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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 indexed in Emerging Sources Citation Index (ESCI) Web of Science, Scopus, Elsevier, DOAJ Directory of Open Access Journals and Google Scholar**. Detritus is an official journal of IWWG (International Waste Working Group), a non-profit organisation established in 2002 to serve as a forum for the scientific and professional community and to respond to a need for the international promotion and dissemination of new developments in the waste management industry.

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Editorial

THE FAILURE OF GEOMETRY: TIME FOR GEARS!

A schematic representation of the manufacture of goods and management of associated wastes frequently results in an extensive use of geometrical figures: lines, triangles and circles.

The task of these geometrical figures is to portray a graphic synthesis of concepts and strategies. Therefore, the actual concept underlying the basis of the representation is of fundamental importance. The concept first and foremost, followed by the figure that should not overpower the concept by straining, conditioning or, worse still, distorting it.

As an example, the concept of "take-make-waste" based on the use of a renewable or non-renewable resource in the manufacture of goods and subsequent disposal of associated wastes represents the so-called linear approach depicted by a straight line. A line starts and ends at fixed points. The simplest linear system in product and waste management (Figure 1A) moves directly from the soil (resource extraction) and ends up in the soil (waste disposal). A line has no branches, does not retrace its steps, does not reflect and has no afterthoughts. Important material resources are buried in landfills. No form of consideration for the safeguard of non-renewable resources, environmental protection is of a passive nature and environmental sustainability is virtually completely overlooked. Indeed, processes such as the recovery of biogas from landfills or energy recovery following waste incineration deviate from the perfect linearity of the system.

As the concepts of a hierarchy and integrated waste management took hold (initially established in Europe by EU Directive 75/442/EEC), the line was gradually replaced by a top-down triangle listing the different stages of waste management in a top-down priority order (Figure 1B). This representation indicates a priority order of Prevention, Preparing for re-use, Recycling, Energy recovery and Disposal based on increasingly minimised quantities of waste. Landfills are viewed as a bin into which the smallest possible quantities of waste should be introduced, thus resulting in a lower risk of pollution and occupation of space.

The rigidity of the triangular geometric form elicits an idealistic order which is scarcely relatable to the real world. For example, nowhere in the world are wastes subjected prevalently to prevention measures, and little is undertaken at a regulatory level to promote and apply these measures. The triangles are steeped in moralism. Landfills should be in a position to become virtually obsolete, but people simply seem to be unable to manage without them (Cossu et al., 2020). The triangles moreover seem to convey dreams rather than the actual situation. Furthermore, the rigidity of

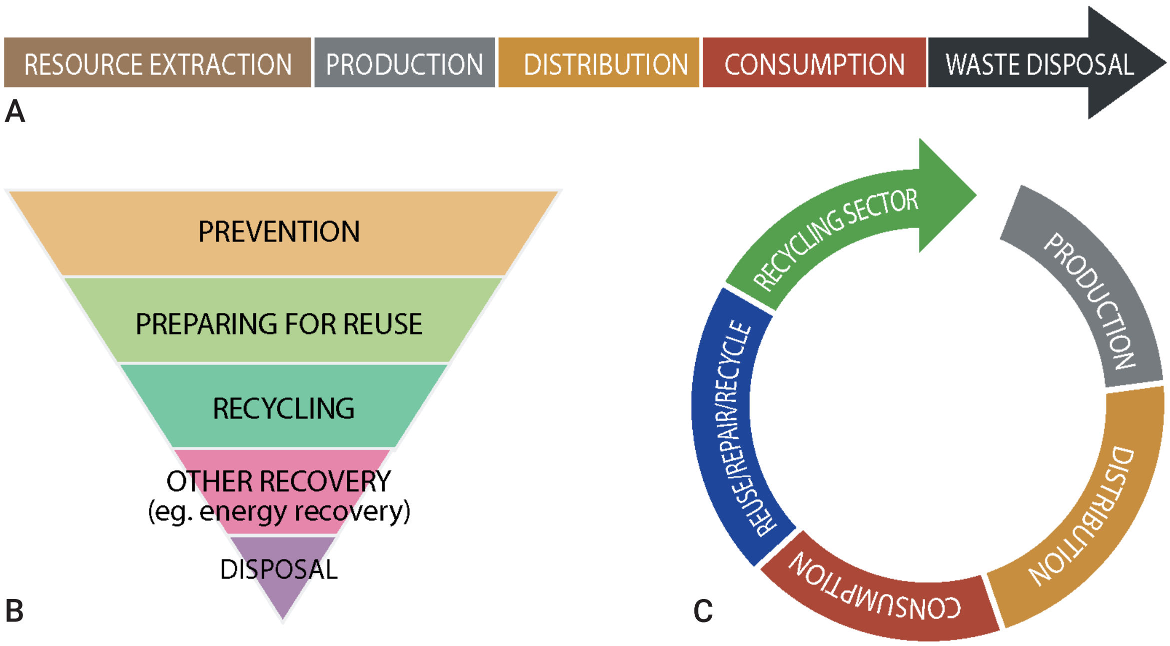

FIGURE 1: Schematic representation of a series of concepts relating to the production of goods and management of associated wastes.

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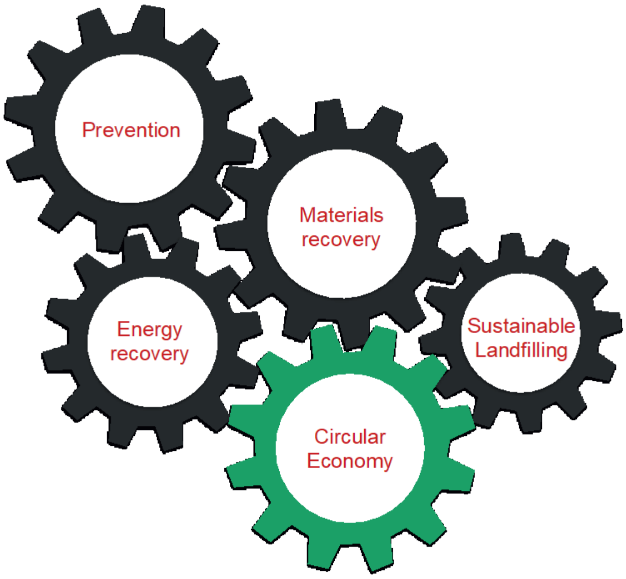

FIGURE 2: Representation of a balanced solid waste management strategy aimed at achieving the functional (environmental, technical and economic) management of Circular Economy.

the triangle makes us lose sight of the virtuous role played by landfill in closing the material loop. Within triangles, materials seem almost to volatise, to disappear!

To now proceed to the circle. In all cultures, the circle has long conveyed the notion of perfection and movement. In waste management, the circle is inevitably elevated to the symbol of Circular Economy, i.e. a system rotating around the prevention, reuse, recovery and recycling of wastes, stoking various production cycles with secondary raw materials. On a European level, the aim declared was to "close the loop" of product lifecycles bringing benefits for both the environment and the economy. The circle is well aware of and satisfies the need to contrast the depletion of non-renewable resources through the reuse of resources present in wastes, thus highlighting the benefits to be gained in reducing waste volumes forwarded to landfill and raising awareness amongst those who generate the wastes with an aim to improving waste quality and reducing waste quantities. However, this geometrical image of the perfect circle creates dangerous expectations and boosts the myth of the Zero Waste principle, seen idealistically as a waste management system no longer reliant on landfills or thermal waste treatment plants. In particular, the perfect circle fails to take into account the following fundamental aspects:

- not all materials are recyclable, and those that are cannot be recycled ad infinitum;
- hazardous and persistent chemical substances present in the products forwarded to recycling tend to accumulate in the recycled materials and residues;
- the material cycle should necessarily be closed in line with the principle of Back to Earth to control the global diffusion of contaminants; accordingly, if what is taken from the land is not returned to the land in an uncontaminated form, it will linger perilously in the environment (Grossule, 2020);
- in closing the material loop, the strategically important role played by landfill is not taken into account;
- the entire waste management system is intended to be sustainable and exhaust all negative environmental impacts within the time frame of one generation; this aspect however does not take into account landfills, the regulations for which are not geared towards environmental sustainability nor the technical specifications suited to achieving the same;
- the issue of climate change is viewed prevalently in terms of minimisation of greenhouse gas emissions, neglecting the sequestering and immobilisation of carbon ("carbon sink") which, once again, could be achieved by landfills.

Following these failures of the perfect geometric forms, perhaps the time has come to turn to other forms of graphic representation that indulge less on rhetoric and demagoguery and are firmly anchored to the principles regulating the exchange of energy and materials in the natural cycles in which all stages play a role, in which there are no good or bad solutions, where everything is harmoniously correlated.

The natural cycles do not seek moralistic perfection but tend towards the rationality of nature. Where everything moves in line with the principle of mechanical gears, having instrumental, interconnected roles perpetuated throughout the diverse processes, transformations and reserves of resources, products and wastes.

Explicitly, the representation of product and waste management as a system of gears in which each piece is fundamental clearly illustrates how prevention, material recovery, energy recovery, and sustainable landfill are all of priority importance in guaranteeing an equilibrium in the functioning of Circular Economy.

This is a message of paramount importance to be conveyed to people in order to avoid preconceived oppositions to thermal treatment plants and sustainable landfills.

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THE NORMALITY OF INDUSTRIAL AND COMMERCIAL WASTE: ECONOMIC, TECHNICAL AND ORGANISATIONAL BARRIERS TO WASTE PREVENTION

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ABSTRACT
This study shows that the production of industrial and commercial waste should form part of normal organisational practices. When asked about waste prevention, representatives of food, textile, electronics and construction companies in Sweden have difficulties highlighting concrete waste-prevention objectives, measures and outcomes. Instead, they highlight economic, technical, and organisational barriers that prevent them from engaging in waste prevention, thereby endowing the production of waste with an economic, technical and organisational rationality. This triple rationalisation of waste production amounts to the managerial normalisation of waste that obstructs the implementation of waste prevention policies. Thus, we suggest that these policies aim to de-normalise industrial and commercial waste in similar ways to the measures used to de-normalise household waste.

1. INTRODUCTION: WASTE PREVENTION – FROM PRIORITISED AREAS OF WASTE POLICY TO BUSINESS PRACTICES

After decades of policy objectives, the question remains of how the goal of waste prevention should be achieved in practice. Waste prevention has long been a top priority in European waste policy. The importance of waste prevention was already mentioned in a European directive from 1975 (European Commission 75/442/EEC) and 1977 (European Commission 1977). These directives cited waste prevention as being a priority of waste management, ahead of reuse, material recycling and energy recovery. The Waste Framework Directive from 2008 (2008/98/EC) reconfirmed that the prevention of waste was the most important step in the waste hierarchy. The waste directive from 2018 (2018/851) also emphasised that "waste prevention is the most efficient way to improve resource efficiency and reduce the environmental impact of waste".

In light of these waste directives, it is important that Member States take appropriate measures to prevent waste generation, as is also demonstrated, for example, in Sweden's National Waste Plan 2011–2017. This emphasises that "waste prevention initiatives will also result in smaller quantities of hazardous substances in products and materials." (Swedish Environmental Protection Agency, 2012:28). Furthermore, waste prevention was already mentioned in the special national plan on for waste prevention for the period 2014–2017 (Swedish Environmental Protection Agency, 2015), and in the National Waste Plan for 2018–2023 (Swedish Environmental Protection Agency, 2018).

There is no shortage of individual waste prevention initiatives. This is demonstrated by the approx. 16,570 actions that *The European Week for Waste Reduction* (2019) listed in 2019. The purpose of these actions is to prevent the generation of waste. This includes providing information about the need to prevent waste, increase material efficiency and promote more sustainable consumption (Corvellec, 2016). However, apart from these pilot programmes, it is still not clear how waste-producing companies are working practices on waste prevention measures.

Previous research on waste prevention has primarily focused on household waste. For example, researchers have studied whether or not households should be controlled or supported in recycling more waste and reducing their household waste (Gregson et al., 2013; Corvellec & Czarniawska, 2015; Corsini et al., 2018; Matsuda et al., 2018), particularly food waste (Setti et al., 2018; von Kameke & Fischer, 2018). Researchers have also studied how laws, regulations and policies can be used as a means of control, primarily in households, in order to reduce household waste (Zacho & Mosgaard, 2016; Johansson & Corvellec, 2018).

It may seem odd that researchers have not focused on industrial and commercial waste, seeing that it is several

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times larger in scale than household waste. The few examples of research in this field have focused on the ambiguous role of waste companies (De Jong & Wolsink, 1997; Svingstedt & Corvellec, 2018) or the limits of urban governance regarding waste prevention (Silva et.al., 2017; Hutner et al., 2017; Zapata Campos & Zapata, 2017). Far less interest has been shown in how the companies that produce industrial and commercial waste perceive their opportunities to develop waste prevention practices, i.e. what organisations are doing in their day-to-day work to minimise the volume of waste produced, for example, through zero-waste manufacturing (Kerdlap et al., 2019).

Through the use of a practice theory approach (e.g. Reckwitz, 2002), the purpose of this article is to highlight and address the perceptions and practices of waste producers in the field of waste prevention. Greater knowledge of the perception and practices of producers will provide better conditions for proposing solutions to promoting waste prevention. We propose three unlocking measures that are fundamental to reducing the volume of industrial and commercial waste: making waste prevention more profitable, increasing knowledge of materials, and developing competence in the field of waste prevention.

This article is based on qualitative empirical material. Through a strategic selection of organisations in the four focus areas listed as priorities by the Swedish Environmental Protection Agency (2015) (food, textiles, electronics, construction and demolition), 26 interviews were conducted. Document analyses, focus group interviews and expert interviews were also conducted.

The article is structured as follows: First, there is a brief account of the practice theory approach, followed by a description of the qualitative method we used to collect and analyse our data. This is followed by a presentation of the respondents' descriptions of barriers (cf. De Jong & Wolsink, 1997; Wilts et al. 2013; Bartl, 2014; Corvellec et al., 2013; Shahbazi et al., 2016; Aid et al., 2017) to waste prevention and also a discussion about the normality of waste generation (cf. O'Brien, 2008) and changing social practices to enable increased waste prevention.

2. A PRACTICE THEORY APPROACH TO WASTE PREVENTION

Practice theory focuses on what is done in organisations. It is not an individual theory but a collection of social theories that in different ways assume that people develop routines, habits and ways of doing and saying things in their daily lives – practices. Practice theory is used to understand social reality in all its richness, which is often difficult to interpret, sometimes with confusing contradictions. This is in stark contrast to the focus on structures, which is otherwise a common way of understanding and identifying solutions to the challenges faced by organisations.

In order to clarify what a practice is, we have applied the following, much-quoted, definition of a practice:

A 'practice' is a routinised type of behaviour which consists of several elements, interconnected to one other: forms of bodily activities, forms of mental activities, 'things' and their use, a background knowledge in the form of understanding, know-how, states of emotion and motivational knowledge. A practice is thus a routinised way in which bodies are moved, objects are handled, subjects are treated, things are described and the world is understood. (Reckwitz, 2002:250)

A practice comprises routinised activities that involve several different elements such as objects, body, knowledge and language which are interwoven with, and interdependent on, each other. There is a reciprocity that does not suggest one element over the others. Thus, a practice cannot be reduced to only one of these elements. A practice is always social in the sense that it is about a way of behaving and understanding one's reality that emerges in different places, at different times and with different people. Nor does the practice ever take place in isolation, but together with material objects and/or people and, in order to be a practice, it must be continuously reproduced by those who perform it (Shove & Panzar, 2005).

In an effort to make it easier to study a practice that interweaves several different elements in interdependence, Shove and Pantzar (2005) argue that the practice involves actively integrating the elements of material, competence and meaning. The practice depends on a specific combination of these three elements (Shove et al., 2012). The first element, material, the hardware of practices, is of a more physical nature, such as an object: a tool, a technology or some other infrastructure (Hand et al., 2005; Shove et al., 2007). The human body, which through its bodily movements performs the practice, is also viewed as a kind of material.

The second element is the competence that people possess. Competence includes both knowledge in order to actually perform a task, and the knowledge that is required to be able to evaluate an achievement. Performing a practice also requires a kind of background understanding and know-how that is shared with others. Thus, people who take part in and perform a practice share an understanding of what is being said and done (Schatzki, 2001). To summarise, it is consequently a question of possessing practical competence and knowledge that enables you to actually perform a practice and understand what is happening in a practice. Taken together, these competences form the basis of the cognitive capacities that employees need to be able understand, engage in and actually perform the specific practice, for example – like this study – a waste prevention practice.

The third and final element involved in a practice is meaning or motivation. The fact that a practice creates meaning for those who perform it is a prerequisite for being able to reproduce it and thus maintain it over time. Meaning is partly about history and context (for example, this is what we do in this organisation), and a practice can also show that the organisation is forward-looking – there is a notion of the future of the practice (Shove et al., 2012). This means that when a practice is performed, it is both coloured by history and future-orientated.

In conclusion, the practice comprises and depends on material, competence and meaning, and if changes occur in any of these elements, the practice will also change

(Hand et al., 2005). As a starting point, let us take a practice such as mobile shopping – a digital shopping practice that has emerged in recent years – to illustrate the reciprocity of the three practical elements (Fuentes, Bäckström & Svingstedt, 2017; Fuentes & Svingstedt, 2017). The practice of mobile shopping relies on a technical device, a mobile phone, which consumers use for mobile shopping. The consumer needs a certain level of digital competence in order to shop using their mobile phones, as well as competence that they develop through digital tutorials, chat groups and social media. This enables them to use their mobile phones seamlessly and efficiently when shopping. Mobile shopping has meaning because consumers experience greater independence, as they can control how, where and when they shop. The same applies to practices relating to waste prevention apps, which aim to encourage people to repair their products instead of buy new products, or apps that enable them to find restaurants and shops selling leftover food at a discounted price.

The three practical elements cannot be separated, they are interwoven to the point that they both assume and constitute each other (Shove & Pantzar, 2005). A dynamic combination of material, competence and meaning are necessary in order to make everyday life work.

By applying a practice theory perspective to waste prevention, we provide the basi for an understanding of prevention based on what is being said and done – or not. It becomes a way of understanding and conceptualising prevention as one social practice among other social practices that exist in organisations. In other words, a practice theory approach offers both an everyday and a theoretically substantiated understanding of waste prevention.

3. METHOD: DATA COLLECTION AND ANALYSIS

This study is based on a qualitative study of organisations in Sweden that are active in the four focus areas listed as priorities by the Swedish Environmental Protection Agency (2015) in Sweden's waste prevention programme. The collection of empirical material was undertaken in two phases, while the analysis was undertaken on an ongoing basis.

Before the first phase, we conducted a systematic search of a number of Swedish industry journals in order to identify organisations acknowledged as having an interest in, or that worked with, waste prevention. These organisations were contacted and asked whether they would like to participate in interviews. During the interviews, we asked for additional tips on other actors who they felt could be included in the study. This is known as a snowball selection (Silverman, 2004).

