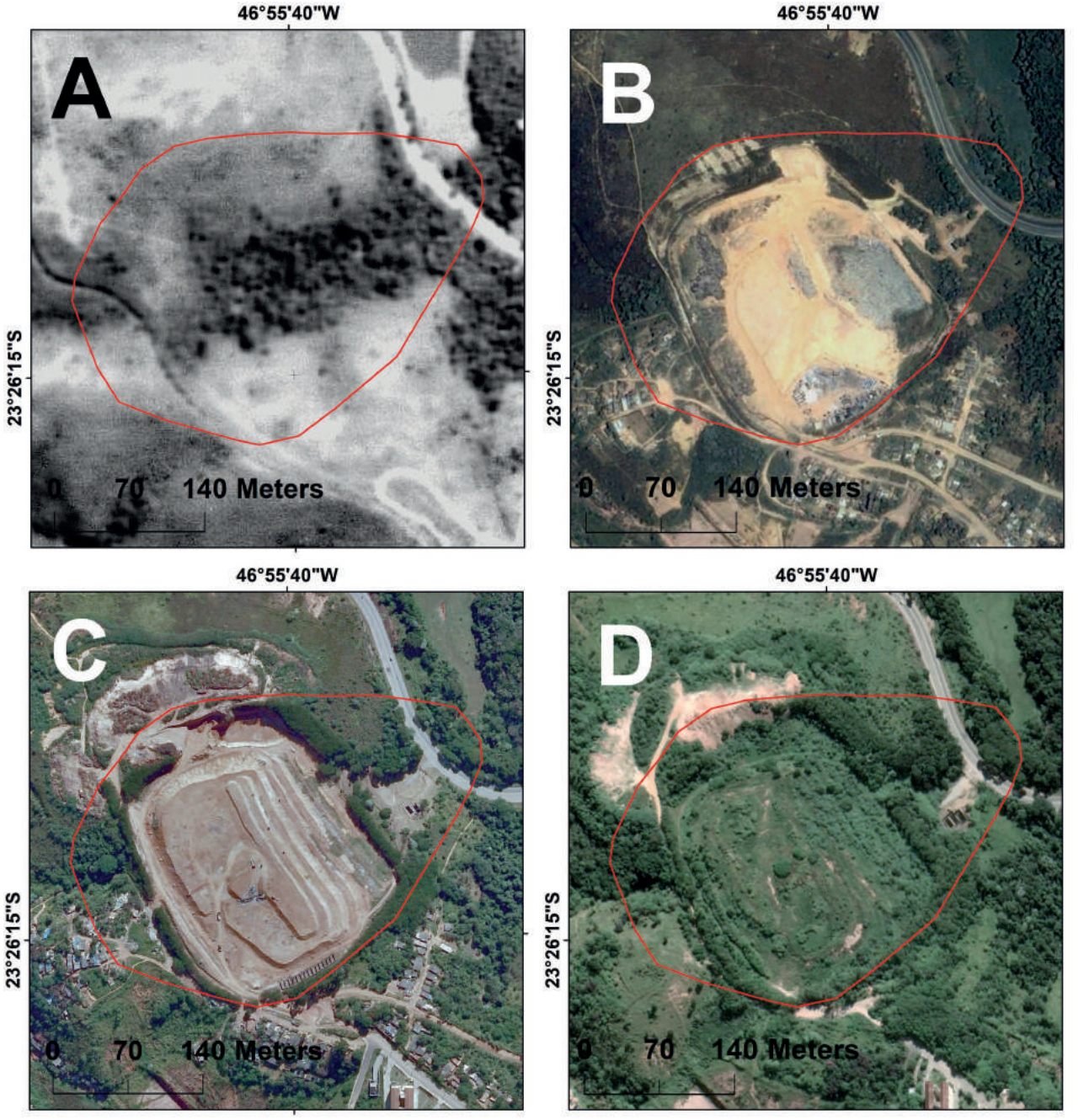

FIGURE 4: Historical analysis of Vila Esperança dump with images from 1962 (A), 2002 (B), 2008 (C) and 2018 (D).

it was possible to interpret the anomalies in a consistent way.

The main anomalies outside the principal waste mass were divided into regions (Figure 5B). In region 1 (green dotted line), it is possible to observe the overlapping of electrical anomalies with one of the former irregular occupation areas. As this area was not served by sewage collection, all the domestic sewage was released and infiltrated into the soil, altering its electrical characteristics. These discharges occurred approximately ten years ago, but their impacts could still be identified through the electric behavior of the soil.

Region 2 (orange dotted line) presents a small anomaly, which can be explained by the infiltration of leachate into the soil. As the dump is located in a water parting region, even though it is on one side of the slope, the leachate reaches a portion of the water table that flows to the opposite side, which was evidenced in the electrical results.

In the northern part of the site (region 3, the pink dotted line), it is possible to identify an area provisionally used for waste disposal that did not appear in the official dump registries, but in the satellite image of 2008, it was possible to observe the area excavation and preparation to receive the waste disposal. Later, these materials were covered with soil, and there is no record of the period or quantity of waste disposed.

Finally, region 4 (the blue dotted line) presents the behavior of leachate coming from the waste mass, which follows the distribution of the drainage systems installed in the site, showing a leakage of liquids from the drains directed to the leachate ponds.

In this case, the remote sensing images were valuable because they could be used to reconstruct the chronology of change of the waste disposal site (Lyon, 1987). Without the historical study of satellite images, it would neither be possible to correctly interpret the geophysical anomalies detected nor to identify the areas where the waste was disposed in the surface, of which there is little current evidence in the area.

3.3 Aroeira dump

During the environmental studies for closure and recovery of former controlled dumpsites, one of the necessary activities is the evaluation of the use and occupation of the surroundings, which provides information on possible influences on the area of interest.

During the interpretation of aerial photographs and satellite images, a large occurrence of vegetation suppression was identified. In the 1972 (Figure 6A) image, it is possible to verify several suppression polygons with an estimated area of 112.000 m^2. In the 1981 image (Figure 6B), a great recovery of the vegetation was observed, with a reduction in the impacted area. The calculated area in 1981 was 57 000 m^2, showing a reduction of 49% in dimension compared to 1972, which represents a vegetation regeneration rate of 6 000 m^2/year.

A portion of the impacted area was opportunistically used for the installation of Aroeira dump, whose activities began in the 1980s and ended in 2009. The site received MSW in this period as well as a large volume of fishing waste, an economic activity of great relevance for the city.

In 1984, the Cananéia-Iguape-Peruíbe Environmental Protection Area was created, in which Aroeira dump was inserted. This contributed to a reduction in anthropogenic activities and to the complete regeneration of the suppression areas. In the 2012 image (Figure 6C), the whole area is already regenerated, and it is visible only by the difference in color tonality.

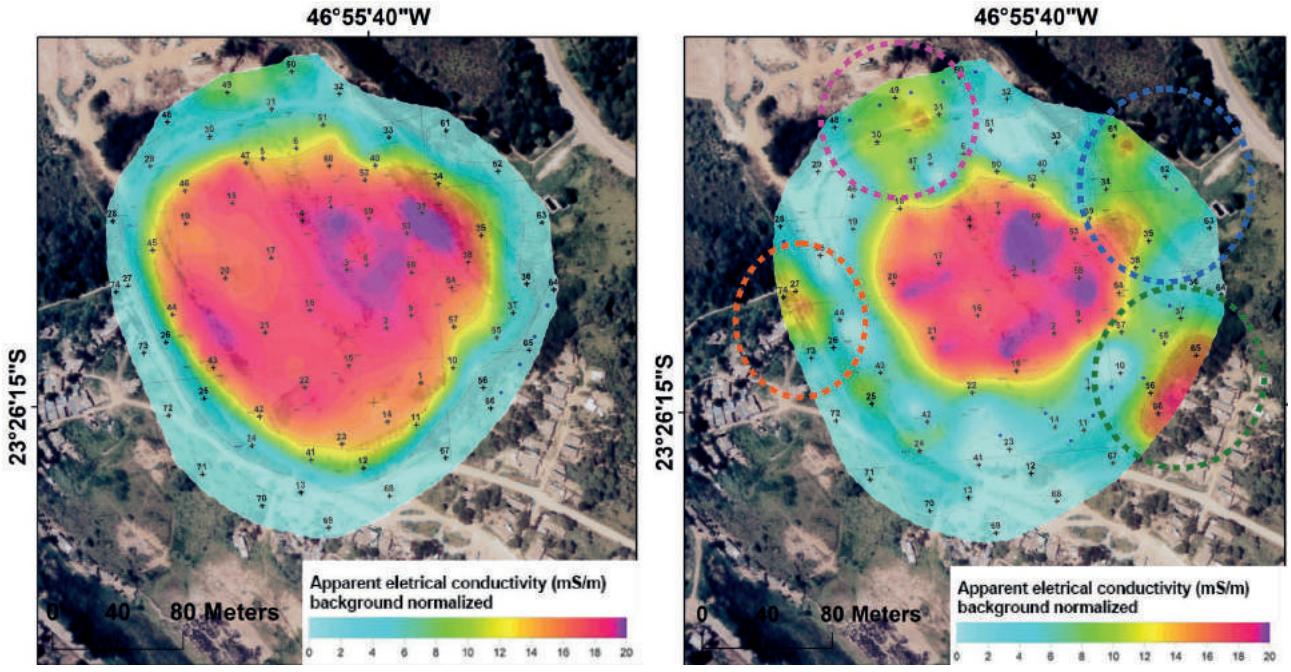

FIGURE 5: Results of the geophysical survey in Vila Esperança dump with electrical conductivity anomalies identified at depths ranging from 0 to 15 m (left) and 15 to 30 m (right)

In the historical study of Aroeira dump, the potential of vegetation resilience and regeneration in the Atlantic forest was evidenced. In less than 40 years, a 112 000 m^2 area was naturally regenerated as the sole result of anthropogenic activity restriction.

The results are also corroborated by the soil and groundwater physical-chemical parameters analysis, which did not show any value above the limits established by the environmental agencies (CETESB in Brazil and USEPA in the United States of America). This shows that even after 20 years of irregular waste disposal, natural attenuation processes were able to facilitate the environmental regeneration of the site.

3.4 Miracatu dump

During the field inspection of Miracatu dump (the pink line in Figure 7), it was observed that the waste was disposed in a natural slope formed between the road level (yellow line) and the river located in the southern portion of the site. The former controlled dump operation was carried out by depositing, compacting and covering the waste with soil, resulting in layers built from the slope base up to the level of the road, surpassing it by approximately 1 m.

Currently, it is possible to clearly identify the waste disposal in the western portion of the site, as shown in the 2006 image (Figure 7B). However, there were no registries of waste disposal in the eastern portion, as seen in the 1991 image (Figure 7A), which occurred during the initial period of Miracatu dump operation.

As a result of this lack of knowledge, in this area, agricultural activities are currently developed with the cultivation of bananas and some vegetables. In this way, the historical analysis of the imagery permitted the precise delimitation of the waste distribution in the former controlled dump, subsidizing the next stages of its environmental evaluation.

In the 2018 image, which was obtained via a drone (Figure 7C), the vegetation reconstitution of a large area in the dump surface is observed, including the development of arboreous species. Despite this fact, the presence of waste can still be evidenced by the lower exuberance and vegetation growth compared to the surroundings, especially in the areas closer to the river.

In Miracatu dump, geophysical studies were also carried out using the electrical resistivity method. The results were expressed in electrical resistivity scales in the subsoil (Figure 8). The areas with a lower resistivity and, consequently, a higher electrical conductivity were associated

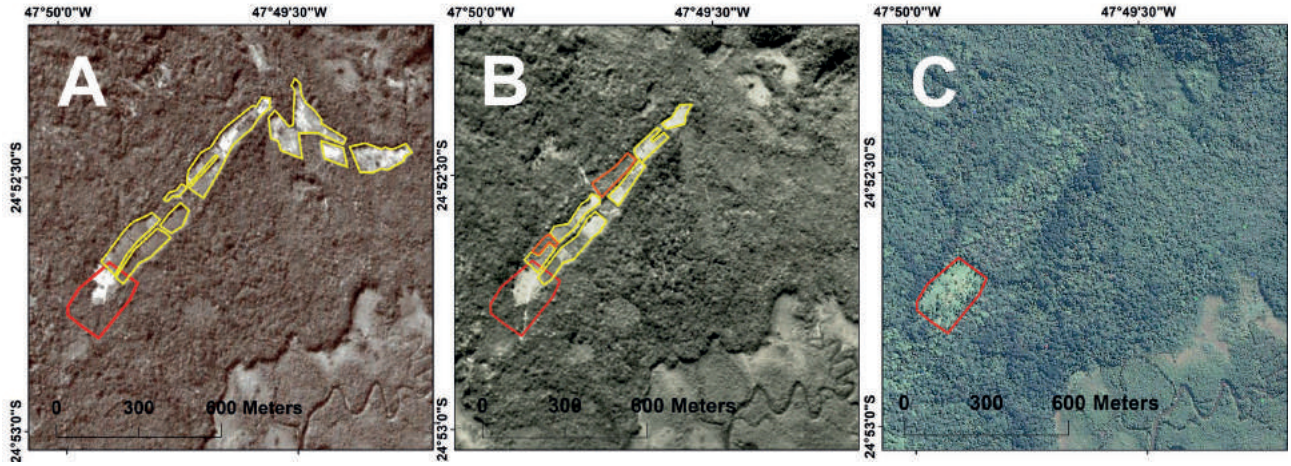

FIGURE 6: Analysis of vegetation suppression and reconstitution in the surroundings of Aroeira dump in 1972 (A), 1981 (B) and 2012 (C).