The collection of empirical material in the first phase was undertaken through 26 individual interviews with sustainability managers, purchasing managers, regional managers or quality and environmental managers: six respondents from the food sector (food organisations, food cooperatives, food wholesalers and municipalities), six from the textile sector (fashion chains, design brands and a textile collection organisation), seven from the electronics sector, (sellers of products such as computers, computer accessories, appliances, household appliances and organisations that are major consumers of computers and computer accessories, as well as organisations dealing with the reuse of computers, and municipalities that undertake extensive purchasing of computers are also included in this category), six from the construction and demolition sector (construction and property organisations). In addition, one interviewee represented an organisation that is active in the sanitation industry.

Seven of the interviews were conducted on site, while 19 were conducted over the phone. The interviews were structured on the basis of a prepared interview guide with a number of specified questions addressing topics such as barriers and opportunities in work on waste prevention, and the circular economy (see Table 1 for examples of questions). We adopted a reflective approach to the qualitative interview (Holstein & Gubrium, 1995), using this to investigate how the respondents perceived waste prevention in their respective organisations, described problems and barriers and discussed the way in which they addressed waste prevention and, in purely practical terms, reduced industrial and commercial waste. The sequence of questions changed after the first five interviews as the respondents tended to become focused on descriptions of their "success stories", leaving less scope for subsequent questions about problems and perceived barriers. Following this change of sequence, the interviews focused more on the barriers and challenges that waste prevention can entail.

Supplementary data were retrieved in documents such as European Directives (2008/98; 2018/851), Swedish waste plans (Swedish Environmental Protection Agency, 2012, 2018) and waste prevention programmes (Swedish Environmental Protection Agency, 2015).

The second phase of collection of empirical material was based on three focus group interviews and expert interviews. Two of the focus groups involved representatives of the food, textile and electronics sectors, as well as the construction and demolition sector, and one group involved representatives of the project's reference companies. The interviews were conducted via Skype, lasted for around 45 minutes, and were recorded and transcribed verbatim. On these occasions, and interview guide was used as a basis with questions about how a company could work to reduce waste or be more resource or material efficient (see Table 2).

We also collected material via four individual expert interviews with people who had extensive experience of working with waste prevention in areas such as textiles and construction and demolition, as well as policy analysis and environmental investigations. The purpose of these interviews was to validate our preliminary results from phase one and parts of phase two (see Table 3).

TABLE 1: Examples of questions in the individual interviews

What experience do you have of problems or barriers in your work on waste prevention?
What kind of measures would you like to see from the state/government agencies/industry associations to contribute to increased waste prevention?
How do you work with waste prevention and in what ways are you encouraged to do more of this work?

TABLE 2: Examples of questions in the focus group interviews.

How do our results appear in relation to your knowledge and experience of waste prevention?
Do you recognise these barriers or have you seen other barriers than those we have identified?
Do you have any other comments or views about the contents of the report?

TABLE 3: Examples of questions in the expert interviews.

What barriers do you see for work on waste prevention?
How are you currently working with waste prevention?

Analysis of the material collected was undertaken on an ongoing basis. Phase one identified the fact that the respondents highlighted three main barriers to waste prevention: commercial, technical and organisational. The quality of this classification was validated via the focus group interviews in phase two. In addition, the material that was collected on each of the three main barriers identified in phase 1 was coded based on the three types of practice elements to which Shove and Pantzar (2005) refer: material, competence and motivation. This coding was conducted based on an effort to explain why work on waste prevention is being prevented hindered or enabled by commercial, technical and organisational practices. Phase two also used a number of expert interviews to validate the analysis of waste prevention practices conducted in both phase one and two. These expert interviews resulted in certain observations being refined, which led to findings to the effect that the production of waste is the norm for companies, while waste prevention remains something that is vague, relatively alien.

4. BARRIERS AND SOCIAL PRACTICES THAT MAKE WASTE PREVENTION MORE DIFFICULT

This section presents the respondents' descriptions of three kinds of barriers to increased waste prevention and the social practices behind them. The first is a commercial barrier, which stems from the fact that it costs more to prevent waste than to produce and manage waste. The second barrier is the lack of technical solutions and/or conditions for preventing at least certain kinds of waste. The third barrier is organisational barriers, for example, that established norms and values lean more towards producing waste than preventing the generation of waste. The technical and economic barriers to waste prevention by public bodies identified by Wilts et al. (2013) can be recognised in these three barriers, as well as some of the technological, economic, organisational, as well as legal, informational and social barriers to material efficiency in manufacturing identified by Shahbazi et al. (2016), and the economic, social, technological, information-related, and policy-related barriers to inter-organisational waste management identified by Aid et al. (2017).

What is interesting about these barriers and the practices related to them is that not only do they provide an understanding of the lock-ins (Corvellec et al., 2013; Svingstedt & Corvellec, 2018) that prevent waste prevention according to the producers of industrial and commercial waste. They also indicate what waste producers could do to start working on waste prevention. Practices that obstruct waste prevention of industrial and commercial waste explain, albeit in reverse, the practices that are being developed in order to reduce and also prevent the generation of waste.

The way that the respondents chose to express themselves in the interviews should be understood in light of the fact that the subject of waste prevention can be sensitive. There may be a need to depict their organisation as a waste prevention organisation. Many of the respondents have a responsibility for working in such a way and would want to defend and portray their organisation's waste prevention work on waste in an overly positive way when we asked critical questions. This is a common example of what is known as the interviewer effect (Silverman, 2004).

4.1 Commercial barriers

Interviewees from all four areas emphasised the key importance of delivering a good financial result. As one of the experts stated: *It is the commercial perspective that dominates the other barriers* (Expert 1). The central role from the commercial perspective assumes different forms. For some organisations, it is a challenge to match purchases and sales to minimise leftovers:

/.../leftovers, that's really our biggest headache I'd say, because it's both an environmental issue and an economic issue. Of course, it's about all this material that is left lying on a shelf. It has a value, after all; we've paid for it. It really is in our interest that it is used. And, of course, there's an environmental impact if you produce something that you don't actually need and is ultimately just thrown away. (Sustainability Manager, textiles)

The optimisation of matching between purchases and sales is a waste prevention practice that is perceived as meaningful by employees because it is focused on reducing costs and increasing efficiency. Cost awareness of waste prevention regarding industrial and commercial waste is evident along the entire value chain.

Another example of how finances control waste practices is an accepted way that the construction sector orders timber products. This takes place on several levels by different actors in the production chain. It usually results in all actors at all levels placing slightly larger orders than are needed in order to avoid causing a halt in production. There is a deeply-rooted cultural awareness in the construction sector that whenever a construction project is halted, major costs are generated.

It is a cultural issue, which people have become used to. Input products cost less in relation to labour and other costs, so sometimes you order a little extra just in case, which naturally results in an increase in the volume of waste. (Sustainability Manager, construction and demolition).

A well-established practice, involving competences and procedures that have been developed over the years, is to to ensure that time is saved on projects, while overlooking the relevance of saving on waste. The cost of handling construction waste is so minimal that it plays less of a role in large construction projects (Svingstedt & Corvellec, 2018), as was also confirmed in an expert interview:

We also noted that waste is considered cheap in the construction sector. The costs associated with construction waste are negligible. (Expert 2)

The relatively low cost of waste and its management contributes to giving the issue of waste a low status and a priority that is difficult for employees to find meaningful.

Conversely, a higher cost of waste increases the interest in waste prevention. If waste in the construction sector is about goods with low value added, waste in the clothing industry is about goods that are ready to be sold and are therefore significantly more expensive. It costs an organisation not to sell a textile product. The production of waste is then regarded as a problem, the cost of which must be reduced. A product that is not sold remains the responsibility of the organisation, something they must pay to dispose of.

A common key driver for many of the organisations in the prevention of industrial and commercial waste is the relative cost of what is thrown away in relation to how much it costs to dispose of it. The production of waste appears to be the result of a commercial balance between, on the one hand, the cost of the material and the degree of processing what is being discarded and, on the other hand, what is required to manage the waste in relation to the costs of any additional work and interruptions to production. Consequently, increased resource efficiency that may result from waste prevention practices is developed not through caring for the environment, but rather by a managerial calculation.

4.2 Technical barriers

Technical barriers are about difficulties in developing, finding and using technical solutions that can reduce waste. When it comes to technical barriers, finances must also be taken into account. Respondents put technical solutions in relation to their costs. What they are able to engage in is usually governed by the costs associated with testing technological innovations.

One Sustainability Manager for a major fashion organisation described how the organisation was participating in a project in which it was testing the prototype production of clothing produced from materials that can be recycled through the separation of fibre:

But to be perfectly honest, there's a great deal of manual tweaking and individual handling when you're manufacturing prototypes. It wouldn't have worked to produce a lot, not even 1,000 garments, which is what we'd need to produce in order to put these on the market. So, this is more like a pilot study. However, it is certainly not commercially viable yet, unfortunately (comment: spring 2018), although we thought it might be in 2013. (Sustainability Manager, textiles)

Within the organisation, people are aware that it is not possible to create technical solutions that work and generate financial returns with immediate effect. However, they are not prepared to bear the development costs that are required for full-scale production.

Innovation is seen as a way of developing the organisation's knowledge base and know-how in order to understand the technical requirements:

At first I thought it was mainly a matter of learning 'what's it all about? What kind of opportunities are there? What are the opportunities for collaboration that maybe don't exist today? (Sustainability Manager, textiles)

Waste prevention practices are something that must be tested, developed, implemented in the operational organisation, accepted by others and developed further. However, at present, the technology (materials), know-how (competence) and motivation (meaning) do not exist for this to become a fully sustainable technology (Shove et al., 2012). The technology is lacking to some extent, making it difficult for new waste prevention practices to be developed. In textiles, technological innovations often require collaboration throughout the entire supply chain. Collaboration with other actors to take part of innovations appearing, for exemple, among suppliers. This means that, in the textile organisation, there is no change in the day-to-day practices. Many respondents also believed that responsibility for prevention rests with other actors, outside their own organisation. They wanted to believe that other, larger actors would assume responsibility for development.

Currently, it's not really clear what you're supposed to do when you recycle textiles; it's quite complicated. We're too small and don't have sufficient resources to take a lead in that kind of development, but we try to keep up and see what is happening". (Sustainability Manager, textiles)

Development is expected to take place primarily outside the business's own organisation or by identifying ways of collaborating with other actors in the supply chain.

The same kind of hope for future technological development was expressed by our respondents in the food industry. A current problem with many food packaging items is that they are designed on the basis that there should be larger packaging items for multi-person households, even though many households in Sweden are small, which increases the risk of food waste, even if it also reduces packaging waste per unit of weight or volume. However, changing the size of packaging involves costs, as a Sustainability Manager in the food industry reminded us:

It's always the case in the food industry that you try to minimise packaging solutions, while at the same time that you have invested in a packaging line, so you're rather stuck for a while. It's quite expensive to change – you don't change the size of the packaging every day. (Sustainability Manager, food)

Here, as is so often the case, it is a combination of economic and technical factors that lock practices into packaging solutions that risk entailing increased food waste. New practices are difficult when both technology, and to some extent competence, are lacking, although there does seem to be some motivation for change.

Apart from a general hope for technical solutions, respondents in all areas described an interested but cautious approach to the introduction of waste preventing innovations. Some respondent stated that they had tested a number of innovative solutions on a smaller scale, but did not express any ambition to invest in the development of new and more effective waste prevention technologies. Becoming involved in innovation projects to test different solutions or to invest in some form of technical aid is considered both unsafe and risky. There is still a lack of technology for new and more effective waste prevention practices.

4.3 Organisational barriers

The respondents also experienced different forms of organisational barriers, procedures and practices in their work on waste prevention, which, according to the respondents, indirectly affected waste prevention in the organisations they represented. These have been created over time, are part of everyday work and contributed to the organisation producing more waste than necessary. Organisationally, the production of waste represented normal practice, and not to do so would then become abnormal.

The respondents were aware that some organisational procedures generate more waste than necessary. There is an understanding that changes are needed in work methods and that such changes are based on a mindset that says *this is the way we've always done it* – it is about habitual ways of thinking, ways that create meaning for employees. One purchasing manager in the electronics sector believed that a culture has developed that makes work on waste prevention more difficult:

Every department that has purchased a phone also owns this phone, even though we have services in which we receive end-of-life equipment that takes care of this. So we haven't really purchased equipment that we want. It's probably the organisational culture, which has always believed that you should keep owning your own hardware. (Purchasing Manager, electronics)

This quote makes it clear that the organisational culture obstructs practices directed towards waste prevention.

Other organisational barriers are a lack of control and influence over the production of waste by other actors in the supply chain. The respondents were aware that waste was produced at every stage of the value chain, while at the same time they felt that their organisation did not have sufficient insight into other stages of the value chain and was therefore unable to influence these parts. They only controlled their own stage, which limits what they can do to reduce the generation of waste. Regarding systems for collecting used clothes and textiles, for example, one regional manager in a textile company stated:

Whether or not a consumer wants to pass on, throw away or burn garments, is left to chance. It's up to the consumer and the collector can also largely do whatever they want. /.../ A lot is left to the interests of individual actors. (Regional Manager, textiles).

Existing practices do not consider the whole production process, and there is therefore neither insight into nor the opportunity to control the processes of other actors that generate waste.

From conversations with the respondents, it is also evident that organisations are not always organised to work on waste prevention. In turn, this results in organisations tending to look to other actors, both inside and outside the value chain, in the hope that they will be organised to deal with the generation of waste.

This study indicates a wide range of social practices that prevent the way of the increased prevention of industrial and commercial waste, based on how the organisation of work and material flows is conducted. Taken together, all of the barriers we have identified represent major challenges for increased waste prevention. Our respondents described costs, technology and ingrained work methods as effectively putting a brake on the prevention of industrial and commercial waste. Legal, informational and institutional barriers (Wilts et al., 2013; Shahbazi et al., 2016) are additional factors. At the same time, these barriers also highlight potential ways of moving forward with the issue.

5. DISCUSSION: CHANGING SOCIAL PRACTICES FOR INCREASED WASTE PREVENTION

The previous section showed how respondents approached the daily practices of waste prevention. They mainly described the barriers and practices that challenge their work on waste prevention.

Several structural impediments to waste prevention have been identified in the literature. Based on the Dutch case, De Jong and Wolsink (1997) have highlighted the entwining of interests of municipal and regional authorities in waste collection and waste disposal. They also suggest that it is in the interests of private collecting organisations to keep governmental policymakers in a situation of uncertainty about the volume and type of waste that will be released in order to avoid the formulation of a deliberate waste policy. Looking at the European Union, Bartl (2014) lists several barriers to waste prevention: a conflict of interests between reducing the quantity of waste and therefore the amount of materials that have to be processed, and securing the turnaround and profit of waste collectors, recyclers, waste to energy facilities, and landfill operators; the absence of decoupling waste generation from economic growth; unclear measurement of waste prevention; the interest of producers and retailers in increasing production and sales that go against extending the life span of products; producers' lack of interest in reusing; the export of waste that shifts the waste burden from one country to another.

Adopting a managerial rather than a structural perspective, our respondents stated that waste prevention is hampered by a narrow, commercial view of waste-related income and expenses, a lack of technical solutions, and organisational standards that are not based on preventing the generation of waste. These economic, technical and organisational barriers overlap with the institutional, technical, cultural and material lock-ins identified by Corvellec et al. (2013) in waste management infrastructures. They discourage any change in the social practices of the organisational management of waste in order to promote waste prevention.

When the respondents offered their managerial views on waste prevention, they primarily talked about what they were doing or would like to do in terms of recycling, despite the fact that the waste hierarchy (2008/98/EC) clearly distinguishes between prevention and recycling. The fact that the interviewees chose to talk about recycling when asked about waste prevention shows that even though the general principle of the waste hierarchy is known, the understanding of the technical specifications that characterise its various stages is more vague.

In terms of Shove and Pantzar's (2005) practice theory, firstly, the commercial barriers imply motivation, technical barriers imply materiality, and organisational barriers imply competence. This study shows that the gap between policy goals, the ambitions for increased waste prevention and what is actually being done is explained by the fact that the respondents lack motivation, material solutions and competence to develop practices that focuses on reducing waste. What characterises a practice is that the different elements are interwoven and interact, which makes it difficult to isolate one individual element. In the three barriers we have identified, it is important how the different practice elements interrelate.

Material – the hardware of practice – is a prerequisite for being able to perform a social practice (Shove et al., 2012). The study shows that technology, technical support systems are either lacking or inadequate for managing waste prevention. The available technology aimed at reducing waste volumes does not meet the financial criteria defined by the organisations. When finances do not permit the development of new technology, and the technology that does exist is not suitable for contributing to waste reduction, it is difficult to create the motivation to change waste practices. The conditions to constitute a social practice aimed at increased waste prevention do not exist. Modern technology shapes a social practice that is focused on waste production. New hardware is needed, as well as new technological solutions, which are also likely to require other types of business models in order to encourage change.

Competence is the second practice element involved in a social practice. This is about both knowledge in order to work in purely practical terms on waste prevention measures and also to evaluate such measures. This study shows that there is reliable information, based on a traditional commercial approach, of how profitability is created. The respondents share an understanding of how products are produced that do indeed generate waste, with the least amount of time wasted. Also, the basic view is that reducing waste is usually not worthwhile, and there is a lack of competence in how reducing waste can be made economically viable. The practice is based on norms and values that describe how to deal with the waste that is generated (cf. Corsini et al, 2018, but for households). Attempts made to change the norms in organisations are primarily about making it more natural to increase the level of recycling or reuse, and not reduce waste per se. An organisational change in norms is needed that helps make it more natural to reduce waste than to produce it.

There must be motivation or meaning – the third and final element that constitutes a practice – in doing what you do. Employees must have a common understanding of the existing practice (Reckwitz, 2002). This study shows that it is meaningful for respondents to have a practice that follows a strict commercial approach to the organisation's waste production. There is, however, no justification for reducing waste, as such a practice may increase the organisation's costs. Current production technology creates meaning by offering what is needed to make its production efficient from a strictly economic perspective. Thus, it is difficult to create the motivation for potential technological development. Organisational values, norms and procedures are based on the production of waste being viewed as natural. On the other hand, it is not meaningful to have practices that focus on waste prevention.

The social practices in the respondents' organisations are not structured for managing waste prevention. There is no available waste prevention technology, there is a low level of competence in waste prevention, and there is no motivation to reduce organisational waste. The normal, everyday social practice in these organisations is to produce waste, not to prevent it from being generated. It is true that waste is equated with the destruction of resources, a kind of wastage that should be limited. Nonetheless, our respondents normalised the production of waste on the grounds that it was commercially justifiable, and also technically and organisationally unavoidable.

As one of the reviewers stated, the respondents provide an example here of what O'Brien (2008, p. 178) calls the paradox of waste: an ability to "simultaneously express value and non-value". They regard waste production as the destruction of value, yet a constituent feature of their business logic of profit-making (cf. Svingstedt, 2012). For them, waste is troubling and regrettable, yet it is the normal consequence of prioritised practices such as delivering on time, monitoring technical developments, offering a wide product range or driving down costs.

Conversely, with the exception of representatives of organisations that have made waste prevention their business concept, the respondents stated that they lacked the motivation, material opportunities and organisational competence to develop waste prevention practices.

Despite decades of political priorities, waste prevention is still an issue that appears to be unclear, uncertain, supposedly difficult, and almost in conflict with the organisation's interests. What appears to be a logical waste policy goal based on the waste hierarchy – that it is better to prevent waste than to have to manage it – seems odd when the everyday practices of waste producers are considered. It is part of the normality of businesses to combat the generation of waste, but only to the point where it is no longer organisationally, technically and economically justifiable.

The waste policy goals of preventing waste generation become stranded in the organisation's everyday practices. This explains why waste volumes continue to grow, despite decades of prioritisation. These goals do not recognise the fact that organisational members have adopted a mindset that views waste production as a normal outcome of organisational activities. While waste is seen as a problem for politicians – such as a risk to public health, an environmental hazard or the waste of resources – waste producers see industrial and commercial waste as something that is inevitable, rational and justifiable. The waste practices of the waste-producing organisations whose representatives we met were not orientated towards reducing waste production. In order to change the practices, a paradigm shift is needed that denormalises waste production and let waste prevention take precedence over, for example, profitability.

6. CONCLUSIONS AND SUGGESTED UNLOCKING MEASURES

Based on a qualitative interview-based study with representatives of Swedish organisations in the textile, construction and demolition, food and electronics sectors, this study has identified three barriers that aggravate waste prevention: a commercial approach that regards waste as an economic balance, an absence of reasonable technical solutions, and organisational habits and procedures that are not directed towards prevention. These barriers give rise to everyday social practices that are not geared towards the prevention of industrial and commercial waste: there is neither the technology, the competence nor the motivation for a social practice that prevents waste.

We find that the production of waste is seen as a normal practice: the norm, an inevitable consequence of the business' existence, even though efforts are being made to reduce waste as long as it is economically sound, organisationally acceptable, and technically feasible. Our study demonstrates what we call the *normality of waste*, and, in contradistinction, the oddity of waste prevention. Our results show a lack of businesses relevance of waste prevention. We therefore suggest that in order to make waste prevention objectives relevant, there is a need for financial motivation, knowledge of materials, and the competence to prevent industrial and commercial waste.

A final question is whether or not a transition to a circular economy creates the opportunity to work on waste prevention, as several of our respondents have speculated. The circular economy is portrayed by its advocates as a way of designing out waste (Ellen MacArthur Foundation 2017). However, firstly, this requires the circular economy to aim beyond an increase in the use of recycled materials and waste recycling, and start relating waste to consumption (e.g. European Parliament 2018). Secondly, there is a need to create an acceptance of and demand for circular innovations (Cainelli, D'Amato & Mazzanti, 2020). In practice, it would appear that getting suppliers, business partners and customers to assume the risk and perhaps also the cost of stopping material flows is a significant challenge, particularly for small businesses that only control a small part of their value chain, and which also lack the ability to change market standards on their own (see, Corvellec, Babri & Stål, Forthcoming 2020). Thirdly, the current linear solutions, with which circular innovations are usually compared, must also be made expensive and unattractive. Above all, the circular economy must not function as a way to divert attention away from the need to try to develop low-waste practices right now.

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PROPOSAL OF A TESTING PROGRAM FOR THE HP14 (ECOTOXIC) CLASSIFICATION OF AUTOMOTIVE SHREDDER RESIDUES (ASR) BY A BATTERY OF ECOTOXICOLOGICAL BIOASSAYS

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ABSTRACT
This manuscript describes a full procedure to be used in performing HP 14 classification of Automotive Shredder Residue (ASR). Sampling instructions and conditions for the transport and storage of ASR samples are included. Additionally, the steps to be followed in preparation of test portions (both solid samples and water extracts) prior to chemical characterization and subsequent ecotoxicological testing are defined. The established test battery includes all bioassays proposed in Pandard and Römbke (2013). Only aquatic bioassays are proposed as mandatory in this paper, which leaves the possibility of performing tests on terrestrial organisms based on the results of chemical characterization of the solid samples. Finally, the proposed set of concentration limits triggering HP 14 classification is fully consistent with thresholds described by Hennebert (2018a).

1. INTRODUCTION

Ecotoxicity is acknowledged as the most frequent property classifying wastes as hazardous (Hennebert et al, 2014). In Europe, wastes are classified for ecotoxicity (HP 14) according to the Regulation 2017/997/EC (European Parliament and European Council, 2017). The latter introduces a calculation method for the HP 14 classification, based solely on chemical concentrations of the hazardous substances occurring in the waste to be classified (i.e., classified with a Hazard Statement Code (HSC) H400, H410, H411, H412 and/or H413) according to the Regulation on Classification, Labeling and Packaging of Products and Substances 2008/1278/EC (European Parliament and European Council, 2008a). However, it does not include any specific guidance for performing the HP 14 classification through ecotoxicological testing. At the same time, it states that each Member State can adopt specific analytical protocols on conducting bioassays and the derived outcomes supersede results from chemical composition analyses.

The lack of a clearly established testing procedure can lead to erroneous waste classifications of those waste streams typically classified as mirror entries by the European Waste Catalogue (EWC), which must be declared as hazardous based on mandatory compliance assessment (European Commission, 2000).

For this reason, the objective of this paper is to propose a testing procedure allowing for a complete HP 14 classification of a classic example of a mirror entry, (i.e., Automotive Shredder Residues (ASR)), which is classified in the EWC with the couple of codes 19 10 03* (hazardous) and 19 10 04 (non hazardous). Therefore, ASR is here intended as the so-called "light fluff", which is the lighter fraction separated through air classification from the shredded hulk (Cossu et al., 2014). ASR is a highly heterogeneous waste stream, both in terms of granulometry and materials composition: it includes plastic, foam, textiles and metallic (magnetic, non-magnetic and PVC covered cables) particles, characterized by broad size distribution (Cossu and Lai, 2015). Significant environmental issues related to ASR management could arise due to the reported presence of trace elements, heavy metals and possible organic contaminants (i.e. PAHs, PCBs and mineral oil) (Cossu et al, 2014).

The choice of addressing HP 14 classification for this

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specific kind of waste was based on opportunity. However, the rationale behind this proposal can be applied to other mirror entries by taking care of fine tuning this procedure according to observable specific features. These specific features mainly relate to sampling and analytical procedures (e.g., particles size distribution and size reduction, extreme values of pH in the water extracts, and the need to investigate the presence of specific contaminant concentration).

This paper also includes detailed instructions about sampling, transport and storage of ASR laboratory samples. Furthermore, it describes stepwise procedures for liquid and solid test portion preparation, which will undergo further chemical analysis and ecotoxicological characterization. Also, the rationale behind the various requirements is explained. Elements of the whole procedure were not proposed from scratch but reference both to specific international and European technical standards as well as European ongoing regulations.

Regulation 2017/997/EC considers as the appropriate biotest procedures the ones which are consistent with the relevant methods established in the Regulation 2008/440/EC, pursuant to the Regulations 2006/1907/EC and 2008/1272/EC on products and substances (naming REACH and CLP, respectively), or "other internationally recognized test methods and guidelines" (European Parliament and European Council, 2006, 2008a, 2008b, 2017). In particular, the proposed ecotoxicological test battery includes all bioassays proposed in Pandard and Römbke, (2013). Therefore, the compliance with Regulation 2017/997/EC is assumed according to the fact that the chosen approach relies on technical standards referring to internationally acknowledged EN ISO series.

The CLP Regulation classifies products and substances (both liquid and solid) only for ecotoxicity in aquatic environment, thus requiring only chemical characterization on the water extracts or biotests performed on aquatic organisms. Although, chemical investigations on solid samples and bioassays carried out on terrestrial organisms could assess the presence of contaminants able to bind to soil particles and related exposure pathways, this behavior cannot be investigated solely by analyses on liquid samples (e.g. non-soluble or non-leachable substances). Given this background as well as considering the principle of the technical and economic feasibility as sanctioned by the Directive 2008/98/EC (European Parliament and European Council, 2008,b), the authors propose to include a chemical characterization of solid waste, which can trigger the need to assess related toxicity patterns through specific terrestrial ecotoxicity testing. In this context, ecotoxicological tests and chemical characterization are assumed to act not independently but synergistically for waste classification.

Lastly, the classification criteria of the results from bioassays were derived based on the concentration limits listed in Hennebert, (2018), which were proposed consistently with the EWC. The authors acknowledge that the EWC is not fully based on scientific evidence, as it is partly the result of political compromises and lobbying. Nevertheless, it is currently the only source which can be used as a reference for classification purposes.

2. METHODOLOGY

2.1 General Procedure

The general objective of the proposed testing program is to assess the classification of ASR as hazardous by HP 14. The consequent level of testing is the compliance testing level. The different phases of the proposed experimental protocol are outlined in the flowchart depicted in Figure 1. The stepwise procedure starts from the sampling phase, which is aimed at producing two laboratory samples that would be assessed by fractional composition analysis and chemical- ecotoxicological characterization, respectively. The obtained samples are then transported and stored in the laboratory where analyses will be performed. One laboratory sample undergoes Fractional Composition Analysis without any further preparation steps. Then, the solid test portion is prepared from the other laboratory sample to undergo i) chemical characterization and ii) leaching testing aimed at obtaining the relative water extracts. Chemical characterization and aquatic bioassays are then performed on the resulting aqueous test samples (i.e., water extracts). Terrestrial bioassays completed on the solid test samples are only executed if the results of the chemical characterization are not compliant with the proposed concentration limits for solid samples (outlined in Table 3). Finally, the ASR sample is classified for HP 14 based on the results of the comparison between the proposed concentration limits and the obtained results from the performed aquatic (Table 4) and terrestrial (Table 5) bioassays.

The rationale behind the requirements described for each phase of the proposed procedure is illustrated in the following sections.

2.2 Sampling, Transport and Storage

The proposed protocol defines a Sampling Plan consistent with the standards outlined in EN 14899:2015, UNI EN 10802:2013, and EN 15002:2015. Furthermore, instructions for Transport and Storage are consistent with EN 14735:2005.

If possible, it is recommended that sampling is performed by picking up the material dynamically from the conveyor belt placed downstream of the occurring aeraulic classifier. This approach is recommended to minimize spatial segregation due to the possible differential gravity settling occurring in the waste heap configuration. If dynamic sampling cannot be accomplished, sampling actions should be performed statically, either by stopping the conveyor belt or by carrying out sampling from temporary storage (heap configuration). Subsampling should be performed according to the "quartering and coning" procedure as explained in EN 15002:2015. Also, the so-called "long pile procedure" can be equivalently performed for the reduction of sample mass, according to EN ISO 14780:2017.

The calculation method used, and the parameters values assumed to determine the minimum sample mass of both primary sample and laboratory sample is described in Paragraph SM.1 of the Supplementary Material. Different values should be justified according to the results of further material investigations (e.g. bulk density) or granulometric analyses (e.g. definition of alternative D95) per-

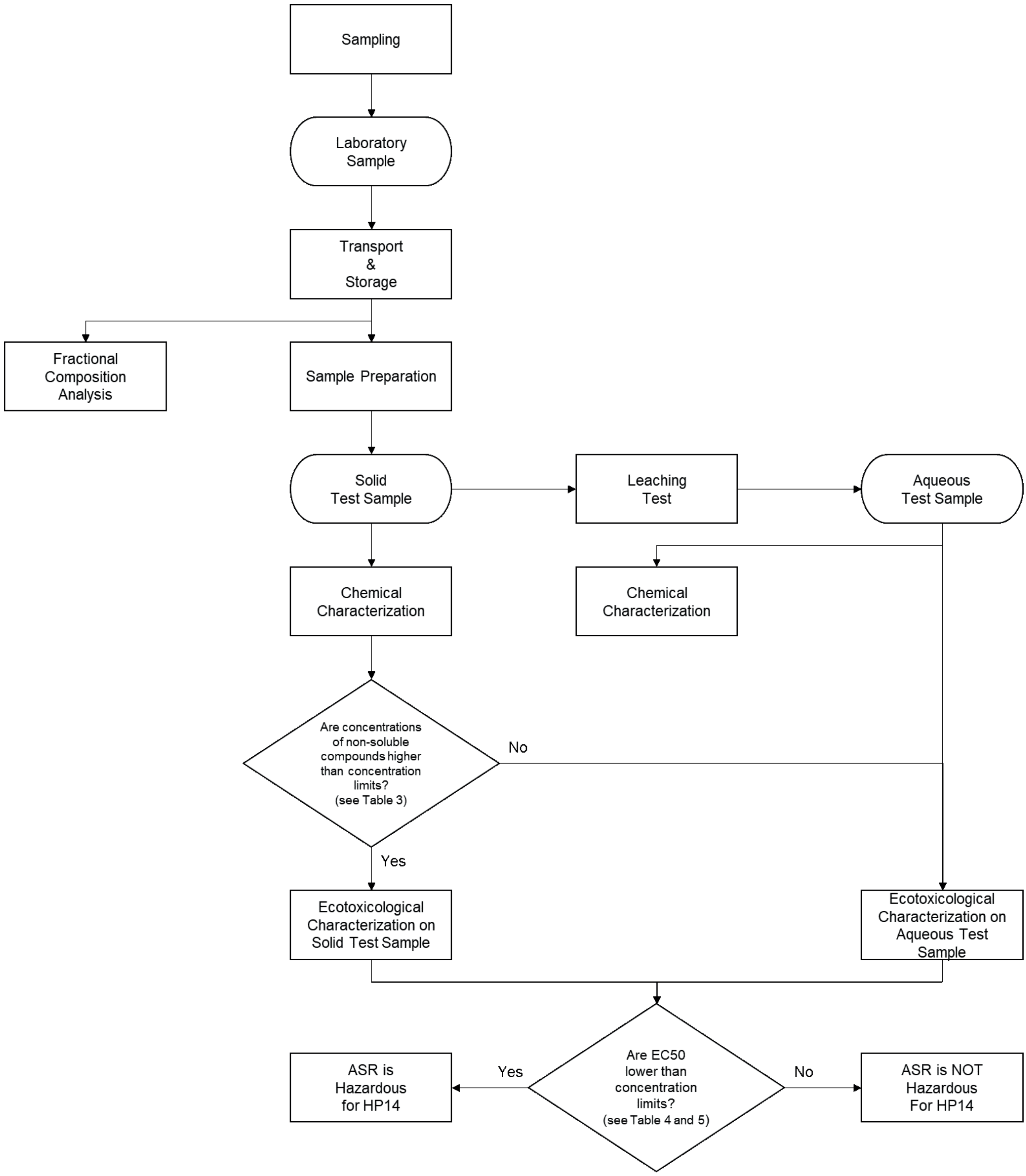

FIGURE 1: Stepwise procedure for HP14 classification of Automotive Shredder Residues (ASR).

formed on untreated sample (i.e., not size reduced). Here, ASR can be characterized by the presence of fines (0-20 mm) up to 70% (w/w) of the total sample, which can significantly lead to overestimation of the size of the laboratory sample (Cossu et al, 2014).

2.3 Fractional Composition Analysis

Fractional composition analysis is included in the procedure in order to determine the different material fractions making up the laboratory sample. Results of the fractional composition analysis are used to determine qualitatively the composition of the resulting laboratory sample and will not be used for any compliance assessments. This assumption allows us to perform such analysis on test portions which cannot be considered representative because of the suggested sample size.

2.4 Test Samples Preparation – Leaching Test

Instructions for the preparation of solid test samples are consistent with the principles reported in EN 15002:2015 for chemical analysis and EN 14735:2005 for terrestrial bioassays. The leaching test procedure is proposed in line with EN 12457-2 to derive water extracts for chemical analysis and aquatic bioassays.

While there is an agreement on the leaching procedure to derive water extracts for chemical characterization (i.e. EN 12457-2, here adopted), two main approaches are acknowledged for the preparation of waste eluates for ecotoxicological testing, both related to specific standards:

1. EN 14735 - Characterization of waste – Preparation of waste samples for ecotoxicity tests (EN 14735, 2005, currently under revision). Here, ASR can be considered as "granular waste". Consequently, specific water extracts must be derived following the "leaching test" procedure as laid out in EN 12457-2:2004;
2. OECD Document, nr. 23 and OECD Document, nr. 29 - Guidance on transformation/dissolution of metals and metal compounds in aqueous media. These standards are recalled by CLP Regulation (OECD, 2001, 2019).

Among the different requirements established by the cited standards (e.g. test duration, particle size), the factor most influencing the chemical profile of the obtained water extracts is the prescribed loading rate, 100 g/L for EN 12457-2 and 100 mg/L for OECD n.29. For the same amount of waste material, the consequent L/S ratios (10 L/kg vs. 10.000 L/kg for EN 12457-2 and OECD n.29, respectively) mimic long term release conditions which differ of several orders of magnitude in terms of years of occurred percolation or contact with pore water. Consequently, EN 12457-2 led to water extracts characterized by higher inorganic chemical concentrations (e.g. Heavy Metals) compared to OECD Documents nr. 23 and 29, when applied on the same ASR samples (Pivato et al., 2019). In this situation, a water extract should likely be considered more relevant to aquatic organism when derived applying the lower L/S ratio as practically feasible (Van der Sloot et al., 1997, Van der Sloot and Dijkstra, 2004). Besides, the requirements laid down in EN 12457-2 results in a more operative process to prepare a representative test portion (i.e. less efforts for both size reduction and subsampling) (Pivato et al., 2019). Finally, chemical and ecotoxicological characterization conducted on eluates identically derived allow an integrated assessment (Van der Sloot and Dijkstra, 2004). For these reasons, the experimental protocol being proposed, adopts the EN 12457-2 for liquid sample preparation prior to chemical characterization and ecotoxicity testing.

2.5 Chemical Characterization

Chemical characterization of both solid sample and waste eluates should be conducted in triplicate to assess the representativeness of the derived solid test portions after the subsampling process. In fact, the analyzed solid test portions can be considered representative if the coefficients of variation, calculated for each measured parameter as standard deviation divided for the mean of concentration, is low enough low to represent the lowest possible degree of variability, i.e. due to the performed analytical protocol. In this context, a CV of 0.1 can be taken as a reference value (CEN, 2006, Hennebert, 2019).

Results of the bioassays prevail on the assessment based on liquid concentration data for the sake of HP 14 classification (European Parliament and European Council, 2017). In this context, chemical characterization of ASR aqueous samples could allow to evaluate the leachable fraction of inorganic compounds (e.g. HMs), which can be considered responsible for the effects on the aquatic organisms. In fact, liquid concentration data can be used to calculate element-specific Toxic Units, by dividing measured values by the respective EC50 values as found in scientific literature or ecotoxicological repositories, to help interpret the outcomes from bioassays. The list of parameters which should be assessed on ASR eluates could be taken by what is prescribed by European Council, (2003).

Conversely, chemical characterization of solid test samples is needed to investigate the presence of non-soluble compounds likely occurring in ASR, i.e. Mineral Oil, PAHs and PCBs, whose effect in terms of toxicity cannot be assessed through aquatic bioassays (Cossu et al, 2014). It should be highlighted that ASR (as any other mirror entry) could be classified as hazardous for HP 7 and HP 11 in case concentrations of substances, classified with carcinogenic HSCs, are reported over specific thresholds established by specific regulations (European Commission, 2014). Therefore, the presence of one of the listed substances below the lowest thresholds specifically established by the regulations should trigger the need to perform the battery of terrestrial bioassays. In particular, the list of parameters which should be investigated in ASR solid test samples should include Mineral Oil (as Total Petroleum Hydrocarbons) i.e. sum of C<12 and C>12, PCBs and the list of PAHs required by the regulation (European Commission, 2014). This is reflected in the proposed protocol.