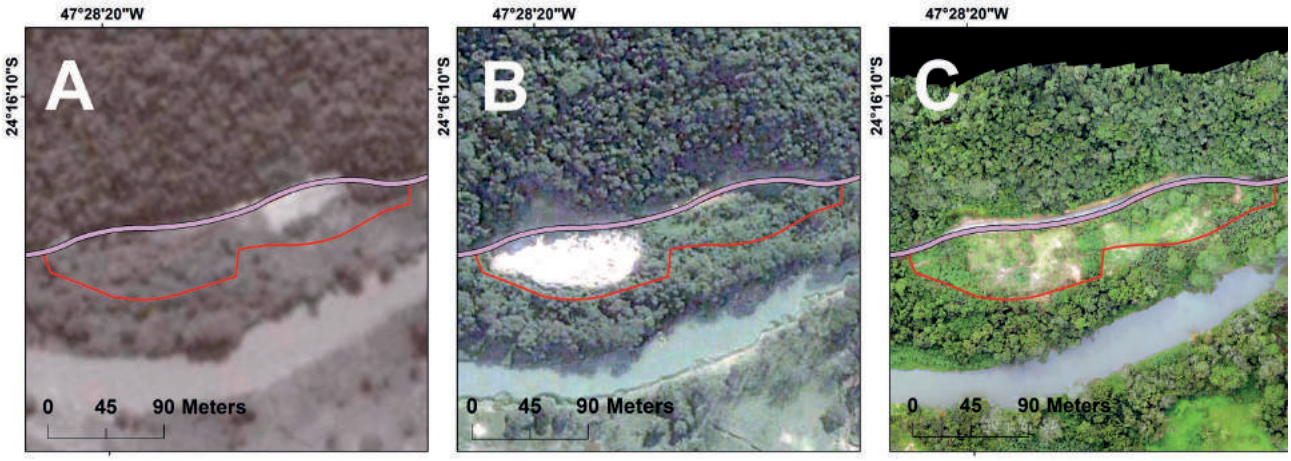

FIGURE 7: Historical analysis of Miracatu dump in 1991 (A), 2006 (B) and 2018 (C).

with the presence of waste and/or leachate. The results indicated a high correlation between the waste disposal areas delimited by the historical evaluation of aerial photographs and satellite images and the low resistivity zones identified at a depth of 4 to 5 m in the subsurface.

Therefore, as in the case of Vila Esperança dump, the comparison of the information obtained in the imagery study with the geophysical results allowed a more detailed and precise delimitation of the waste distribution throughout the former controlled dump. This is essential for planning the next steps of environmental assessment, such as setting the distribution of monitoring wells for the evaluation of soils and groundwater quality, thus optimizing and subsidizing the former dumps' recovery processes.

4. CONCLUSIONS

The visual interpretation of aerial photographs and satellite images for identifying and assessing the evolution of inadequate waste disposal sites has been shown as an excellent tool, principally in areas with a lack of past records. The application allows for retrieving the operation histories of former dumpsites and helps create an adequate investigation model that suits each particular situation.

Information from different databases, which has been generated in recent studies, can be crossed with previous databases, thus allowing the obtainment of greater data structure for a better interpretation of the available information. For the obtainment of current and historical imagery, this paper sought to suggest a methodology that can be widely applied in terms of database availability, with the use of high quality materials from free and open sources. This could be particularly interesting for municipalities that lack the technical and financial resources for investigating inadequate waste disposal sites.

In addition, field inspections and activities are also essential during this process, as good data interpretation also depends on the knowledge of a site and its particularities, allowing, in this way, the obtainment of robust results and contributing to the advancement of dumpsites recovery processes.

ACKNOWLEDGMENTS

The authors acknowledge the Municipalities Technological Support Program (PATEM) of the Government of São Paulo State as well as all the IPT team working in the investigation and recovery of former dumpsites in the state of São Paulo.

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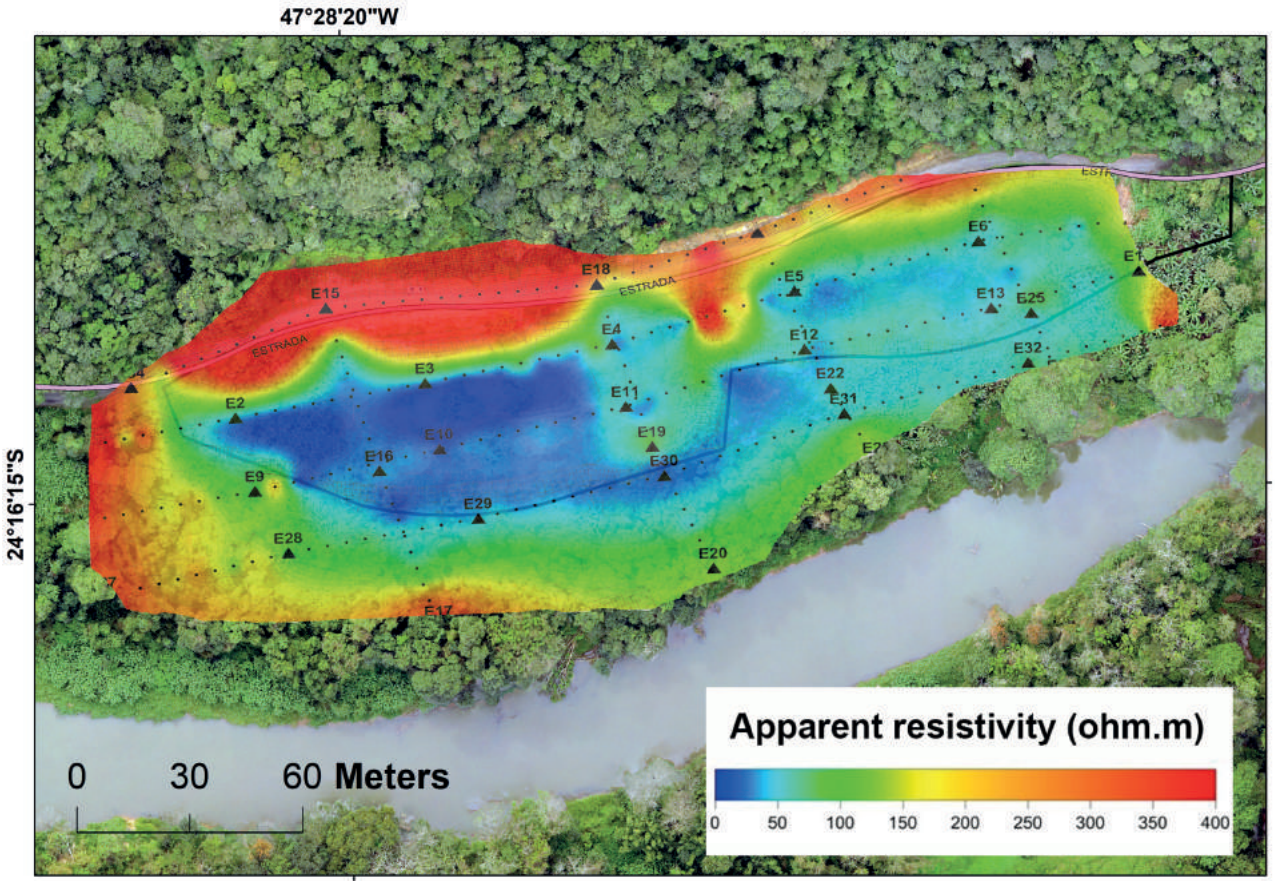

FIGURE 8: Geophysical results in Miracatu dump expressed in scales of electrical resistivity in the subsoil.

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A REVIEW OF THE APPLICATION OF GIS IN BIOMASS AND SOLID WASTE SUPPLY CHAIN OPTIMIZATION: GAPS AND OPPORTUNITIES FOR DEVELOPING NATIONS

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Article Info:
Received:
4 January 2019
Revised:
20 May 2019
Accepted:
3 June 2019
Available online:
28 June 2019

Keywords:
Waste
Bioenergy
Supply chain
GIS

ABSTRACT

The application of Geographical Information Systems (GIS) enhanced modelling techniques in biomass and solid waste supply chain problems is hinged on a common denominator for both systems: the spatial distribution of supply points and variability of resource quantities. Since the sustainability of bioenergy or waste-to-energy projects around these resources will be affected significantly by the cost of supplying them, it is important to optimize decisions around facility location, size and transport routes. GIS is an important tool that can be used to capture the spatial and temporal dynamics of the biomass and waste. It can then be used alone or integrated with other software tools, for strategic and tactical level optimization of biomass and solid waste supply chains. In as much as a lot of progress has been made globally in research and application of GIS enhanced modelling techniques in biomass and solid waste supply chains, developing nations have trailed behind. This explains why spatial and temporal waste or biomass statistics are not readily available in these areas. This paper reviews recent developments in the application of GIS in biomass and solid waste supply chain models, with the ultimate objective of identifying the gaps and opportunities that exist. It is especially biased towards the use of the biomass and waste in renewable or waste to energy schemes- a fast growing field within the green economy.

1. INTRODUCTION

1.1 General background: Waste to energy and bioenergy systems

Waste to energy (WtE) and biomass to bioenergy (BtB) are both significant highlights within global green economy schemes, representing the use of 'renewable' waste and biomass (Kennes, et al., 2016; Vlachos et al., 2008). Recent green initiatives are hinged on the fact that these two resources can be an invaluable substitution for fossil based fuels both in the power and fuels industries since both can be converted into fuels, heat and power using various technologies (Batidzirai et al., 2012; Nkosi & Muzenda, 2014; Pantaleo & Shah, 2013; Pilusa & Muzenda, 2014; Sobrino et al., 2011). Biomass can be thermally or biochemically converted into renewable biofuels, while selected fractions of Municipal Solid Waste (MSW) like tyres, rubber and plastics can also be thermo chemically converted into heavy oils and fuels (Pilusa & Muzenda, 2014; Pradhan & Mbohwa, 2014). Due to the rising awareness and advocacy for a green economy, both fields have registered a significant growth in the past decade. The global Waste to Energy (WtE) market was valued at US$25.32 billion in 2013, having grown by 5.5% from 2012. It has then been projected to grow by a Compound Annual Growth Rate (CAGR) of over 5.5% from 2016, reaching a value of US$40 billion by 2023 (World Energy Council, 2016). The BtB industry is also growing with the follow segmented CAGR projections: 44% for advanced biofuels from 2017-2021; 9.6% for all biofuels (2013-2019); 7% and 8.1% for biomass power generation and biodiesel respectively in 2018 (Sapp, 2014a, 2014b, 2017). The main, common drivers for both are the global lookout to increase Renewable energy sources (RES), rising environmental consciousness, the advent of circular economies, government policies and support through grants, tax credits, incentives and special loans (Sapp, 2017; World Energy Council, 2016).

A significant fraction of the biomass available for energy exploitation is essentially waste- especially agricultural and forestry residues- which then form an intersection with solid waste (SW) (Iakovou et al., 2010). In such WtE ventures, the green economy value is double pronged: comprising

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Detritus / Volume 06 - 2019 / pages 96-106
https://doi.org/10.31025/2611-4135/2019.13819

ameliorating the environmental problem of the waste and deriving useful energy products from it (Pilusa & Muzenda, 2014). According to Gasparatos et al. (2015), the weight of each of these value propositions varies from developing to developed nations; with the latter according a significant weight to climatic and environmental issues, while the former are more interested in socio-economics (Gasparatos et al., 2015). According to Maslow's hierarchy, this is perfectly normal, since developing nations have to meet pressing subsistence needs before thriving for safety and environmental issues (Yawson et al., 2009). Indeed, the potential socio-economic benefits for such WtE or bioenergy ventures can be significant, spanning increased economic development (more income and tax revenues), employment creation, increased national energy security, alternative & cleaner fuels and alleviation of energy poverty in remote/rural communities (Ji & Long, 2016). In light of such potential benefits, optimized management and utilization of MSW and biomass could help developing nations derive more value from these abundant resources in them, tackling both socio-economic and ecological issues in significant ways.

1.2 WtE and BtB supply chain dynamics

Due to the spatial distribution of supply points and variability of quantities for both biomass and SW resources, one of the critical decisions to be made would be site locations and optimal transportation routes (Chalkias & Lasaridi, 2009; Kinoshita et al., 2009; Shi et al., 2008). Beyond the common applications for both resources in renewable energy, an interesting fact to note is the similarity of the supply chain systems around the two feed stocks. A number of authors have concurred that the two major constraints that hamper widespread uptake and dissemination of WtE and bioenergy projects are cost (a function of technical complexities, especially in the conversion technology) and the feedstock supply chain (SC) dynamics (Amundson et al., 2015; Batidzirai et al., 2012; Iakovou et al., 2010; Vlachos et al., 2008). Even though the feed stocks can be cheap, as in the case of MSW, agriculture and forest residues, the total cost for the feedstock supply significantly contributes towards high production costs; ranging from 40-70% (IRENA, 2016; Ji & Long, 2016). This is due to the low energy density of biomass and MSW, the spatial distribution of supply points and variability of resource quantities at those points compared to fossil fuels (Amundson et al., 2015; Iakovou et al., 2010). The sustainability of bioenergy or WtE projects around these resources is therefore affected significantly by the cost of supplying them, making the optimization of supply chain factors like facility location, size and transport routes important considerations (Iakovou et al., 2010). In waste management, the WtE facility can be a direct replacement of landfills, where demographic patterns begin to influence the location of the site, or can be based on the landfills available.

1.3 Methodology

This study examines the growing opportunity for GIS application in SW and biomass WtE supply chains for developing regions, keeping the trends in developed regions in perspective. Africa presents itself as an interesting focus case for developing nations by combining high population growth and MSW generation with slow technology (GIS) uptake and low biomass waste utilization (Abarca et el., 2013; Nwosu & Pepple, 2016; Pradhan & Mbohwa, 2014). The specific countries in Africa were then picked based on availability of research reports around GIS applications for bio-energy or MSW waste and they should categorized among the developing countries (Fantom & Serajuddin, 2016). A couple of successful applications from developed nations were also picked in order to draw out a parallels compared to developing regions. Using this criteria, a total of eight studies (four for each case) were carried out in detail. References are made however, briefly to other relevant studies.

The main method used in obtaining data was a desktop survey using google search engines and mainly targeting peer reviewed scholarly articles. To stream line the search, phrases containing 'GIS' along with 'biomass', 'bio-energy' or 'MSW waste' were formulated. The study however, excludes solid sewage waste from both biomass and MSW wastes. The review timeframe is from 2001 to 2018, capturing both historical and state-of-the art trends in the application of GIS.

2. GLOBAL VIEW OF THE APPLICATION OF SC OPTIMIZATION IN BIOMASS AND WTE VENTURES

The complexities associated with the design and planning of bioenergy and WtE SCs have emanated from the associated high costs of handling per unit energy, seasonal and uncertain nature of feedstock supplies, variability of feedstock locations and other factors (Iakovou et al., 2010). These and other reasons have made it imperative to optimize these SCs, with various objectives such as maximizing conversion throughput, maximizing social returns like employment, minimizing GHG emissions and minimizing costs. Despite an equally compelling case for research around feedstock supply chain dynamics and costs, most research has focused on the conversion technologies (Paolucci et al., 2016). There has, however, been a recent upsurge in research around bioenergy and WtE SCs, though the initial focus was the assessment of potential resource volumes, allocation of collection sites and location of production facilities (Iakovou et al., 2010). In MSW management particularly, the initial focus was transport routes for waste and location of landfills rather than energy conversion facilities (Nwosu & Pepple, 2016). However, SC optimization has increasingly been covering a broader scope owing to recent advances in computational tools, subsequent improvements in mathematical models and the realization recent awakening to SC logistic issues as a major bottleneck in most bioenergy and WtE projects, (Ba et al., 2016; Hadidi & Omer, 2017; Pantaleo & Shah, 2013). Still, more research is required to ascertain the viability of bioenergy and WtE projects through SC optimization. Such research outputs could contribute to a significant reduction in the cost of the integrated bioenergy system (Gold & Seuring, 2011; Hombach et al., 2016; Iakovou et al., 2010).