2.6 Ecotoxicological Characterization

The choice of the test battery has been determined following method proposed by Pandard and Römbke, 2013. Currently, there is consensus within the scientific community about this proposition, specifically when performed for fully comprehensive ecotoxicological characterization of waste (Moser and Römbke, 2009, Pandard and Römbke, 2013, Römbke, 2018). It should be assumed as a minimum set of assays, since it can be proved that the results from each ecotoxicological test will not be correlated with the other required tests, thus highlighting difference in the mechanisms of toxicity.

According to the principle of technical and economic feasibility (European Parliament and European Council, 2008b), each suggested bioassay should be performed on a unique test portion, constituted by the combination of three equal aliquots of the available test samples undergone chemical characterization (both solid and liquid).

The need to perform the set of terrestrial bioassays is triggered only if the derived concentration level of a defined list of substances in the solid test samples is not compliant with specific concentration limits proposed in national and international regulations.

2.7 Criteria for HP 14 Classification of ASR

The choice of concentration limits triggering the HP14 classification of the tested samples were established according to the proposition of Hennebert (2018a and 2018b). At the time of developing this approach, the proposed set of concentration limits were not applied in the national or international regulations. However, what is proposed in Hennebert, (2018) is currently the only available set of limits which can consistently be compared with the results of the proposed test battery, when performed according to the cited international guidelines (from EN ISO series, e.g. EN ISO 11348-3, EN ISO 8692 and EN ISO 6341).

3. PROPOSED PROCEDURE FOR HP 14 CLASSIFICATION OF ASR

3.1 ASR Sampling

- The technical goals of the sampling plan should be:
  1. the production of a laboratory sample representative of ASR regularly produced in a specific treatment plant considering the potential variability of input materials and produced waste in terms of both composition and granulometry;
  2. the production of a laboratory sample characterized by a sample size suitable, in terms of mass, to perform the planned analysis.
- To achieve the aforementioned technical goals, sampling of ASR is performed according to a probabilistic random sampling approach.
- The laboratory sample is obtained through a stepwise procedure:
  1. Primary sample (i.e. composite sample) production, and
  2. Quartering procedure.
- Two sampling techniques can be performed to obtain the primary sample:
  1. Sampling from temporary storage (see Paragraph 3.1.1);
  2. Sampling from a conveyor-belt (see Paragraph 3.1.2).
- The proposed methodology ensures that the laboratory samples are representative in terms of both quality and composition of each part of a monthly produced amount of ASR.
- A primary sample must be characterized by a total fresh mass of about 1.300 ± 50 kg.
- At the end of the quartering procedure, the laboratory sample which will undergo chemical and ecotoxicological characterization should be characterized by a total fresh mass of 30 ± 5 kg. The obtained laboratory sample should be characterized by a D95 ≤ 2 cm. Therefore, on-site size reduction (e.g. through portable shredding equipment) could be needed before performing the Quartering procedure.
- At the end of the quartering procedure, the laboratory sample which will undergo fractional composition analysis should be characterized by a total fresh mass of 60 ± 5 kg and should not undergo any size reduction for any reason.
- Different mass values for both primary and laboratory samples should be justified according to the results of further material investigations (e.g. bulk density) or granulometric analyses (e.g. definition of alternative D95) performed on untreated sample (i.e., not size reduction).
- Paragraph SM.2 of the Supplementary Material contains the minimum set of information that the sampling operator should complete and send to the laboratory with the collected sample.

3.1.1 Sampling from temporary storage

- A primary sample from a temporary storage heap must be performed monthly.
- The primary sample is a composite sample obtained by mixing together at least 20 increments, withdrawn from the ASR storage heap. Each increment must be characterized by a minimum increment weight of about 65 kg (i.e. 1.300 kg divided by 20 increments). It is preferred that increments are withdrawn with a mechanical bucket. Otherwise, increments should be collected using a shovel characterized by an opening size of at least 30 cm.
- Increments must be taken from the temporary ASR storage at different heap heights (see Figure 2 for examples of sampling locations based on heap subdivision). At least 3, 7, and 10 samples must be taken from the top, medium, and bottom layers of the heap, respectively. Additional increments could be obtained but care must be taken to maintain a ratio of 2:4:6 between increments withdrawn from the top, medium, and bottom layers, respectively. Each layer must be characterized by approximately the same height.
- The primary sample is then obtained after homogenizing (e.g., through mixing with shovels) the increments collected.

3.1.2 Sampling from conveyor-belt

- A primary sample from the conveyor belt must be collected monthly.
- The primary sample is a composite sample obtained by mixing together at least 20 increments, withdrawn from the conveyor belt during normal daily operations. Each increment must be characterized by a minimum size of about 65 kg (i.e. 1.300 kg divided by 20 increments).
- Each increment is sampled directly from the conveyor belt (during normal operation) preferably by use of cross stream sampler. Otherwise, a container can be used with enough of a capacity (i.e., characterized by an opening size of at least 30 cm), handled by a lifter, ensuring that a constant velocity is maintained through all the cross section of the conveyor belt.

3.1.3 Quartering procedure

- The production of the laboratory sample is achieved according to the so-called quartering procedure, as follows (Figure 3):
  I. Distribution of the primary sample of ASR on a cemented pavement and formation of a circular cake;

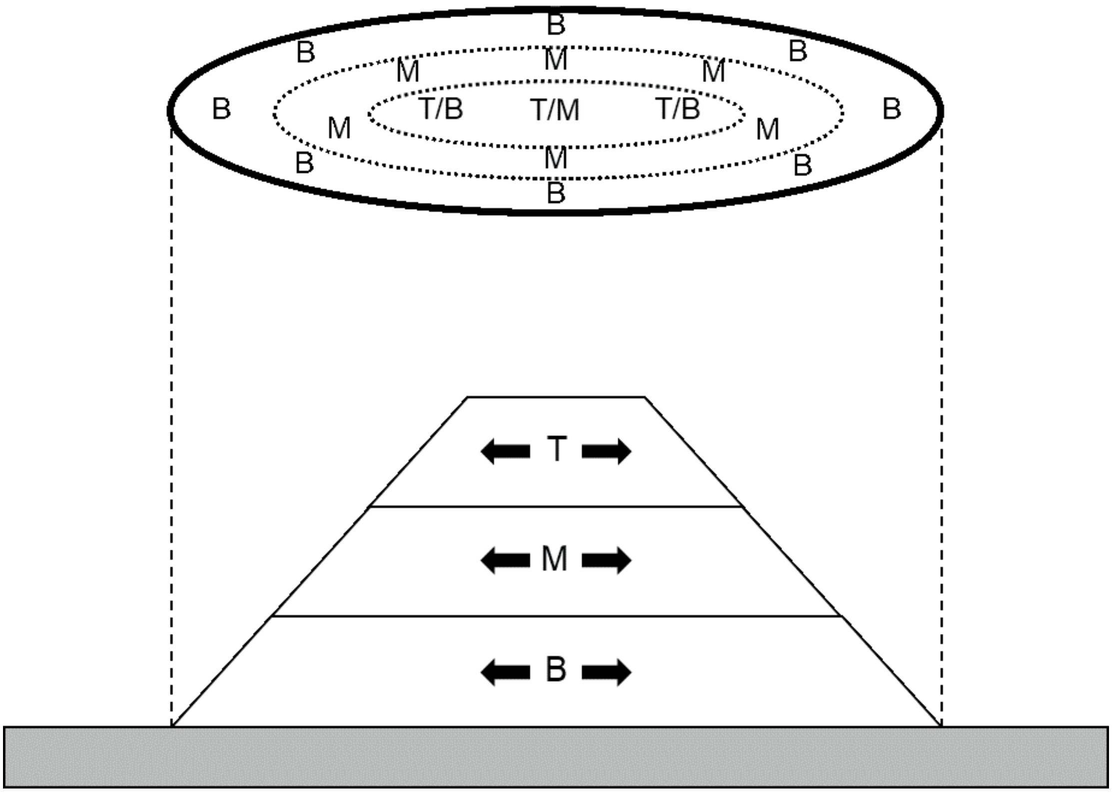

FIGURE 2: Schematization of ASR storage heap subdivision for increments withdrawal (T = Top layer, M = medium layer, B = Bottom layer).

II. Superficial subdivision with two perpendicular diametrical lines and complete removal of the material belonging to two opposite quadrants with a shovel and broom;
III. Mixing of the ASR from the remaining quadrants and formation of a new circular cake;
IV. Superficial subdivision with two orthogonal diameters, displaced by 45° with respect to the two previous tracers and complete removal of the material belonging to two opposite quadrants with a blade and broom;
V. Mixing of the remaining parts (i.e., about a quarter of the original one) and formation of a cake, ensuring to maintain approximately the same layer thickness while reducing the overall diameter;
VI. On the new circular cake, the same steps carried out previously on the initial cake are iterated enough times so that with the last quartering results in a sample characterized by a weight of 30 ± 5 kg, which is defined as the "laboratory sample".

- Instead of the quartering procedure as here described, the "long pile" procedure can be performed equally for the reduction of mass of the primary sample, according to EN 14780:2017.

3.2 Transport and storage

3.2.1 Transport

- The laboratory samples should be sent to the laboratory and stored in a resealable container that also ensures that the samples are kept in dark conditions.
- The container material shall be appropriate and not be a cause any type of contamination.
- Transport times of the laboratory samples should be as short as possible. Transport times shall be included in the overall storage time. A transport time of less than 12 h under refrigerated conditions (4 ± 2) °C shall be maintained in order to preserve the original properties of the laboratory samples and to avoid the migration of volatile fractions. If it is demonstrated that volatile compounds are not present in the sample, different transportation conditions can be permitted.

3.2.2 Storage

- Storage should be carried out in the same containers used for transport. Possible changes may be considered, and storage conditions shall be designed accordingly in order to limit the effects of such changes on the results of any tests. Any applicable changes must be reported in the final test report;
- Storage time starts from the collection of laboratory samples and ends with the start of definitive tests and should be as short as possible;
- A storage time of less than two months and at low temperature conditions (4 ± 2) °C shall be established in order to appropriately maintain the properties of the waste samples.

3.3 Fractional composition analysis

- Fractional composition analysis is performed according to the following stepwise procedure:

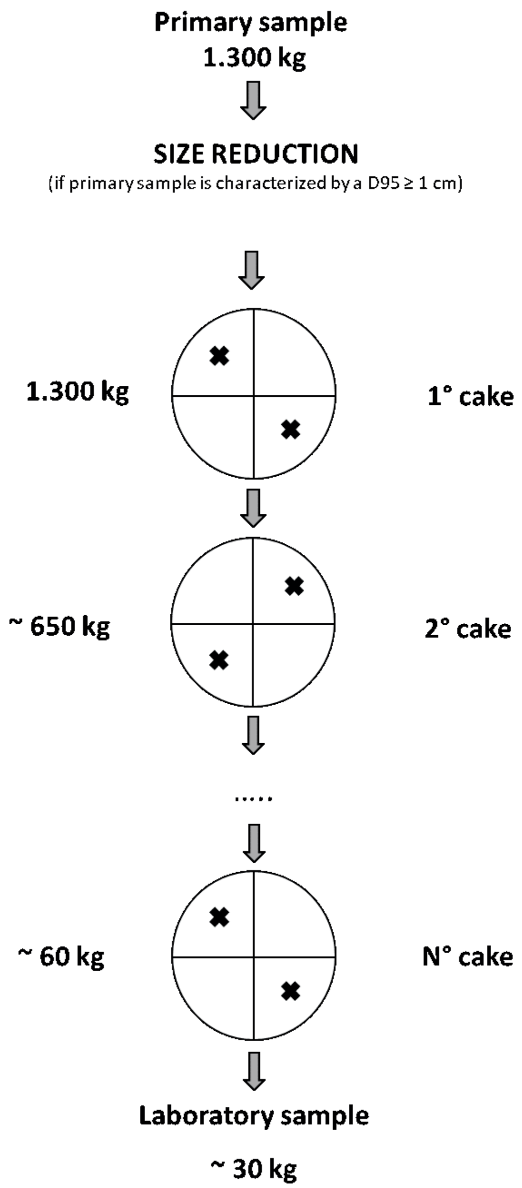

FIGURE 3: Quartering procedure for the preparation of the final laboratory sample (derived from EN 15002:2013).

I. Three test portions of about 20 kg are obtained through the quartering and coning procedure from the subsample derived from the previous step. No size reduction processes are performed on the 3 derived test portions;
II. Each obtained test portion is sieved through a 20 mm sieve;
III. The over-sieve fraction is divided into different categories (e.g., plastic foam and rubber, metals, cellulosic materials (wood and paper), textiles, electric cables, composites, and undersieve < 20 mm);
IV. Each fraction is weighed to determine the specific weight percentages;

- The analysis is performed in triplicate, (i.e., 3 test portions (in total 3*20 kg = 60 kg));
- An example of the report table to complete while performing the analysis and a exemplificatory figure for the graphical representation of the results are included in Paragraph SM.3 in the Supplementary Material.

3.4 Preparation of test samples for chemical and ecotoxicological analysis

- Chemical analyses are performed on the following samples:
  - ASR solid test samples (see Paragraph 3.5.1), and
  - ASR aqueous test samples which are derived from the leaching procedure (see Paragraph 3.5.2).
- Ecotoxicological Analyses are performed on a defined mixture of a dilution media (as required by each selected bioassay) and:
  - ASR solid test samples for terrestrial tests (see Paragraph 3.6.1), or
  - ASR aqueous test samples for aquatic tests, derived from a leaching procedure performed on the solid test portion of the ASR subsamples (see Paragraph 3.6.2).

3.4.1 Solid test sample preparation

- The preparation of the solid test sample is a stepwise process: each phase must be performed according to the following approach:
  I. A subsample of the laboratory sample, characterized by a weight of about 15 kg, must be obtained through a quartering and coning procedure.
  II. Non-crushable material occurring in the subsample characterized by a particle size > 4 mm (e.g. metallic parts such as nuts, bolts, scrap) that can possibly damage crushing equipment, must be selected and removed from the subsample before size reduction occurs. The particles withdrawn must be reported and classified according to the material composition and weight. The removed fraction should not exceed 10% (mass) of the total aliquot of the sample considered, otherwise the selected subsample must be discarded.
  III. At least 95% (on a weight base) of the ASR test sample must be characterized by a grain size of less than 4mm. If oversized (i.e. grain size > 4mm) materials exceed 5% (on a weight base), the oversized materials must be reduced in size with an appropriate crushing apparatus (e.g. shredder). Materials with a high particle size (e.g., plastic foam, long electric cables, bigger pieces of textiles) can be cut with other kinds of manual or mechanical devices prior to crushing, in order to avoid blockages of the crushing equipment.
  IV. Place the size-reduced subsamples within a container of adequate capacity and mix carefully with an appropriate tool (e.g. shovel, scoop, or trowel).
  V. The size of the prepared test samples must be determined according to the need of the following analytical determinations. Therefore, the needed test samples are:
     - Test sample (about 300 g TS) for aqueous test sample preparation through a leaching test (see Paragraph 7.2),
     - Test sample (about 300 g TS) for chemical cha-

racterization of ASR, and

- Test sample (the remaining amount of ASR test sample) for ecotoxicological characterization on terrestrial organisms.

3.4.2 Aqueous test sample preparation – Leaching test

Leaching test is performed in triplicate and according to the following stepwise procedure:

I. Place a solid ASR test portion characterized by a total dry mass of 90 (± 5) g TS (i.e., prepared according to Paragraph 9.1) in a glass container with a nominal volume of 2 L.

II Add distilled water to reach 900 ml of water including the moisture of the sample to ensure a liquid-solid ratio (L/S) of 10 L/kgTS ± 2%.

III. Agitate the glass container containing the ASR solid test portion and the distilled water using an end-over-end tumbler at 5-10 rpm at room temperature (15°C to 25°C) for 24 ± 0,5 h.

IV. Allow suspended solids to settle for 15-30 min and centrifuge the eluate for 30 min at 2500 g, whether or not incomplete separation of solid and liquid phases occurs.

V. Filter the obtained eluate through a 0,45 µm membrane filter using a vacuum or pressure filtration device. Rinsing the filter is not permitted after filtration.

VI. Measure and report conductivity in µS/cm (EN 10523:2012) and pH (ISO 10523:2008) of the resulting water extract immediately.

VII. Do not adjust the pH in any case during the leaching test procedure;

VIII. Take a subsample (e.g. ~1/3 L) from each of the water extract replicates to reconstitute the test sample to be used in ecotoxicological bioassays on aquatic organisms.

IX. The required volume (in ml) of the prepared test samples must be determined according to the

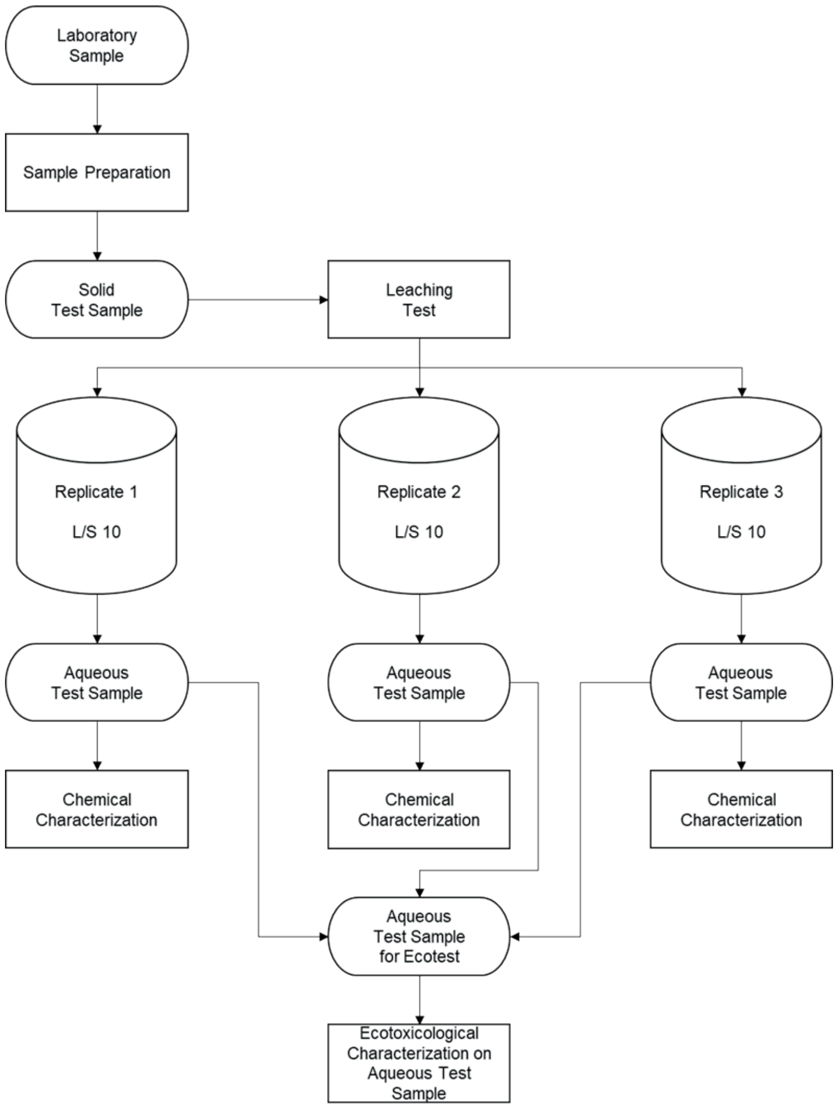

FIGURE 4: Detailed graphical procedure for deriving aqueous test samples through leaching tests for chemical and ecotoxicological analysis.

requirements of the following analytical determinations. Therefore, the necessary amount of test samples are:
    - Test sample (about 100 ml for each water extract replicate) for chemical characterization;
    - Test sample (about 400 ml) for ecotoxicological characterization of aqueous test sample reconstituted as described in step VIII.
- The proposed procedure is graphically described in Figure 4.
- Further details of the leaching procedure may be found in EN 12457-2 and EN 14735.

3.5 Chemical characterization

- Chemical Analysis are performed on:
    - ASR solid test samples, and
    - ASR aqueous test samples, (i.e., derived from a leaching procedure (see Paragraph 3.4.2)).

3.5.1 Chemical characterization of ASR solid test sample

Table 1 lists the chemical concentrations that must be measured in the prepared ASR solid test samples

Chemical analysis listed in Table 1 must be performed on 3 ASR test samples, where each one is characterized by an amount of 90 (± 5) g TS.

3.5.2 Chemical characterization of ASR aqueous test sample

- The aqueous test samples, derived from each replicate leaching test (see Paragraph 3.4.2), must undergo chemical characterization for the parameters established in Table 2;

3.6 Ecotoxicological characterization

- Compare concentration values obtained from solid test samples (see Paragraph 3.5.1) with the concentration ranges listed in Table 3. Whether at least one concentration values of non-soluble compounds is within the listed ranges, it will trigger the need to implement terrestrial bioassays. Otherwise, only aquatic test must be implemented.
- Results of ecotoxicological tests are expressed in terms of EC50 (%vol/vol and %w/w for aquatic and terrestrial tests respectively), which is the tested dilution of the test samples (with respect to the specific dilution media) generating 50% of the considered effect in the specific bioassay.

3.6.1 Bioassays on terrestrial organisms

- The test battery presented in Table 4 must be implemented on the ASR solid test samples prepared following the procedures described in Paragraph 3.4.1.
- The ranking of the sensitivity of the test battery listed in Table 4 is: Arthrobacter > Eisenia > Brassica.
- Each test is carried out following the instructions laid out in the specific standard listed in Table 4.

TABLE 1: List of parameters to be measured in solid test samples.

Analytes	Unit of measure	Standards
Total Solids	% Organics	EN 14346:2007
∑ TPH (C<12 + C>12)	mg/kg_{TS}	EN 14039:2005
Benzo(a)antracene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Benzo(b)fluorantene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Benzo(j)fluorantene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Benzo(k)fluorantene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Pyrene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Crisene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Dibenzo(a,h)antracene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
Naftalene	mg/kg_{TS}	EN 16181:2018 or EN 15527:2008
∑(PCB/PCT)	mg/kg_{TS}	EN 12766-1 and EN 12766-2

TABLE 2: List of parameters to be measured in aqueous test samples.

Analytes	Unit of measure	Standards
DOC	mg/L	EN 1484:1997
TDS	mg/L	EN 14346: 2007
Chlorides (as Cl^-)	mg/L	EN ISO 10304-1:2009
Fluorides (as Fl^-)	mg/L	EN ISO 10304-1:2009
Sulphates (as SO_4^-)	mg/L	EN ISO 10304-1:2009
Metals and metalloids		
Antimony	mg/L	EN ISO 11885:2009
Arsenic	mg/L	EN ISO 11969:1996
Barium	mg/L	EN ISO 11885:2009
Cadmium	mg/L	EN ISO 11885:2009
Chromium Total	mg/L	EN ISO 11885:2009
Copper	mg/L	EN ISO 11885:2009
Lead	mg/L	EN ISO 11885:2009
Mercury	mg/L	EN ISO 12486:2012
Molybdenum	mg/L	EN ISO 11885:2009
Nickel	mg/L	EN ISO 11885:2009
Selenium	mg/L	EN ISO 11885:2009
Zinc	mg/L	EN ISO 11885:2009

TABLE 3: Concentration ranges on ASR solid samples that trigger the need to perform terrestrial bioassay. a – derived from European Council, 2004 and further amendments. b – derived from European Parliament and European Council, 2019.

Substance	Unit of measure	Concentration range[a]
∑ TPH (C<12 + C>12)	mg/kg_{TS}	0.001 ≤ x < 1,000
Benzo(a)antracene	mg/kg_{TS}	0.001 ≤ x < 1,000
Benzo(b)fluorantene	mg/kg_{TS}	0.001 ≤ x < 1,000
Benzo(j)fluorantene	mg/kg_{TS}	0.001 ≤ x < 1,000
Benzo(k)fluorantene	mg/kg_{TS}	0.001 ≤ x < 1,000
Pyrene	mg/kg_{TS}	0.001 ≤ x < 1,000
Crisene	mg/kg_{TS}	0.001 ≤ x < 1,000
Dibenzo(a,h)antracene	mg/kg_{TS}	0.001 ≤ x < 1,000
Naftalene	mg/kg_{TS}	0.001 ≤ x < 1,000
∑(PCB/PCT)	mg/kg_{TS}	0.0001 ≤ x < 50[b]

3.6.2 Bioassays on aquatic organisms

- Test battery presented in Table 5 must be implemented on the ASR aqueous test samples prepared following procedures described in Paragraph 3.4.2.
- Ecotoxicological tests should be carried out within 72 h from the preparation of the aqueous test sample, which shall be stored in glass bottles with a minimal headspace at (4 ± 2) °C.
- No pH adjustments of the test sample shall be carried out. pH of the mixture should be measured immediately at the beginning and at the end of the test.
- The ranking of the sensitivity of the test battery listed in table 5 is: Pseudokirchneriella > Daphnia magna > Vibrio Fischeri.
- Each test is carried out following the specific standard listed in Table 5.

3.7 Classification criteria

- ASR is classified as HP 14 if at least one of the performed bioassays resulted in an EC50 not compliant (i.e. strictly lower) than the concentration limits listed in Tables 4 and 5 for terrestrial and aquatic tests, respectively.

4. CONCLUSIONS

This paper proposes an experimental procedure for the HP 14 classification of ASR through ecotoxicological testing. In addition, this manuscript includes the founding principles behind the proposition.

TABLE 4: Recommended ecotoxicological test battery on terrestrial organisms. Rounded limits are suggested according to Hennebert (2018b) based on thresholds proposed in Hennebert (2018a) for HP14 classification of waste. Concentration limits are reported as dilution % expressing dry mass of solid test sample on dry mass of specific dilution media, as required from the specific test.

Organism	Concentration limits for HP 14 classification	Standards
Soil bacteria (Arthrobacter globiformis)	EC50 < 5% (w/w)	ISO 18187:2016
Plants (Brassica rapa)	EC50 < 15% (w/w)	EN ISO 11269-2:2012
Soil invertebrates (Eisena fetida)	EC50 < 5% (w/w)	ISO 17512-1:2008

TABLE 5: Recommended ecotoxicological test battery on aquatic organisms. Rounded limits are suggested according to Hennebert (2018b) based on thresholds proposed in Hennebert (2018a) for HP14 classification of waste. Concentration limits are reported as dilution % expressing volume of aqueous test sample on volume of specific dilution media, as required from the specific test.

Organism	Concentration limits for HP 14 classification	Standards
Aquatic bacteria (Vibrio Fischeri)	EC50 < 15% (vol/vol)	EN ISO 11348-3
Algae (Pseudokirchneriella subcapita)	EC50 < 10% (vol/vol)	EN ISO 8692
Crustaceans (Daphnia magna)	EC50 < 10% (vol/vol)	EN ISO 6341

The document includes technical detailed instructions for each step of the procedure, from the production of the laboratory sample to ultimate classification based on the comparison of the results of a battery of bioassays with a proposed set of concentration limits. Furthermore, the chemical and physical characterization is required for both solid test samples and aqueous test samples. These samples are obtained through the performance of a leaching test. The proposed ecotoxicological tests include 3 aquatic and 3 terrestrial tests. However, the need to perform terrestrial tests is triggered when concentrations of several non-soluble and non-leachable contaminants in the solid test samples are recorded within proposed concentration ranges.

The compliance with European Regulation for HP 14 classification is ensured through the reference to international technical standards derived from the EN ISO series for each step of the proposed procedure.

Through this paper, the authors aimed to share with the scientific community the principles of this procedure and their technical application on a specific waste stream, while providing a basis for the development of sound scientific procedures for HP 14 classification by testing for other types of wastes, classified as mirror entries in the EWC.

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SEQUENTIAL EXTRACTION PROCEDURE: A VERSATILE TOOL FOR ENVIRONMENTAL RESEARCH

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ABSTRACT

The sequential extraction procedure as a tool to assess the environmental risk of metals in solid matrices has been widely studied. In this work, another promising application of these methods is proposed: the evaluation of the recoverability of critical raw materials from a solid matrix. To this aim, the normalized sequential extraction procedure BCR was applied to a contaminated soil from the south of Spain. In addition to this, the influence of the incomplete dissolution of carbonates contained in the soil on the fractionation results has been also studied. The high percentage of metal in the most mobile fractions suggested the potential use of the solid matrix as secondary source. The use of this approach together with environmental and economic feasibility studies would be an approach toward the circular economy.

1. INTRODUCTION

The use of Sequential Extraction Procedures (SEPs) to assess toxic elements in a wide range of sample types has rapidly increased since their introduction in 1979. Tessier et al. (1979) presented the first influential SEPs based on a five-stage extraction to fractionate several metals (Cd, Co, Cu, Fe, Pb, Mn, Ni and Zn) in river sediments. This analytical procedure consists of the partitioning of particle trace metal into five fractions: exchangeable, bound to carbonates, bound to Fe-Mn oxides, bound to organic matter, and residual using appropriate reagents. The extraction steps intended to simulate real changes in environmental conditions affecting metal binding in freshwater sediments, such as acidification by rainwater, reduction because of the post-depositional burial in a sediment column and oxidation after dredging and land-deposition of anoxic sediments. Although SEPs were originally described for sediments, they were soon applied to soil considering the similarities between soils and sediments (Sutherland, 2010). The increasing use of SEPs in a variety of samples at different experimental conditions made the standardization of the procedure necessary. Hence, the Commission of the European Communities developed the protocol known as Community Bureau of Reference (BCR) consisting in a three-stage procedure to split the total metal into three fractions: exchangeable, reducible and oxidizable (Rauret et al., 2000). In addition to the aforementioned fractions, it was recommended to carry out an acid digestion of the solid residue from oxidation step to obtain the total metal in the solid sample. Since the development of the standardized protocol, a wide variety of samples, such as soil, sediment, mine spoil, sewage sludge, compost, incinerator ashes and electric furnace dust, has been analyzed following this approach (Bacon & Davidson, 2008; Khadhar et al., 2020; Qureshi et al., 2020; Shehu et al., 2009).

It is widely accepted that the extensive use of SEPs entails wrong interpretation of results such as the association of metal fractions with specific mineral phases. Therefore, it is important to highlight that this procedure only divides the metal content into several fractions soluble in specific reagents under particular experimental conditions. Furthemore, some limitations of SEPs should be considered to properly interpret the experimental results. As a case in point, incomplete metal dissolution during the first step of the procedure for solid matrices with high concentration in carbonates entailing the overestimation of the next steps has been reported (Sulkowski & Hirner, 2006). This fact has been related to the non-complete recovery of carbonate-bound metals and the influence of the increased pH value on the partitioning of elements in subsequent steps. Although some studies have proposed different approaches to avoid these interferences, researchers usually follow the standard method as described in the guidelines.

Regarding applications, the SEPs are one of the most promising tool for the risk assessment and the feasibili-

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ty studies of the remediation techniques. The most used approach is the "before and after" application of the protocol to obtain information about the changes in the contaminant mobility because of the applied treatment (Reddy et al., 2001; Yuan & Weng, 2006). In some cases, these changes are even more important than the total amount of contaminant removed during the treatment (García-Rubio et al., 2011). Some studies reported the use of SEPs to predict the maximum amount of metal removable from the solid matrix (Subirés-Muñoz et al., 2011; Villen-Guzman et al., 2014, 2015). The evaluation of the time-dependence of contaminants in solid matrixes, named as "aging", has been also based on SEPs. In these studies, the incorporation of heavy metals from more mobile fractions into more stable fraction could be detected applying SEPs (Alexander, 2000; Jalali & Khanlari, 2008; Villen-Guzman et al., 2018).

In this work, the use of a SEP to evaluate the recovery of Critical Raw Materials (CRMs) from solid matrixes is proposed. On this matter, the European Commission (EC) launched a directive in 2008 aiming, among other goals, at the identification of CRMs. After the assessment of 41 raw materials, 14 materials were identified as critical in 2011 (European Commission, 2011). The list of CRM is regularly updated to consider the market and technological development (European Commission, 2014, 2017a). The contrasting of two parameters (Economic Importance and Supply Risk) with established threshold values allows the evaluation of the criticality: Economic Importance (EI) in terms of end-use applications and the value added of corresponding European Union (EU) manufacturing sectors; and Supply Risk (SR) based on the concentration of primary supply from raw materials producing countries, considering their governance performance and trade data. Therefore, CRMs are both of high economic importance for the EU, i.e. relevant for industry sectors that create added value and jobs, and vulnerable to supply disruption since the supply is associated with high risk of not being adequate to meet EU industry demand (Blengini et al., 2017). The use of BCR protocol to assess the recovery of metals from two soils with a high carbonate content is presented in this work.

2. EXPERIMENTAL

2.1 Soil collection

The soil samples (S1 and S2) were collected at 10–30 cm from two different places of the mining district of Linares, Spain (Figure 1). A detailed description of these soils has been previously presented (Villen-Guzman et al., 2018). In brief, the particle-size distribution indicated that soils S1 and S2 are classified as sand-loam and clay-loam soil, respectively, according to the International Soil Science Classification. Other relevant properties of these soils are a low organic matter content, an alkaline pH, a medium-low cation exchange capacity, a low humidity, a large carbonate content and a very low hydraulic conductivity.

2.1.1 Total metal concentration and carbonate content

The total content of metals was determined following the EPA method 3051A (Microwave assisted acid digestion of sediments, sludges, soils and oils): a solid sample of 100 mg was extracted with 6 mL of concentrated nitric acid and 9 mL of concentrated hydrochloric acid for 10 minutes using microwave digestion. This procedure was carried out in triplicate to assure reproducible results. The metal concentration of aqueous solution was determined by Atomic Absorption (Varian SpectraAA-110). The carbonate content was obtained by thermogravimetry (TGA) together with differential scanning calorimetry analysis (DCS) (TA SDT Q600 instrument).

2.1.2 Sequential extraction procedure

The BCR protocol consists of three sequential steps to obtain different fractions of each metal contained in the solid with different mobility (Rauret et al., 2000). A solid sample of 1 g is first treated with 40 mL of acetic acid solution 0.11 M by shaking for 16 h at 22°C to release the exchangeable and the acid-extractable metals known as the weak acid soluble (WAS) fraction.Then, the remaining solid phase is separated by centrifugation (Sigma 2-6) at 3000 g for 20 min. To obtain the reducible fraction (RED), 40 mL of solution of hydroxylamine hydrochloride 0.5 M is added to solubilize metals by shaking for 16 h at 22°C. The remaining solid phase is separated by centrifugation at 3000 g for 20 min. In the third sequential step, 10 mL of hydrogen peroxide acid-stabilized to pH 2-3 is added to the residue and it is digested at room temperature for 1 h with occasional manual shaking. Then, the digestion is carried out at 85°C in a water-bath until the final volume is reduced to less than 3 mL. After adding 10 mL of hydrogen peroxide, the same procedure is repeated until reducing the volume of liquid to about 1 mL. The residue is treated with 50 mL of

FIGURE 1: Sampling zone located at the south of Spain for soils: a) S1, b) S2.

ammonium acetate 1 M by shaking for 16 h at 22°C to obtain the oxidable fraction (OXI). The remaining solid phase is separated by centrifugation at 3000 g for 20 min. As a final point, an acid digestion of the solid sample following the methodology presented in 2.2.1 is performed to obtain the residual content (RES) of metals in the soil. A modified version of the BCR method was performed to evaluate the influence of carbonate content on metal extraction during the first step of the SEP. To that end, the first step was repeated for 11 times and the pH of solution after extraction was monitored. With the aim of assuring reproducible results, the procedure was performed in triplicate.

3. RESULTS AND DISCUSSION

3.1 Total metal concentration and carbonate content

Figure 2 shows the total concentration of the most representative metals for the soil samples S1 and S2. The total concentration of metals under study was significantly higher in the soil sample S2. The most relevant metal was Pb, which represents a serious environmental risk. The high concentration of Pb is due to the historical exploitation of galena ore (PbS) in this region. In addition to Pb, high concentrations of Fe and Ca, 37 g kg^{-1} and 34 g kg^{-1}, respectively for soil S2 were reported.

Regarding the carbonate content, it was 37.25 ± 8.00 and 85.00 ± 1.90 g kg^{-1} for S1 and S2 samples, respectively. As mentioned before, high concentration of carbonates could lead to incomplete metal dissolution during the acetic acid step, i.e. the first step of the BCR.

According to the European Commission, the metals contained in the soil to be assessed for its criticality were: Pb, Mn, Cu, Zn and Mg (European Commission, 2017c). The values of the two parameters analyzed to evaluate the criticality of raw materials SR and EI, for each metal are presented in Figure 3. The assessment methodology to calculate these parameters was revised in 2017 to include important improvements, such as: use of data from over the last 5 years instead of the last available year, introduction of data source priority for calculation and determination of the stage of material production (extraction or processing) with the highest supply risk for the EU (European Commission, 2017b). The SR represents the risk of a disruption in the EU supply of the material. This parameter is based on the concentration of primary supply from raw materials producing countries, considering their governance performance and trade aspects. It is calculated considering the stability of the producing countries, the extent to which a raw material could be substituted and the recycling input rate. The EI is associated with the potential consequences of an inadequate supply of the raw materials. This parameter is calculated by considering the fraction of each material associated with industrial mega sectors at EU level and their gross value added. This parameter is corrected by the substitution index related to technical and cost performance of the substitutes for individual applications (Ferro & Bonollo, 2019).

As observed, Mg is classed as critical since the metal exceeds the minimum SR and EI criticality thresholds which are 1 and 2.8, respectively. These results are associated with the lack of dolomite and refined Mg production in the EU (European Commission, 2017c). Mn could be defined as a borderline case with a value for the parameter EI exceeding the threshold and with a value of SR almost at the threshold. These results indicate that Mn should be carefully evaluated in a future EU assessment of CRMs. On the other hand, Pb, Zn and Cu exceeds the established threshold for EI. Nevertheless, the SR value, which is based on the concentration of primary supply from raw materials producing countries considering their governance performance and trade data, is far from the established limits. In other words, Pb, Zn and Cu are far from being considered CRMs for the European Union.