SC optimization literature has generally concurred that

supply chain complexities have to be addressed at 3 decision levels: strategic, tactical and operational (De Meyer et al., 2014; Iakovou et al., 2010), (Awudu & Zhang, 2012). These are defined in Table 1, along with the activities normally tagged along these levels.

A number of studies have looked into SC optimization at the different levels demonstrated in table 1. Most of the researches take a Multicriteria decision analysis (MCDA) approach based on many hierarchical attributes or objectives, often conflicting, which are analyzed mathematically to obtain an optimal choice (De Meyer et al., 2014).

In principle, the entire supply chain comprises the production, harvesting or collection of biomass or MSW; transportation; pre-treatment; storage; subsequent conversion to bioenergy (heat, power or fuels) and supply to markets (Ba et al., 2016). Consequently, the other important factor in the SC optimization studies is the part of the supply chain they focus on, as shown in Figure 1. The upstream process includes the generation, pre-treatment and delivery of MSW or biomass in the appropriate form to the conversion facility. The midstream SC covers the bioenergy or WtE conversion facility, while the downstream SC concerns the supply and distribution of the bio-product (heat, power or fuels) to the market (Ba et al., 2016).

Though there has been a push towards integrated SC optimization models that span all the stages and variables within the whole chain to maximize or minimize certain objectives, such models could be complex, requiring a multi-disciplinary approach (Amundson et al., 2015; Nogueira et al., 2017). The conventional practice therefore, has been to separately optimize the upstream and midstream parts, since they jointly represent the largest fraction of costs incurred in the whole SC (Amundson et al., 2015; Batidzirai et al., 2012; Iakovou et al., 2010; Vlachos et al., 2008). The upstream SC optimization is mostly an operations research problem, while the midstream is largely process engineering and associated unit operations.

3. THE ROLE OF GIS MODELLING IN SC OPTIMIZATION

3.1 GIS and its functionalities in the context of SW and biomass supply chains

Recent technological advances in computational tools have presented GIS as an innovative and versatile tool in both SW management and biomass SCs (Hadidi & Omer, 2017; Sufiyan et al., 2015). Consequently, there has been a significant increase in the use of desktop GIS in the last few decades, encouraged by the expansion of PC capabilities and reduction in cost of using them. GIS software vendors have, since, been redesigning their packages to conform to global trends and demands- one such being the green economy (Nwosu & Pepple, 2016).

GIS is a sophisticated modern technology used for capturing, storing, displaying, analyzing and manipulating spatial data (Chalkias & Lasaridi, 2011). One key advantage of the platform is that it can combine the spatial datasets with non-spatial quantitative or qualitative data including quality and quantity of the resource, vector and raster data from satellite imagery, digital elevation model data, topographic data and operational environment. The data is then arranged into thematic layers represented by digital maps (Chalkias & Lasaridi, 2009). Quinta-Nova et al. (2017) applaud GIS's embedded capability to provide a Multi Criteria Decision Analysis (MCDA) support based on spatial criteria (Quinta-Nova et al., 2017). In this case, a set of environmental, economic and social criteria is defined, ranked and weighted, either using some logic system or the Analytic Hierarchy Process. (Quinta-Nova et al., 2017) The GIS can then select optimal sites for conversion or landfill sites using the ranked suitability criteria. For both biomass and SW management, objectives usually include minimizing distance, cost of transporting waste or biomass and GHG emissions, while other site related criteria like topography and legal requirements would also need to be factored in (Chalkias & Lasaridi, 2011; Eason & Cremaschi, 2014; You, Graziano, & Snyder, 2012). In some cases, only socio-economic objectives are incorporated to obtain a wide array of potential sites, then some are eliminated based on other logical and legal criteria. Essentially, a GIS model not only acts as a digital data bank for spatial characteristics (e.g. quantities) of waste or biomass, but can manipulate that data at reduced time and cost to give best location and alternatives for processing or storage facility (Sufiyan et al., 2015). Figure 2 exemplifies a stage wise approach to a GIS optimal site selection problem.

In a number of cases, the GIS is integrated with simulation or optimization tools and can either be embedded in the overall program or be a precursor to predetermine the best candidate sites for subsequent SC optimization

TABLE 1: SC decision levels (Charis, Danha, & Muzenda, 2018).

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

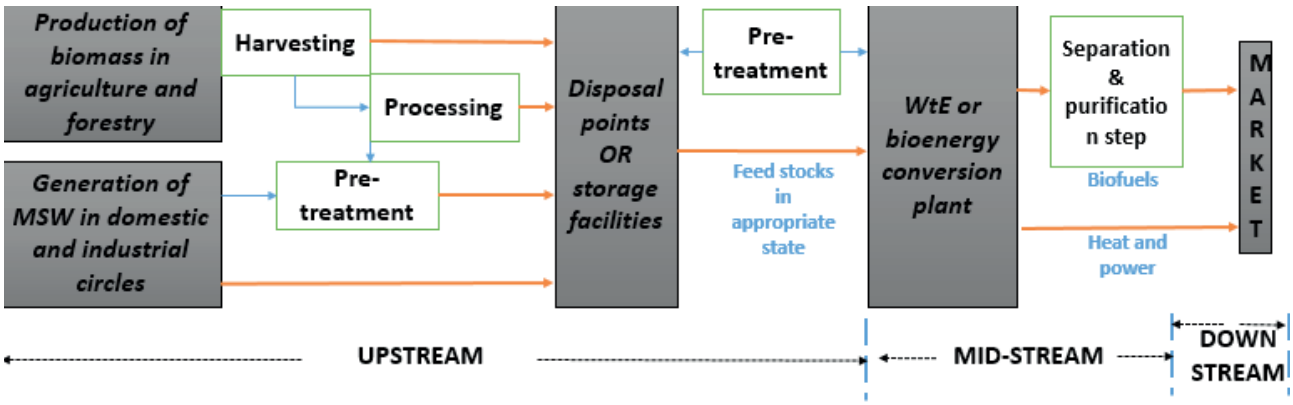

FIGURE 1: WtE and biomass supply chains. Colour filled blocks represent major operation nodes while unfilled blocks represent minor operations. Arrows denote possible transport links.

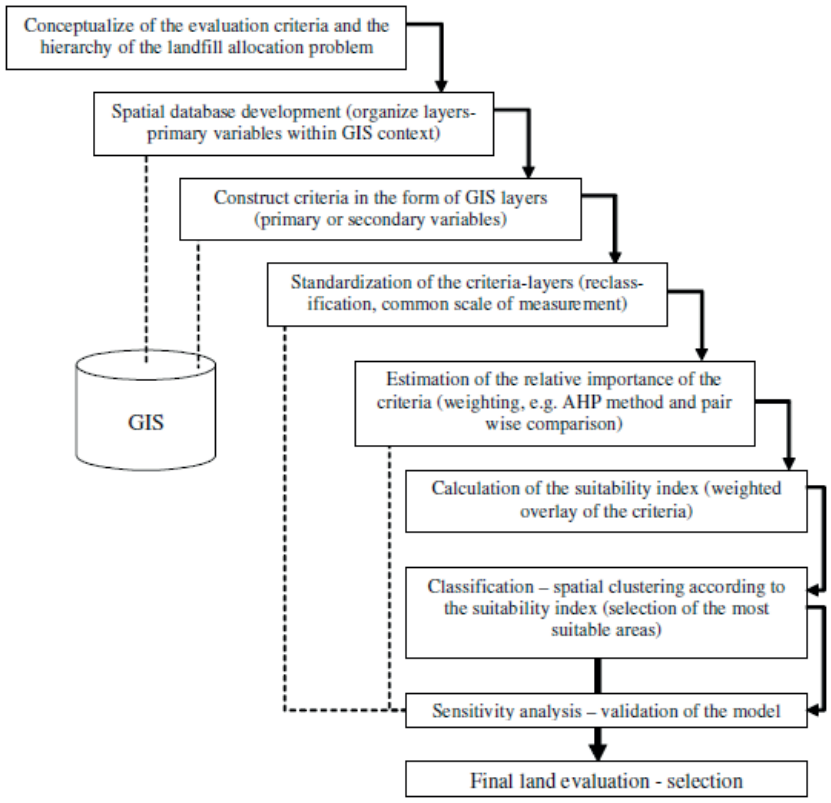

FIGURE 2: Application of GIS for site selection for landfill (also applicable to WtE and bioenergy facilities) (Chalkias & Lasaridi, 2011).

(He-Lambert et al., 2018; Tan et al., 2014; Woo et al., 2018; Zhang et al., 2016).

A review of the use of GIS in routing optimization shows that this application is more prevalent in SW collection and transport problems rather than biomass SCs (Ahmed, 2006). This is due to the weight placed on collection and transport in SW management as a tactical and operational problem; whereas the bioenergy system (most which are at planning stage), have a bias towards the strategic problem of facility sizing and conversion site selection (Ba et al., 2016; Prins et al., 2015; Shi et al., 2008). Moreover, biomass sites are often in remote spaces where routes are not so many, defeating the purpose of 'route optimization'. With the advent of the green economy, it is also likely that SW management problems will gravitate from the conventional landfill site selection and routing problems to also cover supply chain optimization for WtE plants.

Another interesting point is that the both site and route optimization problems can use tools like ArcGIS, Google Earth, Geographical Positioning Systems (GPS) and Google map for collection of spatial data (Ahmed et al., 2016; Sufiyan et al., 2015). For routing optimization like waste transport and collection it is more imperative to then use ArcGIS Network Analyst or a similar tool like GIS router to analyze and optimize the optimum route. In this case, there is need to supply the road network spatial data for the

study area (Chalkias & Lasaridi, 2009). Chalkias & Lasaridi (2009) explain that Network Analyst is an improved optimal path finding algorithm from the classic Dijkstra's algorithm 'which solves the problem of optimal route selection on an undirected, non-negative weighted graph in a reasonable computational time' (Chalkias & Lasaridi, 2009). They present the data flow diagram of their methodology (Figure 3).

3.2 Findings on GIS Applications in MSW and biomass SC optimization

3.2.1 MSW management

Chalkias & Lasaridi (2011) presented a short literature review of two common optimization problems in SW management: Landfill/dumping site optimal location and route optimization. They highlighted that optimal site location of a landfill (applies to a bioenergy or WtE site) is complex, requiring consideration of various technical, environmental, legal and socio-economic constraints (Chalkias & Lasaridi, 2011).Tan et al. (2014) reiterated the increasing complexity and cost of MSW management, especially given the rapid socio-economic development and increased volumes of waste (Tan et al., 2014). Nwosu and Pepple (2016) added that the involvement of so many parameters make the empirical process of selecting such sites complicated, costly and time consuming (Nwosu & Pepple, 2016). The weight of factors to be considered for bioenergy & WtE sites could be similar, however, they may contrast with traditional landfill sites since the latter span more environmental and socio-economic constraints. In their review, Chalkias & Lasaridi (2011) highlighted that in the landfill site evaluation problems in the last few years have used combinations of GIS with fuzzy systems, multicriteria decision analysis (also embedded within GIS suite), analytic hierarchy process and factor spatial analysis, among other integrations (Chalkias & Lasaridi, 2011). Such a flexibility of GIS for integrations, enabling comprehensive spatial analyses, is a major advantage of GIS. Recent GIS applications for both bioenergy/WtE and landfill selections, however, use a 'suitability index' to rank the most suitable sites, rather than binary outputs that would result from the above integrations (Celli et al., 2008; Chalkias & Lasaridi, 2011; Panichelli & Gnansounou, 2008; Voivontas et al., 2001).

Nwosu and Pepple (2016) looked into site selection criteria that meets stipulated standards for dumping sites that includes socio-economics, physical characteristics, and land-use factors in Nigeria (Nwosu & Pepple, 2016). They initially built a spatial database using datasets including road network, topography & geology, GPS co-ordinates for current solid waste dumpsites, land use, water bodies and soil profile of study area. Ultimately, they carried out a spatial analysis using ArcGIS Network Analyst, spanning slope, Euclidean distance, reclassification and weighted overlay analysis. They used the Suitability Analysis Model Builder to identify optimal dumping sites (Nwosu & Pepple, 2016).

Sufiyan et al. (2015) developed a GIS database to monitor trends towards generation and disposal of waste, including preferred dump sites in Nigeria (Sufiyan et al., 2015). The database was meant to inform planning processes in collecting such wastes to reduce aesthetic pollution and curb potential environmental health & pollution problems associated with disposal and burning of the waste. Sufiyan et al. (2015) and Nwosu & Pepple (2016) argued that SW waste accumulation in undesignated places is an acute problem in developing countries due to continued urbanization and the associated increase in consumption and production patterns (Nwosu & Pepple, 2016; Sufiyan et al., 2015). Moreover, the proportion of MSW that has to be disposed is higher in these developing countries due to low recycling and reuse capabilities (Nhubu et al., 2017; Nwosu & Pepple, 2016). As a result, local authorities have not been able to keep up with the disposal of such huge waste volumes, especially in densely populated areas (Ahmed, 2006). Sufiyan et al. (2015) then recommended the use of GIS to determine the spatial & temporal quantities of major illegal dumpsites dotted around such areas them to help in planning, prioritization and mobilizing private and public partnerships in the collection of the SW (Sufiyan et al., 2015).