3.2 Sequential extraction procedure

Based on previously discussed results, the soil sample S2 was selected to perform further studies. The BCR results for Pb, Mn, Cu, Zn, Ca and Mg are presented in Figure 4. The metal obtained in the first step represents the most mobile fraction, namely Weak Acid Soluble (WAS); i.e. natural phenomena as rain can mobilize this metal fraction. A

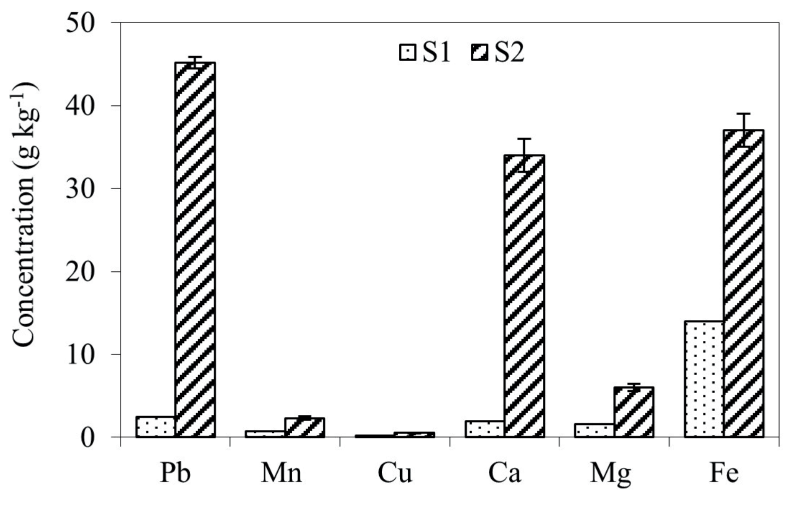

FIGURE 2: Total metal concentration in solid samples.

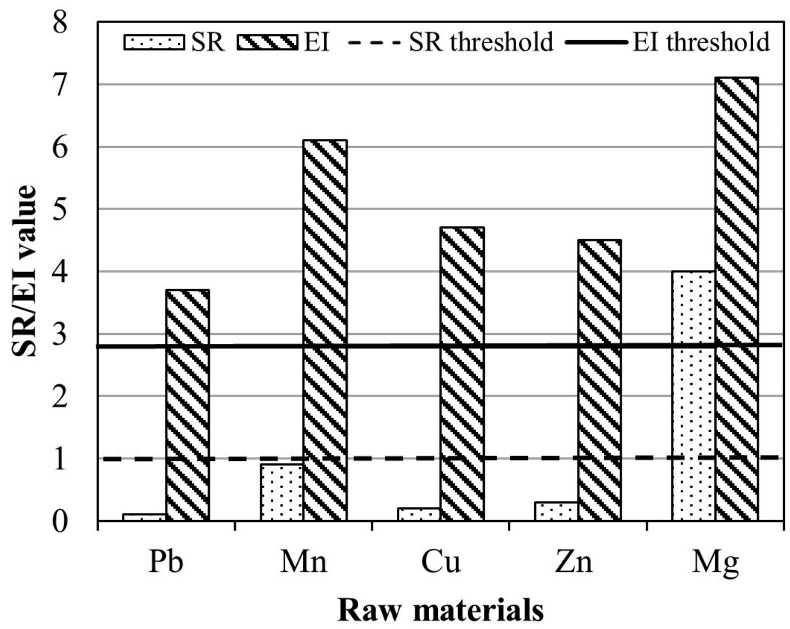

FIGURE 3: Criticality assessment results for Mn, Pb, Cu, Zn and Mg.

very high relative amount of Pb, 64%, is contained as WAS fraction. These results indicate a potential environmental risk and put forward the need for soil remediation. On the other hand, the high concentration of metal in the most mobile fraction could be used to identify the solid sample as secondary source to obtain the metal. That is to say, the S2 soil sample could be an ideal raw material to obtain Pb. The amount of Ca in the WAS fraction was almost 80% which has been associated with high concentrations of $CaCO_3$. The amount of Mg contained as WAS fraction was 23%. The reducible fraction (RED) is associated with the metal bounded to the Fe/Mn oxides. The most important relative amount of metal present in this fraction was found for Mn (58%) and Cu (45%). The metal attached to organic matter and sulphides is present in the oxidable (OXI) fraction. The relative amount of metals present in this fraction was small with the higher values obtained for Zn (16%) and Cu (11%). Finally, the residual (RES) fraction provides a comparison of the results obtained in each step with the total metal content obtained from acid digestion. Except for Mg, no important percentages of metal were found in the less mobile fraction. The metal extracted in the residual fraction does not represent a risk for the environment due to its unavailability. That is to say, the Mg contained in this fraction could be classified as inaccessible. According to the results, 3 g kg^{-1} of Mg is found in the most mobile fractions (WAS and RED). The viability of using this soil sample as secondary source of Mg should be evaluated through economic and

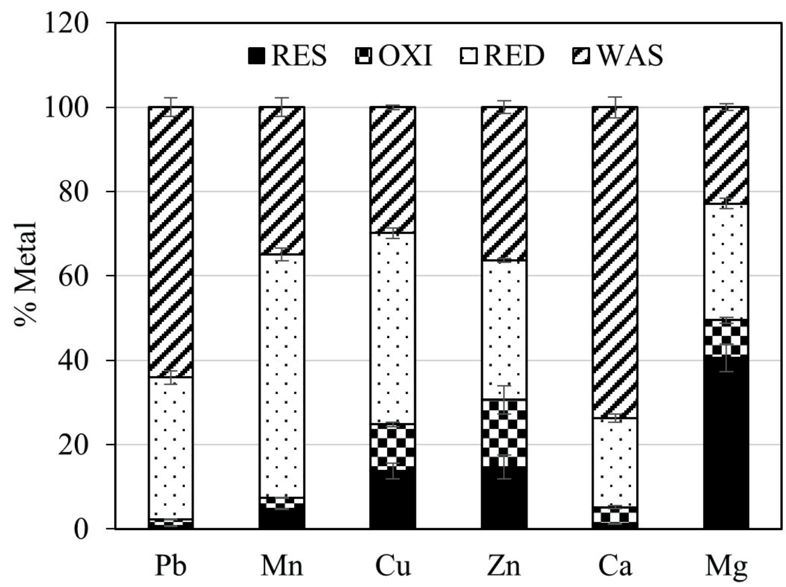

FIGURE 4: Sequential Extraction Procedure results of Pb, Mn, Cu and Zn for soil sample S2.

environmental feasibility studies. Specifically, the extraction of Mg from this soil should be compared with conventional processes of Mg production from dolomite ore, such as the Pidgeon process. At this point, according to Du et al. (2010), the Pidgeon process has an intensive energy usage and generates a large amount of GHG emissions. Hence, the use of secondary raw materials is highly recommended to avoid problems associated not only with the criticality of Mg but also with the negative environmental impacts of conventional extraction methods.

With the aim of evaluating the influence of the high content of carbonates on BCR results, the first step (WAS) was repeated for 11 times before progressing to the next step (RES). The concentration of Pb has been selected to discuss these results due to its high concentration. Figure 5 shows the percentage of Pb in each fraction together with the pH value of the solution after each extraction step.

As can be observed, the dissolution efficiency has been enhanced by repeating the first extraction step. Results indicated that the WAS fraction would be underestimated about 34% following the normalized method. The monitoring of the pH value could be a useful approach to decide if it is needed to repeat the first extraction step before progressing to the next. The influence of the solid properties on BCR results should be considered not only in risk assessment studies but also in CRM recoverability studies through sequential extraction procedure.

4. CONCLUSIONS

The recoverability of metals from solid wastes has been properly assessed through the determination of total metal concentration together with the application of a sequential extraction procedure. According to the results, high percentage of metals in the most mobile fractions (WAS and RED) indicated not only a serious threat to the environment but also the possibility of using the solid matrix as secondary source of CRMs. The most environmentally relevant metal present in the soil, Pb, has been previously categorized as economically important for the European Union. Therefore, the removal of Pb from the soil would entail not only the soil remediation but also the recovery of an important raw material. According to the EU, Mg is the most relevant metal contained in the soil with a view to recovery. The low mobilizable concentration of this metal makes necessary to evaluate the economic and environmental feasibility of the recovery processes. The use of primary source of Mg, such as dolomite ore, should be compared with the use of secondary source to promote circular economy. The procedure here presented offers promising results as a tool to be applied to different solid matrices and CRMs.

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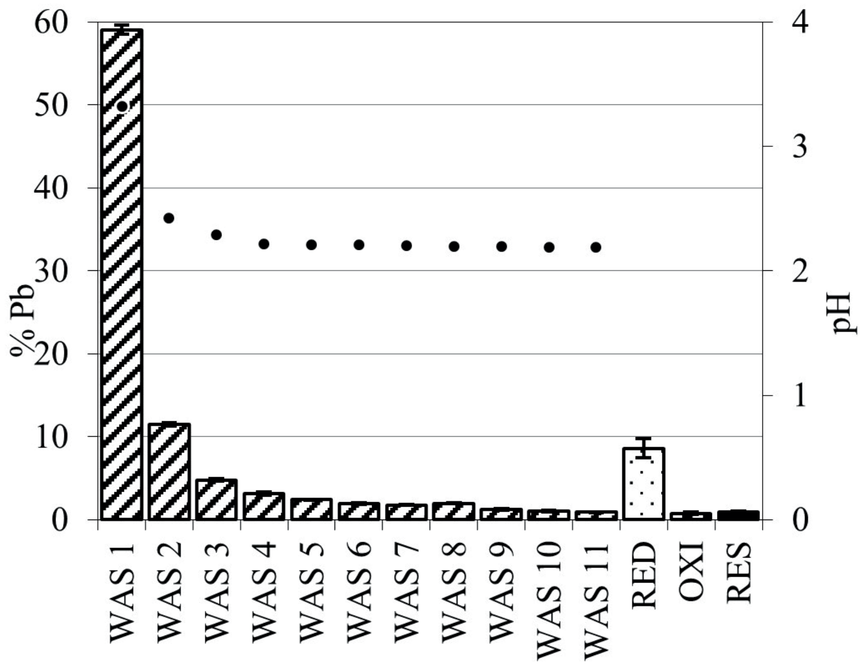

FIGURE 5: Sequential Extraction procedure of Pb for soil sample S2 repeating WAS step.

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INCLUSIVE PACKAGING RECYCLING SYSTEMS: IMPROVING SUSTAINABLE WASTE MANAGEMENT FOR A CIRCULAR ECONOMY

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Packaging

ABSTRACT

Innovative waste recycling methods have been developed in many countries by waste pickers (WP), which reduce overall recycling costs and expand recovered resources, providing income to a jobless population. The Brazilian experience in Extended Producer Responsibility for Packaging, implemented considering the WP as the main participant in the scheme, was investigated using the European P-EPR, the most consolidated experience in the world, as the benchmark. Quantitative and qualitative methods, including systematic literature review, were combined to discuss how the models could learn from each other to compose an inclusive P-EPR scheme, seeking to identify accessible solutions for the implementation of Integrated Sustainable WM in LMIC, taking into account their financial and governance constraints. Results showed that both systems are driving the recycling sector and increasing the efficiency of the WM, although neither has contributed to reducing the generation of waste. The BR scheme provided the recycling of different materials, but only the most valuable materials were recycled in the market-driven EU P-EPR. Mutual learning and networking between packaging producers and WP cooperatives in the BR P-EPR scheme improved the sustainability of the latter and knowledge of the recycling market for the former, in addition to improving the traceability of the informal sector's contribution to the recycling. An inclusive P-EPR scheme is suggested as a proposal for a more effective recovery of resources in many emerging countries, which can be crucial to achieve increasing plastic recycling targets agreed by many producers and to accomplish the ambitious EU's objectives of waste recovery.

1. INTRODUCTION

Waste is an important element of pollution of soil, groundwater and marine waters to which approximately 5% of global greenhouse gas (GHG) emissions are attributed. As a result of a linear economic model based on infinite growth, around 2 billion tons of solid waste are generated annually in the world and this generation is expected to increase by 70% until 2050 (Silpa et al. 2018). Waste recycling, as a part of Integrated Sustainable Waste Management (ISWM) (Wilson, Vellis and Rodic 2013), can reduce the scarcity of natural resources caused by the linear economy (Ellen Macarthur Foundation 2013) and the consequent negative environmental impacts of the growing global trash production. Expanding the extent and depth of waste recycling is a key to building a more circular economy (CE), contributing to the global climate change effort, at local and national scales (Ellen Macarthur Foundation 2013a). Moreover, implementing solid waste management in cities around the world is considered essential for meeting some of the United Nations' Sustainable Development Goals (Lenkiewicz 2016).

The realization of ISWM around the world, however, requires the identification of accessible solutions, especially in low and middle income countries (LMIC), where this policy is hardly implemented due to financial and governance restrictions (Hoornweg and Bhada- Tata 2012; Scheinberg, Wilson and Rodic 2010). In LMIC packaging represents about 20 to 30% of urban waste; in wealthy countries packaging accounts half of the urban waste generated (OECD 2001; OECD 2016). Therefore, reducing the amount of packaging waste that ends up in a landfill is a common target in many countries to increase the recovery of natural resources and save energy (Lifset, Atasu, Tojo 2013; Gupt and Sahay 2015). Reducing waste packaging has been identified also as a pivotal way of controlling marine plastic litter (Ellen Macarthur Foundation 2013a). Measures of Extended Responsibility to the Packaging Producer (P-EPR)

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have been introduced in many countries to achieve these goals since the 1990s (GiZ 2018; OECD 2016). About 100 P-EPR initiatives are in course in the world, most of them in richer countries but also in LMIC as Brazil, India, China and some African countries (IRR 2018, Demajorovic et al. 2014).

In many LMIC, packaging recycling is being carried out by waste pickers (WP) who have developed a low-cost and labour-intensive approach to waste collection and recycling services, which results in more waste recovery, reduces operational costs of waste management (WM) and turns waste into a means of poverty alleviation for a few million people (Wilson, Velis, Cheeseman 2006; Medina, 2007; Gutberlet, 2008; Scheinberg, Simpson, Gupt 2010; Sembiring & Nitivattananon 2010; Wilson, Rodic, Scheinberg 2012; Ezeah et al. 2013; Rutkowski and Rutkowski, 2015). Consequently, some tools e.g. InteRa (Velis et al, 2012) and value chain analysis (Jaligot et al, 2016) have been developed to promote the integration of WP at WM and many international agencies have recommended drawing lessons from WP initiatives to guide the development of WM policies(OECD, 2016; World Bank, 2008; UN-Habitat, 2010). However, these initiatives are poorly understood by WM policymakers and operators, who raise some negative aspects, real or perceived, associated with them (Gupt and Sahay 2015).

This paper aims to address the question of how these informal operators should be invited to work with, rather than against, the formal WM to reach ISWM and make cities more sustainable and inclusive. The Brazilian experience in P-EPR (BR P-EPR), where WP have been historically recognized as a player of the WM, is investigated using the EU experience in P-EPR (EU P-EPR), the first and the most consolidated experience in the world, as the benchmark. Quantitative and qualitative methods were combined to discuss how the two models could learn from each other to expand the effective recovery of MSW resources, improving the CE. An inclusive P-EPR scheme is proposed, which consists of the integration of the WP and their mode of operation in the WM in order to incorporate the respective economic, social and environmental benefits of that social technology developed by them. The advantages and challenges for the dissemination of this scheme to other LMICs are also highlighted.

2. MATERIAL AND METHODS

The comparative case study focused on the P-EPR because packaging is responsible for 30 to 50% of municipal waste, and EPR has been a policy implemented in many countries to address this problem (Hwang 2007; OECD 2016).

Initially, a systematic literature review was carried out to record the way in which P-EPR schemes were organized in European countries, as well as their main results. The scientific literature on EPR is extensive and most of them describe European experiences; however, no comprehensive systematic review was carried out to assess the cost-effectiveness of EPR for recycling waste, which was the main focus of this review. The review also covered gray literature. Due to the discussion of EU CE strategies that resulted in changes to the EU Packaging Directive and others, many reports on the EU P-EPR have been produced recently.

In addition, reports from world experiences to improve the WP working conditions and contribute to their inclusion in WM and P-EPR schemes were analyzed. This review aimed to complement the primary data collected on different participatory activities carried out by the researcher and her partners of ORIS - Observatory of Inclusive and Solidarity Recycling (Rutkowski et al 2017), in the action-research projects that they have been developing with Brazilian WP since 2012. Abstracts of 216 registrations were analyzed, resulting in 47 articles, reports, and theses read in full to complement the information in both models.

EU P-EPR data was also collected from participant observation at some professional conferences, seminars and lectures organized in 2019 to analyse how the changes for CE would impact the EU P-EPR schemes. To complete the understanding of the recycling model in Europe, study visits were made to Austria, where the P-EPR is considered effective and well implemented, and to Serbia and North Macedonia, where the WP' action is registered (Mrkajić 2018; Sapuric 2018). In these countries, field observation through technical visits to recycling facilities was carried out for a deeper understanding of packaging recycling processes in Europe, from a practical, operational, and commercial perspective. Data of the EU WP' operation and main achievements were also collected by participatory observation in visits to the places where they operates like scrap yards, dumpsites and others and by some semi-structured interviews with some key informants (Flick 2004).

Quantitative results related to packaging recycling rates in European countries were obtained on the EUROSTAT website. For the quantitative analysis of the BR P-EPR achievements, the annual productivity and other business data of a group of 277 Brazilian WP cooperatives, partners in the Brazilian P-EPR Agreement, were analysed. The cooperatives' data were accessed from a database formed by the National Association of the Brazilian WP (ANCAT), organized to compose a measure of the WP's participation in the packaging recycling index in Brazil. This is the first time the data has been used for research.

3. PACKAGING EXTENDED PRODUCER RESPONSIBILITY SYSTEMS

The concept of EPR is based on the polluter-pays environmental principle, which implies that manufacturers must be charged for the life cycle of their products, including the post-consumer stage. It should lead manufacturers to create take-back programs and support the reuse and recyclability of their products (OECD 2001). These policies generally have a target-oriented approach and not a command-and-control regulation. They can be implemented voluntarily by producers as a “product stewardship” scheme or be mandatory, which is currently the case for packaging in many countries (OECD 2016).

EPR schemes aim to increase the recycling rates of targeted products and materials and encourage innovation on Design-for-Environment (DfE) in producers (Michaellis

1995; Lindhqvist & Lifset 1998; Tojo, Lindhqvist and Davis 2001; OECD 2001). EPR leads to a change in responsibility for waste - from governments or municipalities and, therefore, from taxpayers to producers and distributors. In practice, this must mean that producers are responsible for the collection of used products and for the necessary subsequent processes like sorting and other treatments, for eventual recycling / reuse (EC-DGEnv 2012). In most cases, this responsibility is purely financial, with producers being obliged to finance, in whole or in part, the collection of recyclable waste previously under the responsibility of the municipalities with subsequent processes being left to the market. Sometimes they also have a responsibility to organize and carry out part or all these processes (EC-DGEnv 2014).

Four main categories of EPR instruments are recognized in different approaches and used for different products worldwide, as summarized in Table 1. They address specific aspects of WM and can be implemented simultaneously. The most widely used policy instruments in EPR are the several arrangements of take-back (72% globally), sometimes in combination with advance disposal rates, the next most used instrument (16%). Deposit / refund instruments (11%) are concentrated in the beverage container and lead-acid battery markets, sometimes in combination with take-back requirements. Other EPR policy instruments - combined upstream taxes / subsidies, recycling content standards and taxes on virgin materials - appear to be used infrequently (OECD, 2016).

Although EPR is, in theory, an individual obligation, producers often exercise their responsibility collectively. Most EPR schemes are operated by producers who organize or support one or more Producer Responsibility Organizations (PROs). These collective systems are said to generate economies of scale, simplify operations and reduce administrative burdens for consumers, retailers, and municipalities, reducing costs for participants (Gui et all 2016; Mayers and Butler 2013). PROs can act in competition with each other or hold the national monopoly and can also manage more than one type of waste and operate in more than one country (Mayers 2007). There is insufficient empirical evidence to determine if a PRO monopoly is more efficient than competing ones (OECD, 2016). Moreover, it is more difficult (and sometimes impossible) to obtain data on fees, costs, and revenues when several PROs are in competition, raising concerns about data availability and system reliability (EC-DGE 2014).

These PROs can be non-profit organizations, which is the most common model, government agencies, quasi-governmental non-profit organizations, or for-profit companies, but in all cases, they are overseen by public institutions. Most PROs charge a fee directly from producers, based on a specific fee structure, and the proceeds are used to pay the costs of collecting, sorting and treating waste, and also administrative and managerial costs (Mayers and Butler 2013). PROs are also responsible for other tasks, such as informing citizens and waste generators about selective collection; document and prove the quantities of waste collected and separated for recycling; bidding and hiring or supervise waste operators for collection and recycling (Mayers 2007), which can be WP co-ops (GiZ 2018; Tearfund et al 2019).

The minimum expected requirements for a P-EPR system must define (OECD 2016; GiZ 2018; EC-DGEnv 2014; OECD 2001): a) the range of products and producers involved in the system; b) the parties involved in the EPR system and the role and responsibilities of each one; c) the measurement methodology, reporting system and target control methods; and, d) ensure equal treatment and non-discrimination between all implicated actors.

Among the expected results of the EPR are a reduction in waste disposal and an increase in recycling and recovery (OECD 2001; Gupt & Sahay 2015); greater cooperation and involvement of the private sector in WM (Černiauskaitė 2013); achieve quantitative recycling and recovery targets (Cahill; Grimes & Wilson 2011; Da Cruz et al. 2014); boost the recycling industry and promote efficient secondary markets (Forslind, 2009; Hotta et al 2009; OECD 2001; OECD 2006).

EPR is also seen as a practical way to introduce "green supply chain management" to broaden the focus on resource efficiency (Massaruto 2014) and to help eliminate the cost burden for local governments on plastic waste (EASAC 2020). On the other hand, the impact of EPR on DfE and product innovation is less than expected (Atasu 2018; OECD 2016; Wiesmeth & Häckl 2011). Although a reduction in the material used in packaging has been reported (Hwang 2007; Gupt & Sahay 2015), innovation efforts have been more often directed towards classification and recycling techniques than towards product design (Lifset, Atasu &Tojo 2013; Walls 2006). Difficult recycling packaging, or even non-recyclable, is still widely used in many goods, even long after the introduction of EPR (EC-DGEnv 2014).

3.1 The European Extended Responsibility to the Packaging Producer Scheme

The EU Directive on packaging and packaging waste, which was approved in 1994 and updated in 2004, 2005, 2015 and 2018, aims to limit the generation and quantities of packaging waste landfilled by promoting recycling, reuse and other forms of waste recovery (PPWD 1994). This directive, coupled with others related to waste (Scharff 2018), requires all EU members to define a packaging waste management policy to achieve mandatory recycling and recovery rates, as shown in Table 2. As a result, most EU countries have implemented selective collection of packaging in different waste fractions (glass, plastic, metal, and paper) to meet legal recycling and landfill targets.

In response to these policies, EPR was implemented as an important instrument to support the European waste hierarchy (EC-DGEnv 2014), and an efficient waste management (EU 2018b). PROs were created in most EU countries after the first European packaging waste legislation implemented in Germany in 1990. Mayers (2007) registered 114 PROs for packaging waste organized in 29 European countries; P-EPR schemes have been implemented in 25 of the 28 EU members (EC-DGEnv 2014) and the levy on producer fees for packaging has been identified in all but the UK.

TABLE 1: EPR instruments, approaches and way of implementation used worldwide.

EPR Instruments	Usual approach	Way of implementation	Examples
Product take-back requirements	Recycling and collection targets defined for a product or material	Mandatory or voluntary	Most used scheme in the World, high transaction costs
Regulations and performance standards	Minimum recycled content on new products	Mandatory standards or applied by industries themselves through voluntary programmes	Some companies' decision: 100% recycled plastic bottles by 2025
Information-based instruments	Reporting requirements, labelling of products to communicate consumers about waste separation, and recyclers about raw materials in products	Mandatory standards or applied by industries themselves through voluntary programmes	Recyclability symbols in packaging
Economic and market-based instruments	Deposit-refund (DRS)	Previous deposit is fully or partially refunded when the product is returned to a specified location	Bottles deposit machine in retailers
	Advanced Disposal Fees (ADF)	Fees at purchase based on the estimated costs of collection and treatment, that may be collected by public or private entities and used to finance post-consumer treatment of some products	Used in 17% of the schemes in the World (PROs)
	Material taxes	Taxes on virgin materials or difficult to recycle materials to create incentives to use secondary (recycled) or less toxic materials	Appropriated for shifting innovation in design
	Upstream combination tax/subsidy (UCTS)	Tax paid by producers subsequently used to subsidise waste treatment	Associated to DfE, but less used

Source: Elaborated by the author from OECD 2001; 2016

TABLE 2: Recycle and reuse targets for packaging regarding the current EU Directives - Packaging and Packaging Waste Directive (PPWD) and Waste Framework Directive (WFD) updated with Circular Economy Package.

		Waste classification									
		Municipal waste		Packaging waste (3)		Packaging material					
		Landfill (max.)	Recycling/ reuse (min.)	Recycling (min.)	Recovered or WtE (min.)	Paper/ Cardboard	Plastic	Glass	Ferrous metal	Aluminum	Wood
Waste legislation	PPWD 2008			55%	60%	60%	22,5%	60%	50%	50%	15%
	PPWD 2025			65%	(2)	75%	50%	70%	70%	50%	25%
	PPWD 2030			70%	(2)	85%	55%	75%	80%	60%	30%
	WFD 2020	(1)	50%								
	WFD 2025		55%								
	WFD 2030		60%								
	WFD 2035	10%	65%								

(1) Landfill ban on separately collected plastic, metal, glass, paper, cardboard and biowaste. Target to be redefined after 2024. (2) Incinerated packaging can´t anymore be counted in the recycling/ recovered targets (3) Equal to the amount of packaging placed on the market. Source: Elaborated by the author from Scharff 2018; WFD 1998; PPWD 1994; EU 2018; EU 2018a; EU 2018b.

These schemes essentially oblige packaging producers to financially support to varying degrees the implementation of packaging waste recycling (EC-DGEnv 2012).

Some common characteristics can be observed in the EU EPR: 1. Normally, one operator organizes the system for several companies; 2. National fees based on material-ton/packing to finance cost of selective collection-waste sorting are paid by producers and importers; 3. Collected waste are sold to independent actors who classify and sell it to recyclers or incinerators; 4. Revenues from sales of secondary material help offset the financial contributions of producers and importers to EPR schemes. Figure 1 schematically represents this model.

Producers generally join national collective compliance schemes, organizing producer responsibility organizations (PROs) to be the operator of EPR requirements. PROs differ mainly in terms of organizational structure, specific operations, costs and reporting requirements, but they all serve the same basic function. The recycling, logistics and waste companies hired by the PROs carry out daily operations to

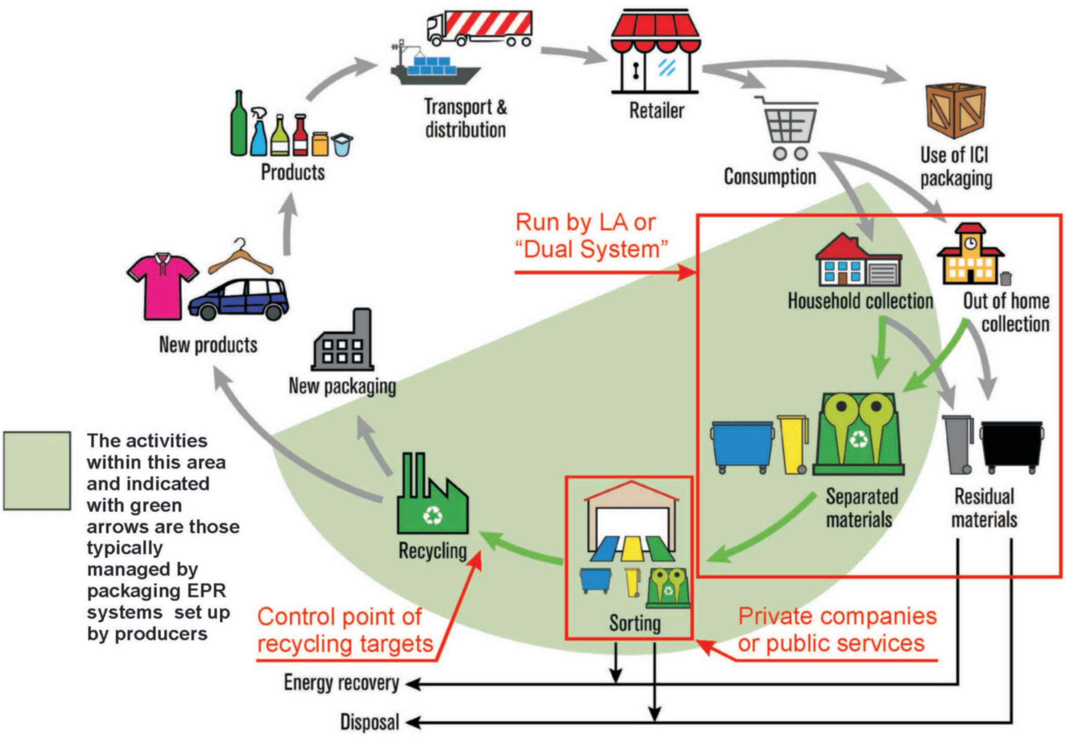


FIGURE 1: EU P-EPR operational model (LA = Local Authorities) Source: adapted from EUROPEN, 2014.

ensure that packaging is collected at designated municipal collection points and treated as needed. PROs recover their costs from producers either through flat rates or by allocating actual costs according to the relative quantity of products each producer sells over a period - usually semi-annually or annually. Most of the time PRO collection services are provided by public WM services, run by local authorities (LA) and sorting services are provided by recycling companies. However, in some cases, there is a "dual system" when PROs and municipalities manage separate collection systems. In others, the LA act also as providers of sorting waste packaging for the PROs, directly or by subcontractors.

The EPR system is considered a factor that boosted the recycling industry in Europe and is also considered important for financing selective collection of waste which, for operational reasons, is generally more expensive than a conventional collection (EC-DGEnv 2014). It is considered also pivotal for achieving the EU packaging waste recycling targets (EU 2018b). The role of PROs has become important in its implementation since they provide an interface for organizing financial transactions, waste collections, and communications between governments, producers, waste treatment companies, retailers, and more than 80,000 municipalities in Europe.

The packaging recycling indices are calculated annually by comparing the total weight of packaging waste entering recycling operations with the quantity of packaging placed on the market in each Member State (MS) and for each packaging raw material. In an attempt to achieve the principles of the CE, increasing recycling targets were recently defined for Municipal and packaging waste as well as for all packaging raw material but aluminum(Table 2).

The EPR scheme operation has common features in EU MS, but small differences can also be observed. For each operation model and different geographic / landscape conditions, the measured / estimated cost for the full cost of collection, classification and treatment of selectively collected packaging waste may be different as well as the fees. This cost may also include costs for handling packaging-containing commingled waste; for public information and awareness to ensure consumer participation in the scheme; for litter prevention and management; and costs related to the enforcement and surveillance of the EPR system, which includes audits and measures against "free-riders" (EC-DGEnv 2012).

In the Austrian EPR system, for instance, fees must cover the costs of collecting packaging in commingled waste, while in the UK, they cover only 10% of the total cost for packaging waste collection and recycling (EC-DGEnv 2014). In Germany, PRO bears the total cost to collect packaging due to the choice of a "dual" system; in the Netherlands, the full cost is covered in the WM bill, while most countries have adopted a cost-sharing mechanism between producers and WM public services (Massaruto 2014). In Belgium, LA are reimbursed for a defined frequency and density of collection and in France, P-EPR is supposed to cover 80% of a 'net optimised costs' system, the reference costs are based on the optimal functioning of the collection and sorting operations (EC-DGEnv 2014). Da Cruz et al (2014) and Da Cruz, Simões and Marques (2014) failed to evidence that the industry bears all the costs of managing and recycling packaging waste in any of the EU countries analysed.

In addition, as shown in Figure 2, there seems to be no

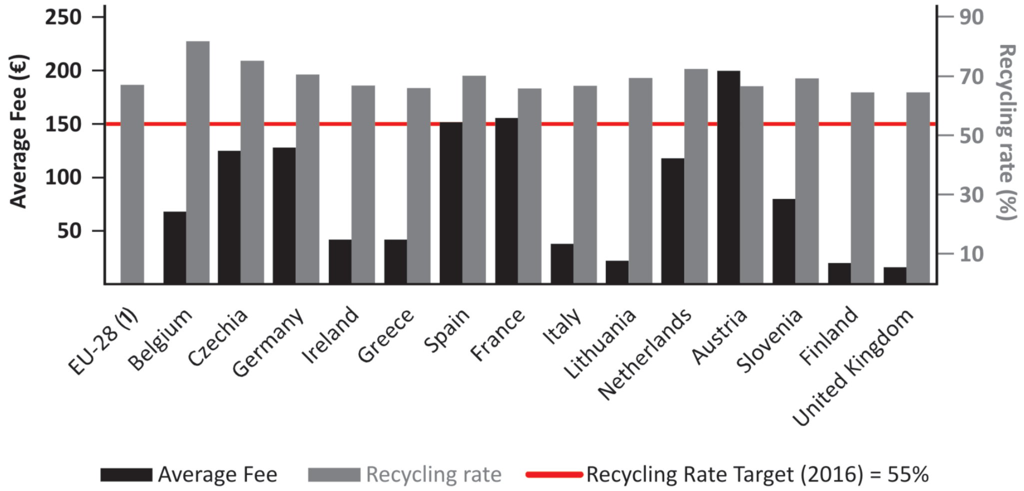


FIGURE 2: Recycling rates (%) of paper, plastic, metal and glass household packaging in some EU countries (right axis) and average fees charged to packaging producers (€) per ton of packaging put on the market(left axis), based on the average share of paper, plastic and glass in total waste packaging (domestic and industrial /commercial) in the EU-28, year of 2016. EU-28 (1) represents the average recycling rate for EU-28 countries. Source: Elaborated by the author from EC-DGEnv 2014; EUROSTAT, 2019.

direct correlation between the fees paid and the recycling rates obtained. It is also difficult to compare the cost-effectiveness due to the lack of transparency regarding the financial aspects and the technical performance of the EPR schemes. The amount of waste generated, collected and treated is hardly comparable between MS, because they are calculated in different ways, with quality issues; costs are not always aggregated on a national scale; data on quantities of packaging placed on the market, financial information and flows are not always available, as well as information on fees paid by producers and the costs they must cover (EC-DGEnv 2014).

There are also different scopes in the P-EPR between EU countries - some cover only household packaging waste, while in other countries commercial and industrial packaging waste can also count towards recycling targets (Cahill, Grimes and Wilson 2011). Commercial and industrial packaging waste is known to be easier to collect and sort (Massaruto 2014; EEA 2005), which reinforces criticisms about the accuracy of comparing recycling rates in EU MS (Hogg et al 2018). Concentrating efforts on commercial and industrial waste also raises concerns, since it is in the domestic flow that most of the packaging recycling challenges lie (Lerpeniere & Cook 2018). Other environmental criticisms arise because fees are not based on principles of ecological modulation. Although the rates are different for different raw materials, the fee diversification does not consider detailed information on the recycling of subfractions. For example, there is a single fee for all kind of plastics, although eco-modulated P-EPR rates should penalize the use of plastics that are difficult to recycle (Brouillat & Oltra 2012).

3.2 Brazilian Packaging Extended Producer Responsibility: an inclusive scheme

Brazil, Argentina, Ecuador, Colombia and several countries in Latin America and the Caribbean (LAC), including the OECD´s members Chile and Mexico, recently took steps to implement their first P-EPR systems. In these countries and in other LMIC countries, the recycling of waste depends on the action of the WP, which is the main and most visible part of the so-called Informal Recycling Sector (IRS).

In Brazil, the recognition of waste pickers as workers "recyclable material pickers" was defined by law in 2002 and, since then, some public policies have been developed to integrate the WP into the formal economy. The Brazilian Solid Waste Policy (PNRS in Portuguese), approved in 2010, defines a hierarchy for the treatment of waste, recognizes waste as a resource of economic and social value and ratifies WP cooperatives (WP co-ops), the main representative of the IRS in Brazil, as an important player of the urban waste recycling process.

As usual in other countries, the selective collection of domestic solid waste in Brazil is part of the public sanitation and urban cleaning services, under the responsibility of LA. LA can hire private service operators to provide these services, which can be a company or a WP co-op, but only WP co-ops can be contracted without public bidding.

The PNRS (2010) prohibits the landfilling of recyclable waste and defines producer compliance for some hazardous waste, including packaging. The packaging life cycle is under “shared responsibility”: LA must offer selective recyclable collection as part of municipal WM and citizens must dispose of recyclable materials separately from other waste. Citizens can also be charged with waste fees. Manufacturers, importers, distributors, and traders must encourage and maximize reuse and recycling, improving the design of their products and manufacturing processes. They also need to comply the diversion of packaging to landfills, through an agreement with the states and federal governments. This "reverse logistics" system, as it was called in the legislation, should be implemented in partnership with the WP co-ops.

Although urban waste collection is carried out in almost 100% of Brazilian municipalities, selective waste collection is poorly implemented despite the PNRS. Less than 40% of municipalities have implemented this service, which reaches all households in less than 10% of them. This results that only 4.1% of the 62.8 million tons of waste generated annually in Brazil - 2.6 million tons / year which represents around 14% of the total recyclable waste- is collected selectively (SNIS 2018). However, industries in Brazil report significant recycling rates for several recyclable waste (Rutkowski & Rutkowski 2017) which are growing each year (IBGE 2015). The PET recycling rate in Brazil, for example, is reported to be higher than in the USA and many EU countries; according to the industry, 83% of recycled PET is supplied by WP co-ops (ABIPET 2016; ABIPET 2019). In most Brazilian cities, selective collection of waste is provided or depends on the WP co-ops' initiative, without the support of LA.