Tan et al. (2014) synthesized a model that 'preferential-

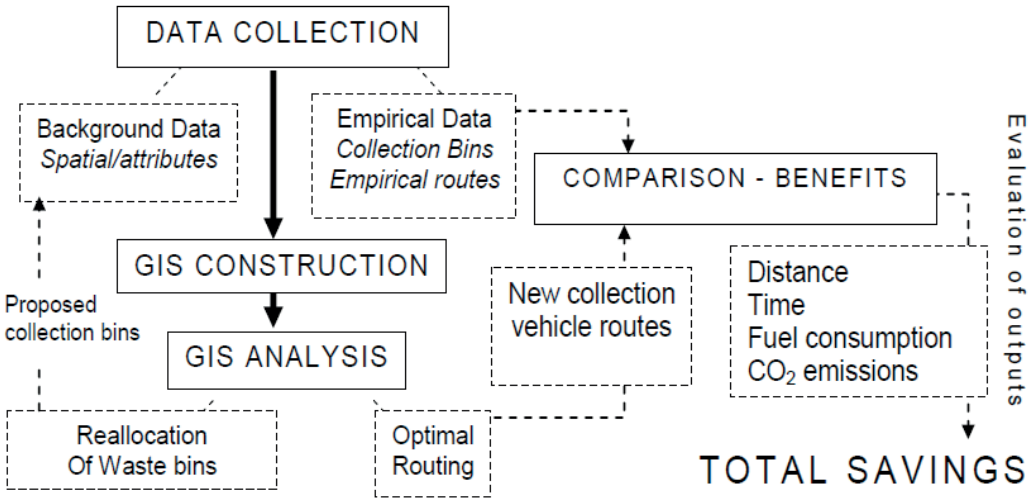

FIGURE 3: Methodology for GIS model for use in optimal route selection (Chalkias & Lasaridi, 2009).

ly utilizes the waste to produce energy to meet the targeted demand with the best mix of WtE technology, types of waste, power plant capacity, location and annual planning of WtE power plant construction' for a Malaysian region in the years 2012-2025 (Tan et al., 2014). They integrated the GIS with a Mathematical model with an overall objective of minimizing the total cost of the WtE strategy. The GIS tool caterered for the location selection for the WtE facility which could either be a combined heat and power (CHP) plant or a Landfill gas (LFG) recovery plant. The suitability criteria comprised maximum driving distance for dump trucks, minimum allowable distances from residential areas, proximity to customers and elevation above sea level. The mathematical model then factored in the technology selection and plant capacity, using cost factors supplied by literature for various technologies and plant sizes (Tan et al., 2014). The objective function for this module was to reduce the cost of producing electricity given constraints of feedstock resources availability, capacity demand, construction lead time and location.

Chalkias & Lasaridi (2009) looked at a route optimization challenge, mainly from a developed nation (Greece) viewpoint (Chalkias & Lasaridi, 2009). They asserted that the sustainable SW waste management paradigm as espoused by the EU waste policy, which requires source separation to recover materials and energy, will require more frugal waste management practices by local authorities (LAs) (Chalkias & Lasaridi, 2009). This is imperative since spatially distributed waste streams like construction and demolition waste, packaging waste, used tyres, biodegradables, electrical and electronic waste have target fractions set for recycling, recovery and landfills. Their research therefore identified GIS as the choice tool for analyzing such a complex spatial problem, where routing optimization can minimize costs. They commented that, although waste sorting is not yet a focus area in developing nations, routing optimization can still deliver value owing to the dense populations and prevalence of open site dumping. The authors then built a model combining spatial/geographical data (road network, location of waste bins, land uses etc.) and non-spatial data; both obtained from analogue maps, on-site data using GPS and digital data from Statistical offices. Chalkias & Lasaridi (2011) obtained an optimal route and bin reallocation model that offers savings in time (3-17%) and distance (5.5-12.5%) compared to the existing route (Chalkias & Lasaridi, 2011).

3.2.2 Biomass to bioenergy (BtB) supply chains

Panichelli and Gnansounou (2008) asserted that the profitability of BtB systems is highly geographically dependent since upstream biomass SC accounts for a significant fraction of total bioenergy costs (Panichelli & Gnansounou, 2008). They pointed out that the key objective is then, how to obtain sufficient biomass quantities above the minimum economic throughput of the bioenergy plant. A number of researchers in biomass SCs have therefore resorted to the use of GIS enhanced tools for optimal facility location at the strategic level and for route optimization at the tactical and operations level (Kinoshita et al., 2009; Voivontas et al., 2001; Zhan et al., 2005).

He-Lambert et al. (2018) combined GIS with a Mixed Integer Linear Programming (MILP) model in a two-stage approach to identify feedstock supply, pre-treatment facilities, and biorefinery locations with high spatial resolution scale to meet the annual biofuel production and demand goal for Tennessee, USA (He-Lambert et al., 2018). The first stage employs the GIS and determines the bio-refinery and feedstock while the second optimizes choice of harvesting options and the location of pre-treatment facilities. They highlighted that the advantage of using GIS only is that one can determine production potential and distribution of available feedstock, optimal biorefinery locations and market distribution routes for the biofuel with no explicit objective functions or resource constraints. There would be limitations however in terms of model replicability, transferability and room to carry out economic analyses and simulations for alternative routes. An integration therefore brings in the GIS advantages and eliminates most limitations (He-Lambert et al., 2018).

Woo et al. (2018) combined GIS with MCA and include a supply chain cost analysis for Tasmania, located in Australia. They argued that a comprehensive SC design that will determine the optimal number, size and location of bioenergy facilities should factor in both economic (especially transport), environmental and socio factors as depicted in Figure 4.

In another integration case, Zhang et al. (2016) combined GIS with simulation and optimization tools where GIS was a precursor to select candidate biofuel facility locations using factors like accessibility to biomass, railway/road transport network, labour availability and proximity to water bodies. The resulting candidates were then used as inputs for the simulations and optimization tools where the former would then be used to track flows within a given SC network, while the latter determines the optimal SC network in terms of various costs (Zhang et al., 2016).

Koikai (2008) used GIS in siting analysis to identify potential locations for bioethanol processing plants using first-generation feed stocks in a Kenyan province (Koikai, 2008). The author first defined and logically ranked the suitability factors for the plants, including proximity to maize farms; access to major highways/roads or railways and access to utilities like water and electricity. Using acquired geo-referenced data, the author then produced vector maps representing suitability profiles for various sites according to each of the suitability factors. All the vector data was then converted to raster, reclassified then compared for suitability analysis using ArcGIS Spatial Analyst. The result was a map of several potential biofuels processing sites in several towns, which could be used by relevant stakeholders in Kenya, considering other factors (Koikai, 2008).

Kinoshita et al. (2009) came up with a GIS database for a spatial evaluation of forest biomass usage. The database model would reveal usage patterns and serve as an information repository for future decisions (Kinoshita et al., 2009). Kanzial et al. (2009) integrated GIS and Multi Integer Linear Programming to model optimal material flows and subsequent plant production costs for different demand scenarios and supply options. They also demonstrated the differences between direct flow and flow via storage

(Kanzian et al., 2009). Panichelli and Gnansounou (2008) developed a methodology that integrates a GIS system with a biomass allocation algorithm to select suitable bioenergy facilities (Panichelli & Gnansounou, 2008). Their model appealed as different from most facility site location problems since it considered a scenario where these sites could compete for the scarce biomass resource. Papadopoulos and Katsigiannis (2002) developed a GIS tool to locate a conversion facility considering economic sustainability (Papadopoulos & Katsigiannis, 2002).

4. OPPORTUNITIES FOR DEVELOPING NATIONS: CASE OF AFRICA

Figure 5 illustrates that the uptake of GIS technologies is still very low for developing regions like South America and Africa. It also shows that the biggest end user is the government, mostly for demographic purposes, followed by the natural resources field. Moreover, the largest leap in market share by 2025 is also reflected by natural resources, where biomass occupies a very significant role. Since

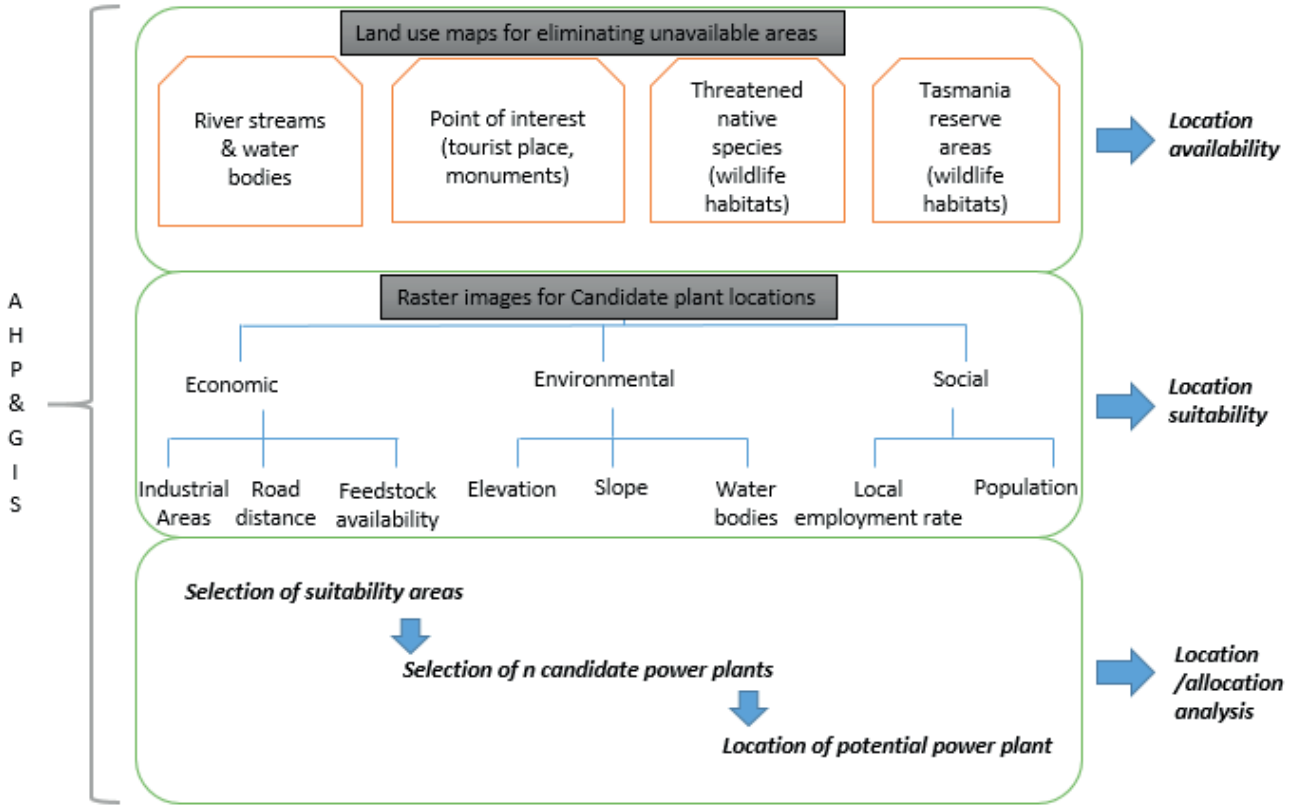

FIGURE 4: GIS methodology for selection of optimal facility sites, adapted from (Woo et al., 2018).

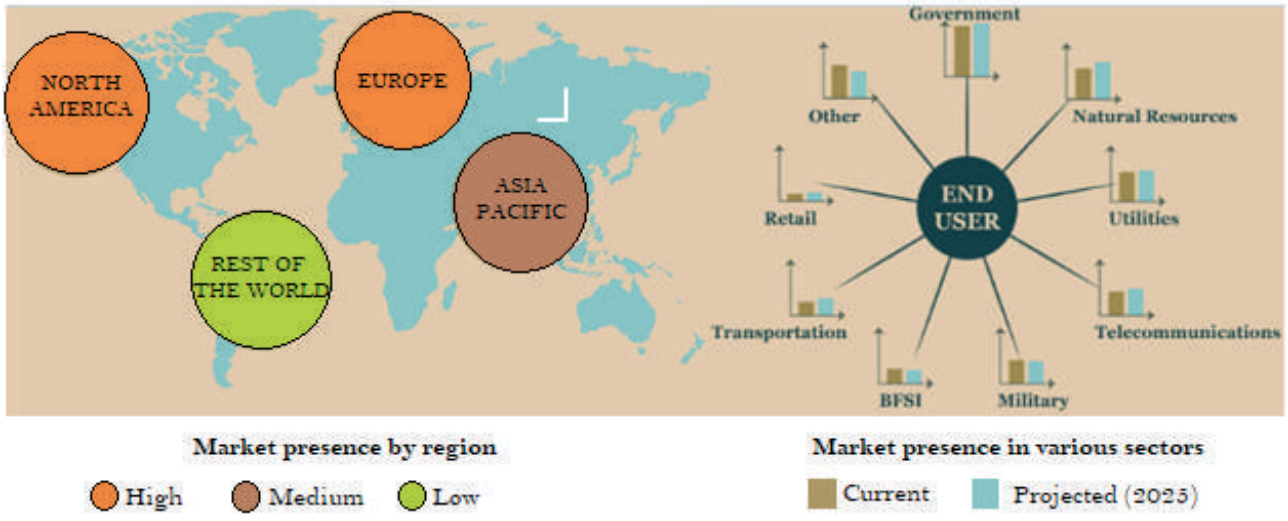

FIGURE 5: Uptake of GIS by various markets and prospects for growth (www.inkwoodresearch.com).

the optimal use of natural resources is increasingly becoming topical, it will be imminent that GIS will soon take centre stage in the planning and allocation of these resources.

On the other hand, most developing regions have experienced a recent rapid urban growth. In the case of Africa mass urbanization since the 1960s resulted in the congestion of areas surrounding major cities and towns, resulting in the increased generation of waste (Matheri et al., 2016; Sufiyan et al., 2015). In the slum areas of some cities, the problem has degenerated into open dumping of SW. Given the booming populations and high urbanization rates in such developing regions, coupled with severe infrastructure and economic constraints, local authorities in these nations should consider optimal management using GIS tools. Integrated GIS technology has been recognized as one of the most promising approaches to automate the process of planning and management of waste management, WtE and bioenergy SC systems (Celli et al., 2008; Chalkias & Lasaridi, 2011; Panichelli & Gnansounou, 2008). Clearly the need for cost-effectiveness cannot be restricted to developed countries for complex segregated waste collection, treatment and recovery. A better opportunity in the green economy, beyond conventional landfill site location, would be planning and location of WtE and pre-treatment sites. These can be a good basis for comprehensive spatial databases revealing demographic data and waste disposal habits (Sufiyan et al., 2015). GIS tools would therefore help developing nations quantify the spatial and temporal characteristics of waste and plan economically for WtE sites.

Africa in particular, along with other developing countries, also boasts of a large inventory of unutilized biomass due to expansive agricultural and forestry land and growing populations. The Stecher, Brosowski, & Thrän (2013), in an International Renewable Energy Agency (IRENA) report, stipulated that bioenergy is a strategic asset in the future of Africa, especially in the light of the fact that it comprises 50% of Africa's total primary energy supply (TPES)(Figure 6) and more than 60% of Sub Saharan Africa (SSA)'s TPES (Stecher et al., 2013). Jingura et.al (2017) remarked that 'biomass is by far the most important renewable resource in SSA'(Jingura & Kamusoko, 2017). Estimates on Africa's collective biomass potential are wide and varied, being classified largely as energy crops, forestry biomass (plantations) then residues and organic waste. The estimates for 2020 are shown in Table 2.

Evidently, the reason why southern Africa would have such a high percentage share of biomass and waste (especially residues) is due to a relatively high abundance of land for energy and food crops, a relatively stable and conducive climate, a thriving agro-forestry industry and a fast growing population rate (Batidzirai et al., 2012; Stecher et al., 2013; Von Maltitz & Setzkorn, 2013). Such fast expanding demographics lay a demand for increased agricultural and forestry products and consequently, the residues accumulated from the activities (Gasparatos et al., 2015; Von Maltitz & Setzkorn, 2013),(Pradhan & Mbohwa, 2014). Given the spatial and temporal distribution of such residues, coupled with global trends, policies and technology advances that are supporting bioenergy, it is imminent that GIS will be widely adopted in the near future for spatial quantification and analyses.

While there is definitely an increase in waste due to population booms in developing nations, developed ones will also experience an increase in variety and possibly, quantities due to better lifestyles (Tan et al., 2014). Regardless of quantities of MSW, the major distinction between the developing and developed nations is the policies framework and legislation, which give developing nations pressure to com-

TABLE 2: Estimates on Africa's collective biomass potential.

Energy Crops	Forestry biomass	Residues (forestry & agriculture) and waste
Up to 13,900PJ/yr (IEA, 2010)	1 billion tonnes annually (~9.4PJ/yr) (Cudjoe et al., 2015)	Just above 0.4billion tonnes per annum (Cudjoe et al., 2015)

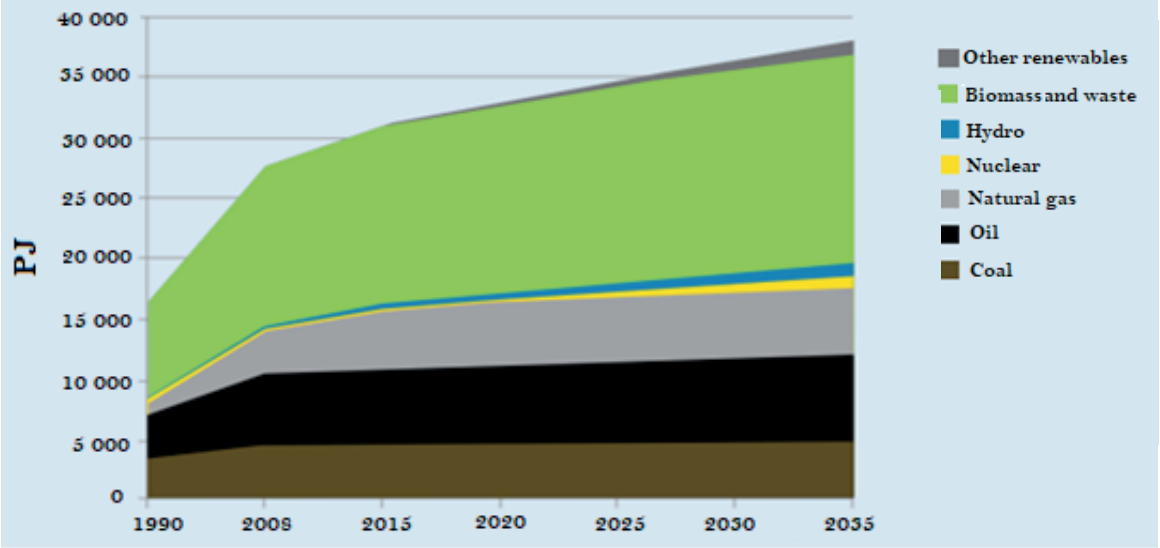

FIGURE 6: Total Primary energy demand for energy sources in Africa (IEA 2010.

ply in efficient ways (Sapp, 2017; World Energy Council, 2016). Chalkias and Lasaridi (2009)'s account of the waste collection and sorting highlighted the EU waste policy that requires source separation and recovery of materials and energy (Chalkias & Lasaridi, 2009). Although waste sorting is not yet a focus area in developing nations, routing optimization can still deliver value owing to the dense populations and prevalence of open site dumping (Chalkias & Lasaridi, 2009). Tan et al (2014)'s model for a developing nation (Malaysia), on the other hand, remains quite instructive in selecting the right mix of WtE technologies using various objectives, when there are limited resources (Tan et al., 2014). In this case, the WtE technologies could also include pyrolysis and gasification alternatives, not only LFG and CHP.

For BtB ventures, it is interesting to note that though not prevalent, cases of unutilized timber residues exist in developed nations as exemplified by the Tasmania, Australia case (Woo et al., 2018). Koikai (2008)'s study is a classic case of the use of GIS alone to identify all candidate biofuels processing sites by ranking according to certain suitability factors in Kenya (Koikai, 2008). Woo et al (2018), however, gave a good example of an even more comprehensive model integrated with simulation and optimization tools to meet various other constraints and objectives, which can only be expressed mathematically (Woo et al., 2018). Moreover, the simulation tool can display results for various scenarios and when expressed graphically, it could be a better marketing tool in developing nations, where more rigour is require to break the ground and convince stakeholders. However, limitations in creating such a robust model may exist in terms of expertise and in some cases, inadequate computational resources in developing nations. This could be solved by having collaborations of researchers in developing nations with those in developed nations for skills transfer and sharing of robust resources. It will also be interesting to note that GIS models as depicted in Figure 4 and the mathematical superstructure will also vary between developing and developed nations due to differences in policies, legislation and socio-economic values or norms.

5. CONCLUSIONS

The review reveals a convergence of various issues like rapid population growth in developing nations, agriculture and forestry growth, advances in computational capabilities and increased policy support for renewable energy schemes. All these constitute a good breeding ground for the application of GIS in creating and analyzing spatial databases with associated, relevant non-spatial attributes. There is indeed a strong case for WtE, bioenergy and improved SW management ventures owing to the wide array of potential socio-economic benefits that could be reaped from them. Given the low energy density of waste and biomass compared to fossil fuels, spatial distribution of supply points and variability of resource quantities, GIS becomes a tool of choice in the optimization of landfill/WtE/bioenergy facility size, site location and routes. In developing nations, where the SW resource has been fast becoming a nuisance and biomass is very abundant, the opportunity for the application of GIS is vast and virgin. An accurate, well conceptualized and built model as exemplified by studies in this review can result in time and cost savings both at the planning and implementation stages. This study also shows that integrations of GIS with other mathematical optimization tools or simulations cover for the former's inability to also optimize on size and technology choice when there are resource constraints. It is also clear that all these SC models will be affected by the differences in policies and legislation between developed and developing nations and the latter may be affected by computational, human and software resources availability. This might warrant an in depth inquiry into the causes of slow uptake of such academic tools like GIS in developing nations to establish the weight of socio-economic or political factors. Perhaps, the bigger gap in literature is on models that will combine the upstream SC, midstream conversion and downstream distribution modules and be able to simulate various scenarios of plant location (determined using GIS), size and technology choice in one package. However, as Charis et al. discuss, such models would require a larger investment in time, a multidisciplinary approach and substantially bigger computational capacities (Charis et al., 2018).

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LEGAL REPORT

CANADA FIRST: NORTH AMERICA ADOPTS CIRCULAR ECONOMY LAWS

With a January 1st, 2019 compliance date now passed, the Ontario Government has been active in implementing North America's first circular economy waste diversion regime[1] ("Circular Economy Law"). The landmark *Resource Recovery and Circular Economy Act*, 2016 (the "RRCEA")[2] adopts the complementary goals of preserving / recovering more resources, diverting more materials from landfills, and reducing greenhouse gas emissions from waste.

A fourth, less overt but truly notable aim, is to create the private market conditions for the growth of a vibrant circular economy driven by innovation, with hope that the resulting infrastructure and expertise will also be exportable across Canada and to the United States.

This paper will consider (i) what was learned in the transition away from a government-coordinated Industry-Funded Organization ("IFO") model to a private sector circular economy law; (ii) the centrality played by Independent Producer Responsibility (IPR) - where brand owners and importers are directly tasked with operating an end-of-life-supply chain with non-transferable liabilities; (iii) the supporting infrastructural measures deemed necessary to IPR's success; and (iv) some of the challenges and opportunities for industry in this nascent circular economy and its intended expansionist future.

PART I: Learning From the IFO Model
Ontario Waste Market Necessitated Diversion

The Province of Ontario is Canada's largest by population and has a waste generation profile all-too typical in North America. In 2014, for example, approximately 11.5 million tonnes of waste were generated in the province - nearly a tonne of waste per person per year[3]. Forty percent of this waste is generated by households with the other 60% coming from industry, commercial businesses, and institutions. Ontario municipalities are responsible for the waste generated from households and collect, process, market and dispose of 4.9 million tonnes of material each year, at a cost of $1.2 billion Canadian dollars.

Municipalities in Ontario have some of the most sophisticated diversion programs in North America, with 95% of Ontario households having access to curbside recycling (Blue Box) and curbside compost programs made available to 71% of households in the province. Ontario's Blue Box program for printed paper and packaging has achieved a recycling performance of 65%.

While municipalities have been driven to increase recycling through programs and regulatory provisions, such progress has not happened elsewhere in the economy and, unfortunately, 3/4 of Ontario's waste has been sent to landfill for the past 10 years[4].

The Ontario government recognized the diminishing waste capacity, the need for resource conservation and the missed economic and environmental opportunities. In fact, waste diversion has been made a critical piece of the province's *Climate Change Action Plan*[5]. The larger benefit, however, as the province has identified, is replacing virgin resources in the economy with recovered resources available in Ontario.

What Did Ontario Previously Create?

The predecessor to the RRCEA is Ontario's Waste Diversion Act ("WDA")[6] which created Waste Diversion Ontario ("WDO") as the regulator tasked with indirectly overseeing the diversion from landfill of a number of waste streams. Included within this group were:

- waste electrical and electronic equipment including computers, screens, peripherals, and audio/visual equipment;
- municipal solid waste streams including glass, metals, printed paper and packaging, and plastics;
- municipal hazardous and special wastes, including batteries, pressurized and aerosol containers, fertilizers, herbicides, insecticides and pesticides, paints and coatings, oil bottles and filters, and antifreeze and solvents; and
- used tires, including on-road passenger and truck tires and off-the road tires.

Industry-Funded Organizations

Through the WDA, the province designated IFOs for each of the target waste streams. IFOs enlisted service providers and coordinated the waste management activities of all of waste diversion participants for each waste stream, including the producers, haulers, collectors, processors, and re-manufacturers (depending upon the waste stream). These IFOs allocated volumes, set fee structures, rated performance, and conducted auditing and performance assessments of all the regulated parties.

With the WDO and the IFOs, two intermediary bodies were, however, placed between the producers[7] (namely manufacturers, first importers, and brand owners) and the end of life supply chain. This was arguably a fatal design flaw that prevented a harmonized approach between regulator and industry.

WDO was to effectively represent the provincial environmental interests regarding waste diversion targets, education, and promotion. The IFOs, in turn, were notion-

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Detritus / Volume 06 - 2019 / pages I-VIII
https://doi.org/10.31025/2611-4135/2019.13827

ally a coordinating body of industry interests, staffed by industry personnel, making specific allocations of waste resources to the various waste diversion participants in order to coordinate the overall waste diversion enterprise (the "Command Diversion Framework"). From inception, IFOs had an unclear enforcement mandate under the Command Diversion Framework in spite of their centrality within it.

An equivalent of the Command Diversion Framework had not been tried on a broad province or state-wide scale previously in North America and there was clearly going to be an element of trial and error in its execution with one or more of the regulated waste streams. The problems, unfortunately, were systemic.

What Went Wrong with the Command Diversion Framework?

Participants in Ontario's waste diversion programs have a long list of complaints with both the structure and administration of the Command Diversion Framework, including:

- the failure to make waste reduction and reuse preferable to recycling - it didn't incentivize these activities distinctly from recycling, making it the near default diversion strategy;
- producers were permitted to externalize their responsibility (both financial and liability), making them disinterested parties with no inducements to innovate;
- as IFOs effectively controlled sector monopolies, service providers, and producers were locked into protocols that left little room for needed deviation;
- the producers' waste diversion fees were, too often, disconnected from the actual costs of diverting the waste;
- poor enforcement and sanction mechanisms permitted a complacent compliance culture in some areas with an unaccounted for producer segment operating outside of the Command Diversion Framework;
- below optimal diversion rates for organics;
- no effective incentives to reduce waste;
- exclusion of important streams, such as Industrial Commercial and Institutional ("IC&I") waste; and
- (perceptions of) lack of fairness, transparency, and certainty in the manner in which the waste resources were allocated among participants, creating uncertainty in the market.

With these attendant problems visible within the first years of inception, calls for a fundamental overhaul have been made repeatedly in the past decade from all stakeholders, with the provincial government finally acceding that the Command Diversion Framework simply did not achieve its goals[8] for most, if not all, of the regulated waste streams[9].

Dismantling the Command Diversion Framework

On November 30th, 2016, the Province of Ontario finally passed the RRCEA which enabled the passage of transitional legislation, the *Waste Diversion Transition Act, 2016*[10], thereby permitting the province to move away from the Command Diversion Framework. The transition from a government-managed scheme to the Circular Economy Law *without the disruption or diminution of waste diversion services and activities* is not going to be easy and the government's planning for the changeover includes:

- moving the government oversight of Ontario waste diversion from the WDO to the Resource Productivity and Recovery Authority (the "Authority"), with the Authority assuming more of a night watchman role;
- enabling the Ministry of the Environment and Climate Change ("MOECC") to directly change current diversion programs; and
- permitting the MOECC to request the wind-up of the IFOs, which is anticipated to be a staged and gradual process in light of the need for continuity.

In short, the WDA and its mixed legacy are near an end, while the waste industry stakeholders (both from within Ontario and elsewhere) scramble to respond to the new regime and its challenges and opportunities.

PART II: Independent Producer Responsibility Enter the RRCEA

Viewed from the vantage point of waste diversion programs across North America and elsewhere, the RRCEA combines ambitious waste diversion goals with the dynamism of a mandated but relatively unfettered diversion market. Shades of the European Union's innovation with waste diversion[11] can be seen in the outcomes sought:

- a registry of all introduced products and their primary, convenience packaging[12] and transportation packaging[13] (giving rise to regulated wastes) will be established;
- "cradle-to-cradle" stewardship obligations imposed upon brand holders;
- design-for-environment; and
- expanded scope of obligated parties to include those with a "commercial connection"[14].

Most important is the clear divide between market participants and the Authority, which shall under the IPR model only:

- operate as a data registry for waste diversion participants (namely producers, generators, and service providers);
- engage in active compliance and enforcement independent from the industry itself (which will include inspections, compliance orders, and administrative penalties); and
- provide limited direct oversight of obligated parties, which may well include producers, municipalities, service providers, and privately-formed collectives of obligated and related service parties.

Notable in its absence is an Authority mandate over policy or waste diversion program development, which shall be assumed directly by the MOECC[15]. Instead, the Authority has signaled a willingness to enforce the RRCEA mandated outcomes but the means used in achieving them will remain with the market participants.

PART III: Ontario's Strategy to Support the Circular Economy Law

In implementing the Circular Economy Law, the province has recognized that it must actively create the conditions necessary for the regime to succeed. Simply supplanting the WDO with a market-based RRCEA is understood as insufficient for a true circular economy to germinate. No fewer than 15 actions[16] have been identified in its Strategy for a Waste-Free Ontario (the "*Strategy*"):

- **Action #1:** *Empower the Resource Productivity and Recovery Authority*
  This is responsive to the widely-held view that the WDO was lacking a sufficiently robust enforcement mandate to compel compliance. The Authority is expressly tasked with "ensuring producer compliance with regulated requirements and a fair system that discourages non-compliance and prevents free riders"[17].

- **Action # 2:** *Issue policy statements to provide clear direction on the provincial interest*
  The province has a laundry list of lofty goals for the reorientation of the Ontario economy away from its current disposal practices. Policy statements are intended to be issued by the Minister and serve as directives to the Authority, updating and supplementing the content of the RRCEA and regulations. They could directly impact municipal decision-making and others that hold environmental approvals.

- **Action #3:** *Establish a registry and build data capacity to provide for evidence based decisions*
  The Authority is to "collect import data from producers and other parties that conduct activities related to waste reduction and resource recovery. These efforts will help the province effectively set targets and develop policies while the Authority monitors and assesses producer performance"[18]. If the province fully succeeds in electronically tracking the introduction and removal of waste volumes (which admittedly is easier in some regulated waste streams than others), compliance levels will no doubt improve, though there may be unintended consequences and resulting industry concern associated with this level of monitoring.

- **Action #4:** *Transition existing waste diversion programs smoothly to new producer responsibility framework without disruption of services*
  The province is highly sensitive to any disruptions in waste diversion services occasioned by the transition. The most difficult waste diversion program to transition to an IPR is Blue Box, a municipally-run waste diversion program for printed paper and packaging and based upon 50-50 shared responsibility[19]. This program is mandated under Ontario Regulation 101/94[20] and requires every Ontario municipality with at least 5000 residents to operate a Blue Box program.

- **Action #5:** *Amend the 3Rs regulations to increase resource recovery across all sectors*
  Provincial regulations under the *Environmental Protection Act* (Ontario), dating from more than 20 years ago, mandated the IC&I sector to take positive steps in the reduction of waste:
  - Ont. Reg. 102/94 Waste Audits and Waste Reduction Work Plans[21];
  - Ont. Reg. 103/94 Industrial, Commercial and Institutional Source Separation Programs[22]; and
  - Ont. Reg. 104/94 Packaging Audits and Packaging Reduction Work Plans[23].

  Together, these process-focused obligations (largely without concrete performance targets) were the original foundational support for IC&I waste reductions goals, frequently referred to as the "3Rs Regulations"[24]. The province no longer views the current 3Rs Regulations as forming part of the future IC&I waste strategy, describing them as:

  > *no longer adequately drive waste diversion. Their requirements are limited to large establishments and only select waste materials, and require only "reasonable efforts" to send source-separated wastes for recycling or reuse*[25].

  Proposed changes to IC&I waste diversion will mean the substantial revision, if not wholesale replacement, of the 3Rs Regulations and are likely to include:
  - concrete diversion thresholds;
  - use of "new technologies" to measure performance;
  - third party monitoring, certification, and audits; and
  - possible imposition of IPR obligations, along with selective disposal bans.

  It is the disposal bans which will be viewed as the most onus of these requirements, if, for no other reason, than the sheer volume of material to be diverted.

- **Action #6:** *Establish service provider requirements to protect the environment while promoting resource recovery*
  The province takes the view that IPR and the Circular Economy Law cannot succeed without a modernization of the support services, including hauling, processing, recycling, diversion, and disposal. As part of this process, the government is seeking to adopt:
  - new national, international, and industry standards for diversion and disposal;
  - new technical recycling standards; and
  - third party monitoring, auditing, and public reporting.

  Ontario's recent end-of-life vehicle environmental standards for disposal sites[26], which covers depollution, waste storage, training, and record keeping, is held up as the model for diversion standards for other regulated waste streams, including diverted waste. There may well be some difficult transitions to come in the event that comparable onerous diversion processing requirements are applied more broadly to all current and impending regulated waste streams.

- **Action #7:** *Ensure landfills are well planned and managed to minimize the need for them and reduce greenhouse gas emissions*
  The province estimates that more than 70% of prod-

ucts within the Ontario marketplace are ultimately sent to landfill[27]. While the *Strategy* acknowledges that there will still be a need for some additional landfills in the province, these will be subject to stringent new approval and operating standards to protect against environmental harms, including to drinking water sources.
More notably, the province plans to develop a landfill gas recapture protocol, focused on methane, which will permit the generation of offset credits under Ontario's new greenhouse gas reduction cap-and-trade scheme, which were made operational in January 2017[28]. This serves as another example of the interdependence of the circular economy and climate change regulation.

- **Action #8:** *Establish promotion and education requirements to support public participation in resource recovery*
The Circular Economy Law places promotion and education requirements for waste diversion upon the industry participants themselves instead of the Authority, who are now obligated under the RRCEA to "ensure consumers are getting the information they need to properly participate in resource recovery efforts"[29].
It is unclear whether those requirements will be tied to waste diversion performance and exactly how such efforts will be overseen by the Authority. Clearly, there will be a role for producers and their Producer Responsibility Organizations (PROs) to coordinate on promotion and training.

- **Action #9:** *Designate new materials to ensure producers are fully responsible for recovering more materials from products and packaging*
The 2009 Canadian Council of Ministers of the Environment Extended Producer Responsibility Plan[30] included a phased plan for the long-term expansion of waste diversion programs to a number of other products and packaging sources. The Circular Economy Law focuses on three of these sources:
- printed paper and packaging;
- food and organic wastes; and
- construction and demolition materials.

  Other waste streams will be resource recovered based upon a host of considerations including:
- viability of end-of-life markets for the diverted waste;
- infrastructure capacity;
- effectiveness of existing non-regulated efforts;
- experience of diversion of such waste streams outside of Ontario; and
- harmonization with existing international efforts.

  It is anticipated that the first phase of additional materials to be subject to circular economy obligations, under the RRCEA, will include:
- appliances;
- electrical tools;
- batteries;
- fluorescent bulbs and tubes;
- mattresses;
- carpets;
- clothing and other textiles; and
- furniture and other "bulky" items.

  It is hoped that the diversion of existing regulated materials, along with others identified through this review and assessment process, will harmonize Ontario's efforts with those of the international community, thereby further opening the door to the collective wisdom of the broader waste diversion industry, including the European Union's Action Plan for the Circular Economy[31].

- **Action #10:** *Implement an action plan to reduce the volume of food and organic waste going to landfill*
Organics bans have yet to be broadly implemented across Canada. Ontario is considering an ambitious province-wide ban on organics which will clearly need to involve municipalities (as the current collector and disposer of household organics) in any such phased in plan. Commercial organic waste generators will need to seek market solutions to their impending diversion obligations.

- **Action #11:** *Implement an Excess Soil Management Policy Framework to increase the reuse of excess soil, while protecting human health and the environment*
The goal of excess soil management is to redeploy excavated soils wherever possible in place of the traditional practices of landfilling such soils as wastes regardless of their current or treatable quality. This can only be done through ensuring that generators of excess soil are implementing proper testing standards which will mitigate environmental risk and managing those soils consistent with any environmental restrictions.

- **Action #12:** *Adopt and implement modern regulatory approaches to build on an promote innovative best practices*
A critical element in Ontario's plan will be a fundamental reconsideration of what constitutes a "waste" so that industry in Ontario can fulfill the fourth Circular Economy Law goal - namely exportable innovation for the use of recovered resources for secondary and tertiary purposes. Notably, the Minister is to develop a:

  > *risk-based approach for compliance and enforcement will also simplify legal requirements and business processes for activities which are lower-risk, less complex or have standard requirements, while continuing to protect the environment and human health*[32].

  In short, Ontario is seeking to revisit traditional waste sector presumptions that material no longer fit for its original use is necessarily a waste to be destined for landfilling. The impediments to innovation posed by the current MOECC waste regulation and practice cannot be overestimated.

- **Action #13:** *Improve and establish environmental standards to provide for a level playing field and a strong foundation for markets*
The province is considering the adoption of a series of environmental standards for recovered materials to

ensure the consistency of feedstock for nascent secondary markets for diverted materials. Standards may include regulatory requirements, guidelines, best practices, and certification programs. The need for robust markets for the diverted materials, fostered by clear standards, is also central to the Strategy.

- **Action #14:** *Use green procurement practices to build market demand for recovered materials*
  Government procurement is to support the circular economy with its preference (in some circumstances) for recovered resources and recycled content, as well as more environmentally-responsible service providers.
- **Action #15:** *Implement disposal bans to direct materials to end-markets*
  Perhaps the most challenging of all circular economy measures coincident with the Circular Economy Law are the proposed disposal bans set for:
  - organics;
  - existing diverted wastes;
  - beverage containers;
  - corrugated cardboard and some paper materials; and
  - fluorescent bulbs and tubes.

  Producers may well have allied interests in seeing that disposal bans can be used to facilitate the diversion markets.

PART IV: From EPR to Circular Economy - Plastics in Canada

The CCME (Canadian Council of Ministers of the Environment)'s Strategy on Zero Plastic Waste, dated November 2018, outlines the commitment of all provincial and territorial ministers of the environment in Canada on plastics waste. The Strategy overtly builds upon various previous multi-lateral initiatives on plastic waste reduction, many of which Canada was only tangentially involved. What is truly new and groundbreaking is the first trilateral federal, provincial, and territorial legislative commitment to a circular economy.

The CCME's last country-wide commitment on product waste was in 2009, with its Canada-wide Action Plan on Extended Producer Responsibility, which largely called for the first forms of targeted waste diversion for a number of target hazardous waste streams. The producers (i.e. product makers, importers, and/or retailers) were tasked to hold some responsibility (financial or otherwise) for such product waste and the move to cross-country, government-sanctioned waste diversion programs accelerated thereafter.

Nine years following its EPR plan, all of the environmental ministers in Canada have wholeheartedly jumped to a commitment to circular economy (on plastics), when it was Ontario alone which had formally committed and legislated these mandates up until now. The Strategy, in fact, goes so far as to call for the involvement of all supply and reverse supply chain parties (a la the Ellen MacArthur Foundation's Global Plastics Commitment) in adopting a systems approach to plastics.

Provincial / Territorial Implementation

The Strategy mandates the adoption of enabling circular economy laws for plastics across Canada, much like the 2009 Action Plan called for, and gave rise to waste diversion laws:

> *The implementation of this strategy will be done within the jurisdictional authority of each order of government and a future action plan will identify complementary measures between governments.*

Further, the Strategy calls for the harmonization of standards and practices across all provinces and territories, in part, to reduce the regulatory burden on business. These are the same calls that have over time gradually pushed provincial and territorial waste diversion laws towards increased consistency within Canada.

PART V: From EPR to Circular Economy - EEE and Batteries in Canada

As the pace quickens in imposing full Extended Producer Responsibility for numerous regulated products and materials under Ontario's Resource Recovery and Circular Economy Act, 2016, the first design-for-environment ("DfE") regulatory standards have been introduced, and e-waste (along with lighting and batteries) will be the test case.

DfE has long featured as an aspirational goal of the European Union circular economy program, but it's been commonly out of reach of institutional product stewardship programs unable to provide incentives for individual producer innovation (see: "Extended Producer Responsibility Models For Delivering Design For The Environment." Jonathan Cocker. Lawtext Publishing. Environmental Liability – Law, Policy and Practice, Issue 6, Volume 24).

This has finally changed under the Resource Recovery and Circular Economy Act, 2016, which mandates DfE as a "provincial interest". Its Electrical and Electronic Equipment Regulation, currently in draft form but with committed compliance dates in 2020 (the "EEE Reg"), is almost certainly North America's first set of DfE regulated standards within a circular economy law.

Electronics Industry Challenged Under EEE Reg

Under the EEE Reg, brand owners and first importers of electrical and electronic equipment (EEE) are obligated to resource recovery the product content for all EEE "marketed" in Ontario for 14 separate categories of informational technology, telecommunications and audio visual equipment (everything from printers to drones).

The EEE Reg sets accelerating rates of resource recovery (or "management") obligations based upon the weight of the EEE marketed (under a prior year experience formula) as follows:

July 2020 - December 2021	57% recovery rate
2022	75%
2023	80%

While these recovery rates are below current practice under the outgoing government-sanctioned product stewardship scheme, there is some expansion in the product

categories caught and there will be a transitional period of adjustment as producers determine how they will individually, or in combination with other like-minded producers, meet their obligations.

EEE Management Obligations

Under an EEE Guideline, which is likely to be a version of the international R2 recycling standard, the principal activities are either:

1. Refurbishment; or
2. Processing for supply for new products and/or packaging.

There is also a very limited ability to count processed glass used as aggregate. Whether any electrical and electronic equipment can be recovered by an EEE producer as compliance under the EEE Reg. isn't clear. In other words, can a company sell thousands of headphones and recycle a few speakers to satisfy its obligations? Clarity is needed.

DfE Reduces Management Obligations

Regardless of how those EEE management obligations are allocated, the real innovation lies in the DfE provisions. There are three types of DfE activities which would qualify as reducing a producer's EEE management obligations, which are capped at 50% of a producer's overall obligation. Each has been proposed as DfE policy but not in North American regulation (and arguably not legislatively in as comprehensive a package anywhere else, including in EU countries):

a) *Post-Consumer Recycled Glass or Plastic Content*
A producer's resulting EEE management obligation will be reduced where its product contains post-consumer recycled glass or recycled plastic content (including, it would appear, recycled content originating from outside of the province.) The consequent reduction in management obligation is reduced by the equivalent amount of the weight of the recycled content. This provision may well align with the push for recycled plastics content standards across numerous categories of plastic-containing products.

b) *Extended Warranties*
Where the EEE marketed in the province contains a 1-3 year warranty or a warranty of at least 3 years, the management obligation is reduced respectively by 5% and 10%. There is no guidance at present as to what the scope or base terms of a qualifying warranty might look like, but clearly there is a push to incent longer lasting products. There might also be a role to be played by third party retailers who commonly offer extended warranties on a range of producers' goods.

c) *Right-to-Repair*
Finally, the EEE Reg. seeks to encourage right-to-repair with an obligation reduction equal to 10 percent of the weight of the EEE marketed in Ontario for products which include both no-cost information on repair (in some medium) and no or cost-recovery only charges for tools and parts to repair the EEE. The logistics and risk management issues associated with the tools and parts supply elements of this scheme may be complicated for some EEE and require more development over time. There is no doubt, however, that right-to-repair is emerging as a product standard.

Most across the electronic and electrical equipment product industries caught by the EEE Reg. will recognize these DfE provisions as works-in-progress at best. Regulated producers should also, however, recognize them as opportunities to finally gain financially for the types of environmentally-beneficial product innovations that have too long laid dormant for want of exactly these types of incentives.

PART VI: CHALLENGES AND OPPORTUNITIES

Will the Circular Economy Law Succeed?

In a vacuum, the introduction of the Circular Economy Law into a "greenfield" municipal-based waste disposal regime would be challenging, given the lack of industry experience, including processing capacity. Given the (sometimes bitter) experience of the past decade under the Command Diversion Framework, however, there is substantial working knowledge of diversion of the existing regulated waste streams which, when supplemented by the collective resource recovery expertise from outside the province, may well be enough to make it viable in Ontario and then beyond. There is certainly a view that some producers will move slowly away from current IFO-based historical groupings.

Used Tires as IPR Test Case

As a first regulated waste stream under the RRCEA, used tires are viewed by many inside Canada and elsewhere as the test case for IPR. To implement, the Ontario government has introduced a draft Tire Regulation under RRCEA[33]. While it remains in draft at of the time of writing, it clearly reaffirms the intention by the MOECC to let private industry decide how to establish and operate an end-of-life supply chain for used tires. Further, the draft Tire Regulation does not propose specific quotas and scoring for the end recovery uses of such tires. This may well engender the type of innovation and industry which can be exported for used tire recovery operations elsewhere.

Many uncertainties remain, however, including the most fundamental quandary facing producers - how can they comply in diverting a waste stream that they don't control and their PROs can't command?

Further, like all resources, there will be more and less preferred sources, with differential costing based upon location, resource collection efficiencies, certainties of supply, etc. There is already concern among producers of various regulated waste streams that the prime diversion sources need to be secured far in advance of the anticipated Circular Economy Law compliance date.

Finally, the Strategy, in some sectors, seems to require an economy of scale that effectively pushes segments of producers into PROs, but this may create an unintended complications under Canada's Competition

Act[34], which contains prohibitions on oligopolies and other restrains of trade. In other words, if some producers act jointly with due diligence and seek to secure the diversion supply they need to offset their product waste, their actions may be punishable by Industry Canada where the resulting PROs are viewed as engaging in anti-competitive activities.

Could Ontario Become the Circular Economy Model for North America?

Every other province and territory in Canada operates waste diversion programs, along with many US states and municipal entities. The products subject to diversion under Ontario's Circular Economy Law will be pre-dominantly the same supply "imported" into the other provinces, states, and territories in North America. As such, a workable solution in Ontario has the real benefit of being replicable at scale elsewhere in North America.

In fact, the Province of Ontario is effectively betting that, as a first mover in North America, that it develops the innovation with Ontario's scheme to then reproduce the know-how and industry elsewhere - a cottage industry with aspirations as a North American leader.

This was the plan with Ontario's Feed-in-Tariff renewable energy program introduced in 2008 where industry was to migrate to the province to produce "domestic content" renewable hardware in support of renewable energy. To a lesser degree, this was the plan for the Command Diversion Framework. Neither has worked as expected in spite of the substantial expenditures of capital. Arguably, the Ontario government did not step aside so as to allow sufficient certainty in the market for long-term investment.

More importantly, however, there is a clear mandate within the province to become the centralized source for waste diversion innovation and industry will be looking for scalable solutions, which can be mass produced across North America as the regulatory schemes mature[35], backed by consistent standards and outcomes.

In short, the ability to replicate the Circular Economy Law elsewhere in North America is arguably a fundamental premise upon which it has been legislated in Ontario so its expansionist goals should come as no surprise.

Resource Recovery "Wild West" in Need of Diversion Expertise / Solutions

What is also clear from the impending Circular Economy Law are the needs of producers and related industry parties to find new environmentally-sound, yet market savvy strategies for their waste streams. As North America's first such program, the experience and expertise from the European Union and elsewhere will be highly valued in giving obligated parties the assurance that resource recovery and environmental compliance will be achieved using market-leading technologies.

The impending move to IPR has also created a period of volatility where innovation, experience, and business relationships are forming and reforming with uncertain results. As the Circular Economy Law will, to some degree, be replicated elsewhere in North America, the current Ontario model offers waste management participants with a rare opportunity to define the future.

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ACKNOWLEDGEMENTS

Special thanks to Peter Hargreave and his invaluable insights upon which this publication depended.

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U. Valiente (2016). A Practical Pathway to Producer Responsibility for Paper Products and Packaging in Ontario, December 6th, 2016, Corporate Policy Group LLP

[1] Defined under the *Resource Recovery and Circular Economy Act,* 2016, 2016 SO 2016, c. 12, Sch. 1 as "an economy in which participants strive: a) to minimize the use of raw materials; b) to maximize the useful life of materials and other resources through resource recovery; and c) to minimize waste generated at the end of life of products and packaging.

[2] SO 2016 c. 12.

[3] Ontario Ministry of the Environment and Climate Change, Strategy for a Waste-Free Ontario: Building the Circular Economy, (Queen's Printer for Ontario, 2016) at p. 4 ["Strategy for a Waste-Free Ontario"].

[4] *Ibid*.

[5] Ontario Ministry of the Environment and Climate Change, Climate Change Action Plan 2016-2020, (Ontario Ministry of the Environment and Climate Change, 2016) accessed online: <http://www.applications.ene.gov.on.ca/ccap/products/CCAP_ENGLISH.pdf>.

[6] 2002, SO 2002, c. 6.

[7] The notion of a "producer" has been supplanted under s. 59 of the RRCEA with "brand holder", defined as a person who owns or licences a brand or who otherwise has rights to market a product under a brand.

[8] Strategy for a Waste-Free Ontario, *supra* at p. 4.

[9] Notably, IFO Ontario Tire Stewardship had the highest diversion rates for reasons which are only partially attributable to its relatively manageable and specialized set of waste diversion participants.

[10] SO 2016, c 12, Sch. 2.

[11] European Commission, *Closing the loop - An EU action plan for the Circular Economy*, (EUR-Lex: European Commission, Brussels, 2.12.2015)

[12] Defined under RRCEA as "material used in addition to primary packaging to facilitate consumer's handling, or transportation of one or more products, such as boxes and bags.

[13] Defined under the RRCEA as "material used in addition to primary packaging to facilitate the handling or transportation or one or more products by persons other than consumers,, such as pallets, bail wrap and boxes, but does not include shipping containers designed for transporting things by road, ship, rail or air.

[14] Under s.61(3) of RRCEA, a person who imports, wholesales, leases or retails product or is otherwise involved in the regulated product's distribution

[15] It could be argued that WDO also omitted any policy development work, but their central position and obfuscated mandate crowded out the MOECC from much policy initiative.

[16] Strategy for a Waste-Free Ontario, supra at 14-35.

[17] *Ibid*, at 14.

[18] *Ibid*, at 17.
[19] Many of these concerns are highlighted in Usman Valiente's *A Practical Pathway to Producer Responsibility for Paper Products and Packaging in Ontario,* December 6th, 2016, Corporate Policy Group LLP.
[20] *Recycling and Composting of Municipal Waste,* O Reg 101/94.
[21] *Waste Audits and Waste Reduction Work Plans*, O Reg 102/94.
[22] *Industrial, Commercial and Institutional Source Separation Programs*, O Reg 103/94.
[23] *Packaging Audits and Packaging Reduction Work Plans*, O Reg 104/94.
[24] Draft *Strategy for a Waste-Free Ontario*, *supra* at 22.
[25] *Ibid*, at 22.
[26] *Registrations Under Part II.2 of the Act - End-of-Life Vehicles*, O Reg 85/16.
[27] *Strategy for a Waste-Free Ontario*, *supra* at. 24.
[28] *Climate Change Mitigation and Low-carbon Economy Act*, SO 2016, c. 7.
[29] *Strategy for a Waste-Free Ontario*, *supra* at 26.
[30] *Ibid*, at 27.
[31] *Ibid*, at 23.
[32] *Ibid*, at 32.
[33] Resource Recover Policy Branch, Ministry of the Environment and Climate Change, *Draft Tire Regulation under the Resource Recovery and Circular Economy Act, 2016* (Public Consultation) (Ontario Ministry of the Environment an Climate Change, December 2017).
[34] RSC 1985, c C-34.
[35] While it might be easy to be pessimistic in viewing the near term commitment to environmental initiatives at the federal level in the United States, waste diversion continues to develop at the state and local levels.

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RESEARCH TO INDUSTRY AND INDUSTRY TO RESEARCH

GAS EMISSION MONITORING: A GROWING CHALLENGE FOR MSW INCINERATORS

In the era of bustling industrial activity and growing number of industrial plants, regulation and continuous monitoring of emissions have become increasingly important.

Despite emissions monitoring being a standard and fairly straightforward process for all industrial plants, it is of utmost importance: it protects the nature and human beings proactively and ensures the safety of us all. What is more, by effective process optimization we can save a lot of valuable resources.

This is particularly true for MSW incineration plants where the perception of human health risk is particularly sensible and consequently the public attention is very high.

What makes emissions monitoring challenging is the changing operational environment where regulations, standards and processes can change. As the operational environment changes, the emission limits and allowed gas concentrations change consequently. This, in turn, requires continuous adaptability and capability to reassess the procedures and technologies by the emissions monitoring stakeholders.

Besides, emission monitoring is important for research activity because it allows to evaluate the results of process optimization on a full scale or to define the emission factors for environmental or human impact evaluations. This represents a relevant case in which industrial equipment is necessary for improving research of environmental engineering applied to air pollution control.

An interesting chance to fulfill this growing challenge of emissions monitoring is the use of FTIR (Fourier Transform Infrared) gas analyzer technology. FTIR is one of the most reliable and effective technologies out there for emissions monitoring. It is the most powerful technology for simultaneous measurements of multiple gases and the industry standard in many emissions monitoring applications. What is more, its flexibility and versatility make it a very cost-effective and multipurpose tool for measurements of a wide range of gas components in various industrial processes requiring continuous emissions monitoring.

How ftir technology works

FTIR stands for Fourier Transform Infrared spectroscopy. It is a powerful gas measurement technology for simultaneous measurements of multiple gases. The ability to detect and measure almost any gas, combined with the robustness and reliability of the technology, makes FTIR ideal for a wide variety of applications.

Nearly all molecules can be identified by their characteristic absorption spectrum as each molecule absorbs infrared radiation at its characteristic frequencies. Every molecular structure has a unique combination of atoms, which produces a unique spectrum when exposed to infrared light. The only exceptions are noble gases and diatomic elements such as O_2, which do not absorb IR light. IR absorption spectrum can be thought of as a fingerprint unique to each molecule.

The infrared spectrum is a plot of infrared radiation related quantities as a function of wavelength or wavenumber. According to Lambert-Beer's law, the absorption strength (peak height) is directly proportional to concentration, and this way the IR spectrum can be used to determine concentrations of gases in the sample.

Two interesting applications that apply FTIR technology are described below (Figure 1):

- **Gasmet CEMS** IIe is a solution for a broad range of demanding continuous emissions monitoring applications. CEMS IIe has both TÜV and MCERTS certificates according QAL1 EN14181/15267. All parts of the system are heated to 180 C and this extractive system is ideal for measuring pollutants from hot, wet and corrosive gas streams. The solution is typically used (but not limited to this) to simultaneously measure the following 16 gases: H_2O, CO_2, CO, N_2O, NO, NO_2, SO_2, HCl, HF, NH_3, CH_4, C_2H_6, C3H8, C_2H_4 and CH_2O. On top of these, FTIR analyzer can also measure Total Organic Carbon (TOC) and NOx.
- **Portable DX4000** measuring system is an ideal "portable" tool for stack testing. In this configuration, just a single analyzer can be easily and quickly used by a single operator to measure almost all criteria pollutants. This gives a great chance to save a lot of time and effort in field, and grant real time reliable gas concentration analysis but also give the option to save all data for storage or post analysis study. According actual European Legislation EN 14793-2017, portable FTIR can also be approved as AM (Alternative Methods) to replace traditional SRM (Standard Reference Methods) and this fact gives to analysis laboratories the option to use FTIR analyzers to perform QAL2 and AST into EN14181 systems periodical quality checks.

Main benefit of ftir technology

- **It is futureproof** – Quick changes to measurement ranges and additions to list of measured gases are possible and easily done. New gases and gas ranges

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Detritus / Volume 06 - 2019 / pages IX-XI
https://doi.org/10.31025/2611-4135/2019.13825

(a) (b)

FIGURE 1: FTIR technology applications: (a) Gasmet CEMS IIe; (b) Portable DX4000 measuring system.

can be added to the analysis at any time without any changes to hardware. This can be done by the users themselves.

- **Multicomponent capability** – FTIR is the only technology that measures such a wide range of gases simultaneously. All key emissions monitoring compounds such as NOx, SO_2 and CO can be measured. In case of waste incineration, also HCl, HF and TOC are added to the measured gases, and NH_3 if SCR or SNCR is used.
- **Proven technology** – Gasmet FTIR analyzers hold TÜV and MCERTS certificates (QAL1) according EN14181/15267. All Gasmet's FTIR emissions monitoring systems have been certified throughout their history.
- **Reliability** – FTIR is extremely sensitive, accurate and robust technology. It provides low maintenance cost for the user and long system life-time combined with low uncertainty. Additionally, its high availability provides low downtime for the operators.
- **Application flexibility** – Of course with FTIR it is possible to supply CEMS (Continous Emissions Monitoring System) suitable for most of industrial applications as i.e. for waste incinerators. But moreover it is possible to have fully portable gas analyzer suitable for stacks testing (quick checks, process research, emergency situations, reliability field tests, quick backup solution, ...).

Example of ftir "library" for cems application

Gasmet FTIR equipped with CALCMET software is able to detect and quantify up to 50 gas in the same time, of course the software must be equipped with a specific "library" that includes typical absorption spectra for each gas to be analyzed including all possible cross interference components.

Here following Gasmet experience about CEMS application into incinerators for both stationary and portable applications- a typical library. Of course it can be changed/modified for other gas and ranges at any time (Figure 2).

Gasmet Technologies (www.gasmet.com) is a Finnish high technology company that develops, manufactures and markets FTIR gas analyzers and monitoring systems for a variety of industrial, environmental and safety applications.

Ital Control Meters srl (www.italcontrol.it) is an Italian company with more than 25 years experience as exclusive distributor of high level technologies for process automation and emission control aimed to Italian industries, including Gasmet products.

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EXTENDED CEM APPLICATION GAS-APP-003							
#	Compound name	Formula	CAS number	Minimum range	Typical range	Maximum range	Unit
1	Water	H_2O	7732-18-5	25	30	40	vol-%
2	Carbon dioxide	CO_2	124-38-9	10	20	30	vol-%
3	Carbon monoxide	CO	630-08-0	60	500	10000	ppm
4	Nitrous oxide	N_2O	10024-97-2	50	100	500	ppm
5	Nitrogen monoxide (Nitric oxide)	NO	10102-43-9	100	200	2000	ppm
6	Nitrogen dioxide	NO_2	10102-44-0	100	200	500	ppm
7	Sulfur dioxide	SO_2	7446-09-5	30	100	2000	ppm
8	Ammonia	NH_3	7664-41-7	20	50	500	ppm
9	Hydrogen chloride	HCl	7647-01-0	10	50	500	ppm
10	Hydrogen fluoride	HF	7664-39-3	3	10	100	ppm
11	Methane	CH_4	74-82-8	20	100	1000	ppm
12	Ethane	C_2H_6	74-84-0	*		200	ppm
13	Ethylene (Ethene)	C_2H_4	74-85-1	*		200	ppm
14	*n*-Propane	C_3H_8	74-98-6	*		200	ppm
15	*n*-Hexane	C_6H_{14}	110-54-3	*		200	ppm
16	Formaldehyde	HCOH	50-00-0	15		70	ppm

* The CEM hydrocarbon ranges depend on the application. Higher ranges and additional compounds are available upon request from Gasmet Technologies Oy.
Note: Standard GAS-APP-003 application package includes one range per compound

FIGURE 2: Example of FTIR "library" for CEMS applications into MSW incenerators.

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PORTRAITS

Prof. Dr. Frederick George Pohland
born: 1931 in Oconomowoc, Wisconsin
died: 2004

Prof. Pohlands research led to fundamental advances in anaerobic processes. His concept of a landfill as a bioreactor through controlled leachate recycling was fundamental. He was honoured:
"For advancing the theory of anaerobic treatment processes and applications to solid waste management."

Education

- M.S. 1958 in environmental engineering at Purdue University.
- Ph.D. 1961 at Purdue University.
- Sc.D. (honoris causa) from Valparaiso University in 1996 for "excellence in service to the human family" as an engineer, student, explorer, teacher, professor, and writer interested in "the ways in which humans use and misuse that most vital of all elements - water".
- He was awarded the prestigious Harrison Prescott Eddy Medal by the Water Environment Federation (WEF) in 1964 for his excellent dissertation and research.

Scientific carrier

- He led the environmental engineering programs at the Georgia Institute of Technology (1961-1988).
- and then at the University of Pittsburgh (1989—2004).
- He was also Visiting Scholar at the University of Michigan (1967-1968).
- and a Guest Professor at the Delft University of Technology in the Netherlands (1976-1977).
- Prof. Pohland was Professor and Edward R. Weidlein Chair of Environmental Engineering at the University of Pittsburgh; after his death, he was awarded an Emeritus Professorship, the first time the university conferred this honour posthumously.

Professional Activities & Research work:

- Prof. Pohland's research led to fundamental advances in anaerobic processes. His concept of a landfill as a bioreactor through controlled leachate recycling was adopted by the Delaware Solid Waste Authority, among many others. He originated and chaired the International Water Association (IWA) Specialist Groups on Anaerobic Digestion (1985-1992) and Landfill Management (1995-1999), which brought together practitioners, researchers, and educators from all over the world.
- In keeping with his commitment to applying research to practice and disseminating vital information, he served as regional editor of *Water Research* and *Water Science and Technology* (both 1993-2002) and as honorary executive editor of *Water Research* (1994-2000).
- From 1991 to 1998, Prof. Pohland chaired the American Academy of Environmental Engineers (AAEE) WASTECH Program, which developed two series of books for practitioners in remediation.
- Prof. Pohland was the author of more than 150 technical and scientific publications.
- Chair of the WEF Program Committee (1989-1992); he worked to ensure that all professionals in the water quality field were supplied with the latest science and technology.
- Prof. Pohland served on the Environmental Protection Administration Science Advisory Board (1989-1997), where he worked hard to ensure that science and technology were used in the regulatory process.
- Election to the National Academy of Engineering (1993),
- Honorary memberships in WEF (1993) and IWA (2000).

Honors

- In 1989, WEF honoured him with the Gordon Maskew Fair Medal for "exemplary demonstration of proficient accomplishment in the training and development of engineers in the environmental engineering field".
- Bestowal of the Gordon Maskew Fair Award (2000),

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Detritus / Volume 06 - 2019 / pages XII-XIII
https://doi.org/10.31025/2611-4135/2019.13829

- Presidency of AAEE (1992-1993),
- Selection as the American Society of Civil Engineers (ASCE),
- Simon Freeze Memorial Lecturer (2001),
- Bestowal of the Association of Environmental Engineering and
- Science Professors Frontier Award (2003).

Citation
"Frederick George Pohland." National Academy of Engineering. 2007. Memorial Tributes: Volume 11. Washington, DC: The National Academies Press. doi: 10.17226/1191¬2.

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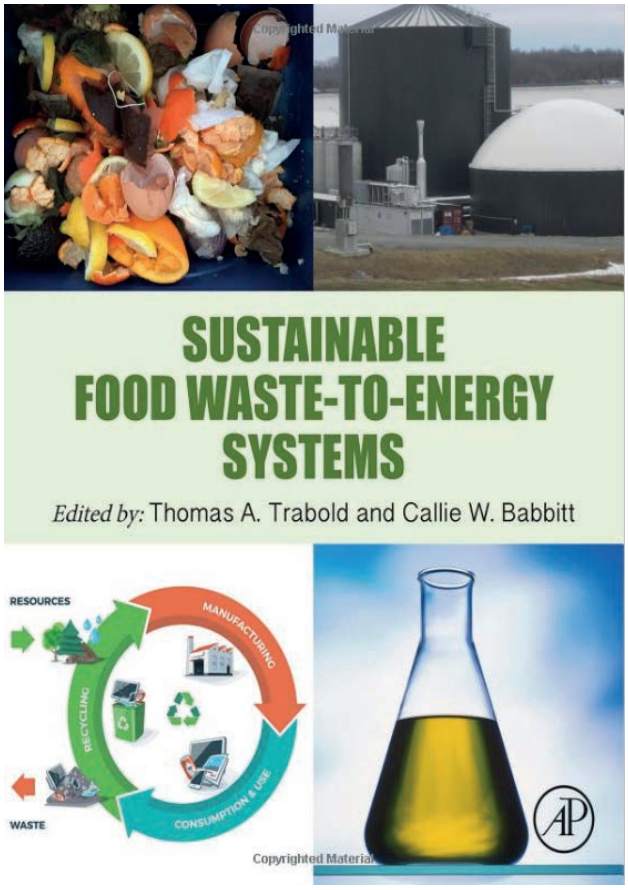
BOOKS REVIEW



SUSTAINABLE FOOD WASTE-TO-ENERGY SYSTEM

Edited by Thomas A. Trabold and Callie W. Babbitt

The relationship between human beings and food has evolved considerably throughout the ages.

Food has assumed a huge significance in representing cultural and religious traditions, ethnicity and national identity; however, in developed economies the population has very little direct involvement in the production of food, thus failing to view wastage of this commodity as particularly arrogant or vulgar.

Taking into account all contributions provided by wastes ranging from agricultural residues to postconsumer waste, the food waste fraction is estimated at 30 to 40% of total food produced, the majority of which is sent to landfill. Whilst much of the developed world faces issues related to minimization of food wastage, other much larger populations continue to be affected by severe food insecurity.

"Sustainable Food Waste-to-Energy System" is an extensive overview focused on the production and general management of food waste and conventional and innovative waste-to-energy technologies. The figures and examples reported refer largely to a US context, thus resulting in a series of evident differences compared to the European, and particularly South European Mediterranean Area, where food preparation is a large part of daily life. Nevertheless, the technical solutions discussed for promoting food waste-to-energy systems are valid worldwide.

Despite the interest in discussing the political and social implications of food waste production and reuse, this book approaches the problems from a technical point of view and, highlighting the imbalance between food supply and demands, indicates the inefficiencies present in the global system (Chapter 1). To develop a viable strategy to address the food waste issue, a series of trends have been considered, including: - the abundance of food (upward trend in per capita food supply); - the inexpensive cost of food (average food expenditures have dropped by 23% of disposable income in 1929 to 11% since 2000); - the production of food is minimally labor-intensive (agriculture employs less than 3% of available labor force); - it is no longer only a local resource (in 2013 the US imported 19% of food based on volume). Increasing food production, transport and consumption have all impacted on packaging and food waste production and greenhouse gas emission.

Available waste resources (Chapter 2) are defined by the FAO as "food loss" caused by inefficiencies in the food supply chain, "food waste" indicating food discarded by humans, and "food wastage" food affected by deterioration. On a global perspective, consumption losses in industrialized economies are much higher than throughout the developing world, where the majority of food waste is generated during the production, handling and storage processes. It is highlighted that despite the differences in distribution, in Europe, industrialized Asia and sub-Saharan Africa the percentages of food waste are similar (22-25%), and lower than those detected in North America and Oceania (42%).

Conventional food waste management options (Chapter 3) are described using a modified waste hierarchy (following source reduction as the preferred alternative). Food donation and animal feed production are viewed on a par with waste-to-energy (WtE) technologies, whilst less desirable management alternatives include composting, wastewater treatment (liquid food waste), incineration, and landfilling. Successful policies applied in Japan and Korea in the development of animal feed products from food waste are illustrated, and it is reasonable to expect that this waste management strategy will play a bigger role in the future. Composting is discussed related to US databases,

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Detritus / Volume 05 - 2019 / pages XIV-XV
https://doi.org/10.31025/2611-4135/2019.13828

but also as a simple technology acknowledged worldwide for its effectiveness in restoring nutrients to soil.

Anaerobic Digestion (AD) (Chapter 4) for methane production is presented through process parameters and operating conditions, biogas production potential, performances of the different type of reactors (the table on key design parameters is particularly well described), co-digestion with animal manure and biogas utilization. On the contrary, the discussion on digestate fate - one of the main features of AD - is completely lacking.

Fermentation (Chapter 5) as a biochemical process used to convert organics into bioethanol is mainly discussed as the most common commercial system in use, although biobutanol and biohydrogen production are also mentioned, closing the chapter with a brief discussion on future perspectives based on US data.

Transesterification (Chapter 6) is an acid or base-catalyzed reaction performed to convert triglycerides from oil into biodiesel. The potential waste streams suited to biodiesel production are discussed, with waste cooking oil deemed the main feedstock. The chemical reactions and operating parameters are described by a series of comprehensive tables and use of the by-product glycerol utilization is also discussed.

Bioelectrochemical systems-BES (Chapter 7), including microbial fuel cells (MFCs) and microbial electrolysis cells (MECs), using bacteria to convert organic and inorganic matter and directly produce electrical current or hydrogen are perhaps the most interesting systems presented in the book due to the innovative aspects relating to food waste-to-energy conversion. A wide range of feedstocks is illustrated by means of an extensive literature review, in addition to the MFCs and MECs processes, potentiality, current limitations and perspectives.

Gasification and Pyrolysis (Chapter 8) as thermochemical conversion of specific biomasses such as animal-sourced food waste, poultry and fish processing residues, vegetable skins, rice and corn residues, featuring a less than stoichiometric level of oxygen required for full combustion or incineration, are discussed. Production of hydrogen-rich syngas, liquid bio-oil, and biochar are potential options, although additional research, particularly bench-scale to pilot-scale studies, should be conducted to investigate any food waste technology combination in order to avoid scale-up problems.

Hydrothermal liquefaction (Chapter 9), in the same way as the chapter on BES, this technology is relatively new and highly promising for the conversion of food waste, particularly wet biomass, by depolymerization under conditions of moderate temperature and high pressure into a high energy density bio-oil product. Conversion of carbohydrates, lignin, oils, fats and proteins as single materials are discussed, as well as source-specific food waste (plant based, animal based, mixed food waste). Further research should be undertaken due to the complexity of food waste, in an attempt to better understand the influence of different operational conditions and successfully market the end-product.

Other chapters of the book present a series of environmental and economic aspects, as well as the regulatory framework based largely on US cases.

This book provides a collection of excellent contributions covering numerous aspects of the food waste-to-energy systems. It represents a useful resource for professionals (researchers, managers, engineers, technicians, operators) working in the field of waste management, energy systems, energy supply and management, and process systems.

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ABOUT THE EDITORS

Thomas A. Trabold

Thomas A. Trabold is an Associate Professor and Department Head of the Golisano Institute for Sustainability (GIS) at Rochester Institute of Technology (RIT). Dr. Trabold's primary research focus is in development of alternative energy technologies, including fuel cells, bio-fuels, and food waste-to-energy processes. Prior to joining RIT, he had 20 years of experience in industrial research and development at General Electric, Xerox, and most recently in research and demonstration of proton exchange membrane (PEM) fuel cell systems for advanced zero-emissions vehicles at General Motors. In the latter position, he was a Professional Technical Fellow and Laboratory Group Manager, with responsibility for engineering research activities in the U.S. (New York and Michigan) and Germany. Dr. Trabold has a strong record of accomplishment in sustainable energy research, with more than 100 technical publications and 50 U.S. and international patents.

Callie W. Babbitt

Dr. Callie Babbitt is an Associate Professor in the Golisano Institute for Sustainability (GIS) at Rochester Institute of Technology (RIT). Her research group focuses on developing and applying tools to understand and manage the life cycle implications of emerging technologies. Specific focal areas are consumer electronics, nanomaterials, photovoltaics, and lithium ion batteries, and food waste-derived biofuels. These sectors represent complex sustainability challenges, as they are characterized by rapid development, adoption, and evolution; high potential for environmental impact across all life cycle stages; and a lack of comprehensive data that can be used to accurately quantify potential environmental impact. Dr. Babbitt is a recognized expert in environmental assessment of emerging technologies, and was awarded an NSF CAREER Grant in 2013.

Book Info:

Editors: Thomas A. Trabold, Callie W. Babbitt
Imprint: Academic Press
Year of publication: 2018
Page Count: 292
Paperback ISBN: 9780128111574

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A PHOTO, A FACT, AN EMOTION

"These men live in Kolkata, India, collecting waste plastic, glass bottles, and ferrous materials, which are then marketed for recycling. The wastes are collected from the highly polluted "Bagjola" waste water canal - the men have no protection to use on their hands, but their livelihood and that of their families depends on this work."

"SOURCES OF LIVELIHOOD"

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Detritus / Volume 06 - 2019 / page XVI
https://doi.org/10.31025/2611-4135/2019.13830

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VOLUME 06 / June 2019

ISSN 2611-4135 / ISBN 9788862650601