The Brazilian waste recycling supply chain can be described as consisting of two parts (Figure 3) (Rutkowski 2008). The most visible component is a "value chain", in which recyclable material becomes the raw material for paper, plastic, etc production chain. This chain depends on a "service chain" needed to transform mixed waste materials into a resource (Scheinberg&Simpson 2015). WP co-ops play a valuable role in these two chains and are at the intersection between them (Rutkowski&Rutkowski 2017). This way of action, known in Brazil as 'Solidarity Selective Collection' (CSS in Portuguese initials), has been observed also in other LMIC (Ferronato et al 2019; Giovannini &Huybrechts 2017; Batista et al 2018; Scheinberg et al. 2016; Nahman 2010).

CSS is a "social" technology built from the WP's practical knowledge and skills, organized in a Social and Solidarity Economy framework (Rutkowski & Rutkowski 2015; Lima et al 201; Gutberlet 2009). Through CSS, WP co-ops divert household packaging from landfills for recycling. Door-to-door CSS expands the selective packaging collection route, increases the volumes collected, reduces operating costs and reduces GHG emissions (King & Gutberlet 2013). WP co-ops also raise people's awareness to carry on better separation of recyclables through recycling education, which includes letting them know that the material will be turned into income for WP families (Rutkowski & Rutkowski 2015). In WP co-ops´ sheds, packaging is manually classified into more than 20 different subcategories, using visual and tactile sorting skills and adjusted to the quality requirements of secondary material buyers, which implies a substantial and sophisticated contribution to waste recycling (Purshouse et al 2017).

In some large cities, private cleaning companies contracted by LA are responsible for collecting recyclable waste from door to door (SNIS 2018), and WP co-ops are

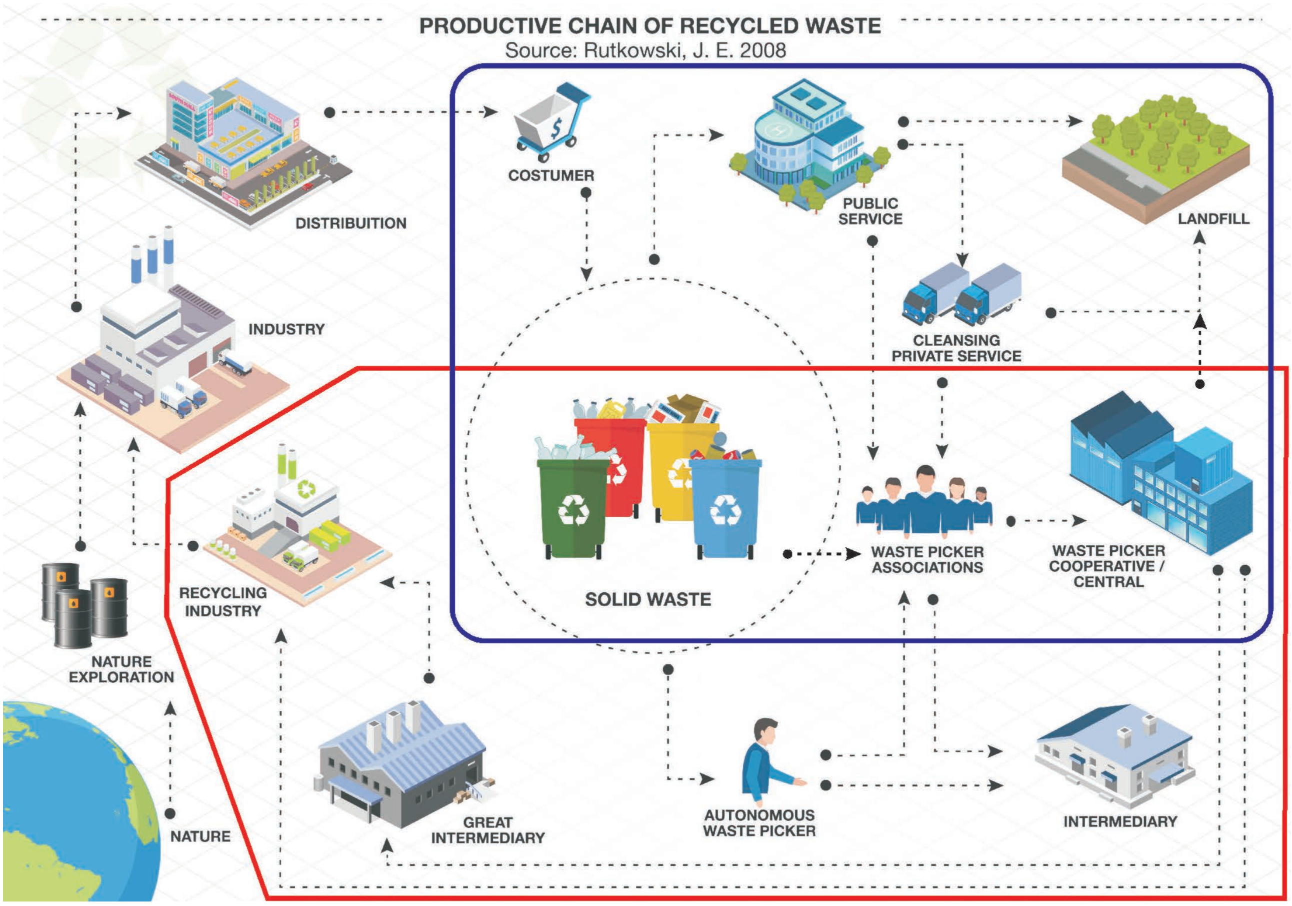


FIGURE 3: Waste recycling supply chain in Brazil. Blue line marks the service waste management chain and the red one marks the value recycling chain. They are interconnected by the WP co-ops that play paramount roles in both. Source: Rutkowski 2008; Rutkowski&Rutkowski 2017.

responsible for the sorting, packaging, storage, marketing of the material, acting as non-automated Material Recycling Facilities (MRF) (Purshouse et al. 2017).The packaging can also be recovered, even if they have been discarded by householders without prior separation. In these cases, the autonomous and independent "street WP" picks them in trash bins and, using their carts or big bags, takes them to WP co-ops´ sheds or to scrap yards. In many cities, a smartphone app offers a connection service between residents and the street WP for donations of recyclable materials. Scrap yards act as intermediary businesses in the recycling value chain, collecting recyclable materials also from commercial and industrial organizations, sometimes buying these materials, sometimes receiving them as donations in exchange for offering a waste correct disposal (Rutkowski &Rutkowski 2017).

BR P-EPR schemes were organized considering this context and recycling actors, as shown in Figure 4 (CEMPRE, 2018). LA is not directly involved in the system, although in many cities WP co-ops work in partnership with them. Producers provide financial support to WP co-ops, which report them data related to recyclable materials marketed to compose recycling targets. Packaging producers also implement a "dual" system to receive voluntary delivery of recyclable materials (PEVs in Portuguese initials) and to promote information to improve citizens willingness to recycle. The Brazilian Government's Environmental System is responsible for controlling targets achievement.

The relationship between some large packaging companies and WP co-ops has been experienced at LAC since the first decade of the 2000s (IRR 2018; Fernandes 2016). These companies have offered support to WP co-ops through social-environmental responsibility actions. This model has been replicated in different countries (DANONE ECOSYSTEM 2016). The BR P-EPR scheme, implemented in 2015 by an agreement between a coalition formed by about 4000 companies from 22 different business sectors and the BR government, was based on these experiences and contributed to formalize the model (Demajorovic & Massote 2017; IRR 2018a). The goal was to reduce packaging waste disposal in landfills by 22% and increase the recovery of dry waste fraction by 20% compared to the situation in 2013. The target for landfill diversion must be 45% by 2031. The agreement also aims to improve the capacity of WP co-ops, increasing the efficiency and productivity of the recycling sector (CEMPRE 2018).

In BR P-EPR experiences, companies finance non-governmental organizations (NGOs) to offer technical assistance and invest resources in the infrastructure of WP co-ops, such as building retrofits, maintenance and purchases of machines and trucks. In return, WP co-ops provide information on the monthly amount of recyclables sold. The financial resources offered by producers are negotiated directly between them and the NGOs and are usually not related to the amounts recycled or to the costs of WP services, but they usually cover the costs and revenues of the NGO's operations. NGOs and producers vary in the way they support WP co-ops. ANCAT coordinates the largest experience of the BR P-EPR and organized a national bidding process to choose which WP co-ops would receive financial and technical support each year. In 2019, 277 cooperatives were supported in the P-EPR scheme managed by ANCAT, corresponding to 22,5% of the 1232 WP co-ops registered on the Brazilian Sanitation Information System (SNIS in Portuguese initials) (SNIS 2018).

FIGURE 4: Flow diagram and actors involved on Brazilian P-EPR. Orange line marks the action field of P-EPR. Source: CEMPRE 2018.

The traceability of the BR P-EPR scheme managed by ANCAT is provided by sales invoices from the WP co-ops registered in a database, implemented in 2017. This database also includes information on the WP's income and the material selling prices. This database was the basis for the first Brazilian Recycling Yearbook, launched in 2019, which aims to annually record the WP' contribution to recycling in Brazil. This was a remarkable innovation because it is extremely difficult to quantify the IRS' contribution to recycling (Burcea 2015).

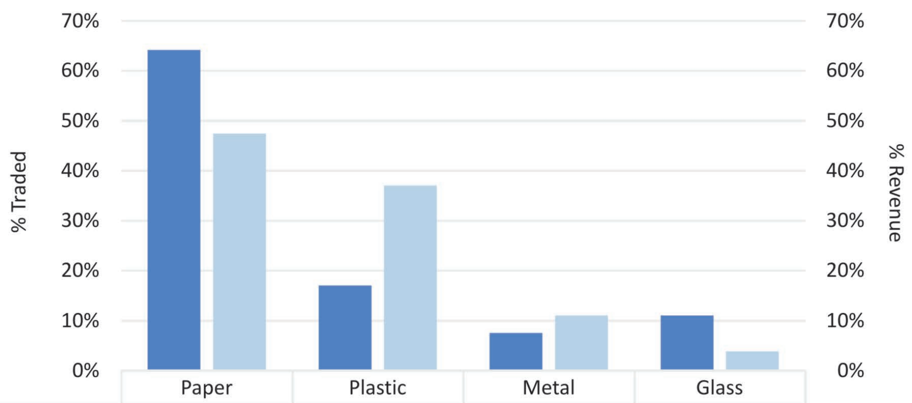
The ANCAT database records that, from 2017 to 2019, WP co-ops diverted 168,101 tons of recyclables from waste, yielding a cumulative cost savings of more than U$ 4.0 Million to municipalities and providing additional services worth another U$ 20.3 Million (ANCAT 2020). In 2019 WP co-ops recycled 9.9% of all packaging market in companies involved in one of the P-EPR programs managed by ANCAT, the "Recycle for Brazil" program (ANCAT 2020). ANCAT data also disclose that WP co-ops had an average productivity of 48.2 tons / month, representing an average physical productivity of 3.48 tons / month for each WP. This productivity is three times greater than that recorded in the literature for an average autonomous picker, who is said to collect and recycle about 40-50 kg of recyclable waste daily (Chen et al. 2018).

Data coming from other BR P-EPR experiences also reveal impressive results. The database of the initiative "Novo Ciclo" manage by the NGO INSEA registered that 77% of PET packaging and 17% of PS packaging marketed in by Danone in Brazil in 2017 were collected and treated by WP co-ops (INSEA 2018), and SNIS registered that WP co-ops collected 30.7% of the household waste diverted for recycling in Brazil in 2018 (SNIS 2018).

Figure 5 shows the materials treated by the WP co-ops supported by ANCAT and the respective revenue obtained from different recyclable materials. Note, for example, that although plastic represents just 17% of the amount of recyclable material sold, this material provides 37% of sales revenue.

ANCAT (2020) records 17 different types of products marketed as metals, 9 different types of paper and cardboard, including aseptic carton packs, 7 types of glass, 22 different types of plastics and 10 types of electro-electronic waste. There are also different selling prices for these materials, depending on the geographic region of collection and trade (Table 3). The South and Southeast regions are those where selective collection is more widely implemented (SNIS 2018) which should improve the quality of the recyclables, but the differences in the sale prices of recyclable materials can also be related to the market context and the recycling industry in each of these regions (Rutkowski & Rutkowski 2017).

Figure 6 shows the results of the BR P-EPR scheme related to improving the operational capacity of WP co-ops, in three different phases in the timeline. Phase I indicates the baseline diagnosis of the cooperatives when the P-EPR scheme was started; Phase II is the diagnostic in 2018 and Phase III the situation in 2019. The conditions of management and operation of the WP co-ops, which are self-managed companies organized under the Social and Solidarity Economy (SSE), have been continuously improved in all seven aspects monitored. The financial assistance received contributed to improve working conditions, operational and workforce management and, consequently, their overall productivity. Many cooperatives could be legalized due to better administrative organization, regularized accounting procedures and settlement of eventual debts, which, in turn, facilitated their relationship with the recycling sector, LA and other partners. The financial and administrative stability also contributed to the systematization and strengthening of its internal self-management procedures.



	Paper	Plastic	Metal	Glass
% Traded	64.1	17.0	7.5	11.0
% Revenue	47.4	37.0	11.0	3.8

■ Percentage of each recyclable material traded among the total traded (2017-2019)

■ Percentage of total revenue earned from each material (2017-2019)

FIGURE 5: Composition of the packaging waste traded and composition of the revenue obtained by the WP co-ops, between 2017 and 2019, in percentages of each recyclable material traded among the total quantities traded and percentages of aggregate revenue for each material to the total revenue obtained by WP co-ops. Source: Elaborated by the author from ANCAT (2020).

TABLE 3: Average selling price of recyclable materials in each Brazilian region in 2019. Bold values highlight the highest and smallest selling price got for each material.

Region	Metal (US$/ton)	Paper (US$/ton)	Plastic (US$/ton)	Glass (US$/ton)
South	172,37	103,93	253,49	20,28
Southeast	205,32	**116,60**	**266,16**	25,35
North	**154,63**	**65,91**	218,00	**38,02**
Center-West	**154,63**	81,12	**197,72**	**5,07**
Northeast	**215,46**	86,19	263,62	22,81
Brazil (average)	182,51	98,86	240,81	20,28

Source: Elaborated by the author from ANCAT 2020

The support for administrative improvement in WP co-ops was considered essential to increase WP's income and the ability to be contracted as public service providers. WP generally have an extremely low school experience - 17% of the WP in the supported cooperatives are illiterate and 60% have less than 4 years of schooling (ANCAT 2019). Low schooling prohibits most WPs from knowing and using management techniques and tools (Rutkowski 2008), considered essential to ensure traceability of P-EPR schemes and control of the waste services offered (Gutberlet 2015; Demajorovic & Massote 2017). This knowledge is necessary to facilitate dialogue and networking with waste management and recycling companies (Rutkowski 2013) and, therefore, the inclusion of WP in formal WM and EPR systems.

Financial support from producers partially addresses the lack of access to working capital, a major problem faced by cooperatives in many LMICs (Gutberlet 2009; Rutkowski 2013). WPs organized in cooperatives share activities and responsibilities, which leads to better business management and better working conditions, allowing them not to have to work in unsanitary conditions, in dumps or on the streets as before (Demajorovic et al. 2014; Gutberlet 2015). They can sell recyclable materials in better conditions than they could as individuals (Medina, 2000; Gonçalves-Dias and Teodósio 2006; Rutkowski&Rutkowski 2017); they reduce the transaction costs of their activities (Rutkowski 2013), doing so under a technology that proves to be energy efficient and socially and environmentally sound (Gunsilius et al. 2011; King & Gutberlet 2013). They are also empowered by being involved in waste management and recycling chains preserving their livelihood (Gutberlet 2008; Gunsilius et al. 2011; Rutkowski & Rutkowski 2015; Dias, 2016). This support, therefore, helps to increase not only the WP co-ops sustainability but could increase the productivity and overall efficiency of the waste recycling system, improving the ability of producers to meet the packaging recycling targets and their commitments for recycling plastics worldwide.

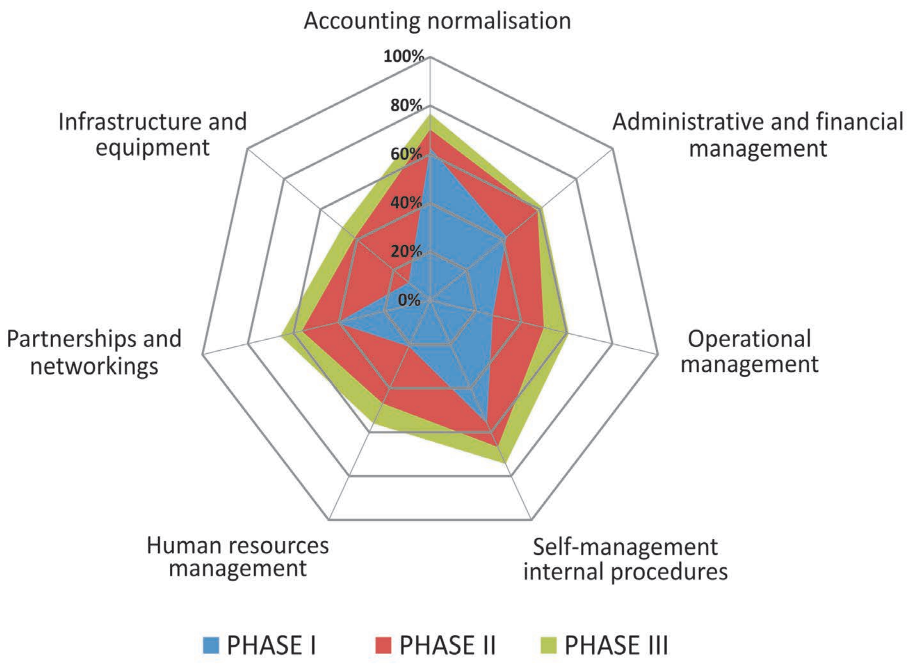


FIGURE 6: Evaluation of the management and operational capacity of the WP co-ops, in three distinct phases. Phase I indicates the cooperatives' baseline diagnosis when the P-EPR scheme was started; Phase II is the situation diagnosed in 2018 and Phase III is the situation in 2019.Source: ANCAT 2020.

4. LEARNING FROM TWO MODELS FOR AN EQUITABLE AND INCLUSIVE EXTENDED PRODUCERS RESPONSIBILITY FOR PACKAGING SCHEME

4.1 Comparing the EU and Brazil P-EPR models

Table 4 summarizes the main characteristics of the two models — the organizational similarities and the different approaches, such as the roles of the stakeholders. It is noteworthy that although organized in a country of continental dimensions, the BR P-EPR scheme operates without a PRO having been organized and without involvement of the LA. The traceability of the data raises criticism in both systems. Also, in both, EPR schemes help to reduce WM costs but do not bear the full cost. It was not possible to say which one is less costly to companies and which is

TABLE 4: Summarized main characteristics of the two studied P- EPR models.

Aspect	EU P-EPR	BR P-EPR
Principles	Polluter pays principle and Circular Economy are clearly expressed in legislation	Polluter pays principle is expressed in law but although waste hierarchy is cited, there is no mention to CE.
Legal Framework	Directive 94/62/EC and amendments Directive 2008/98/EC and amendments	PNRS- Law 12.305/2010, Decree 7.404/2010, Decree 7.217/2010, Sanitation Law nº 11.455/2007
Targets	Related to % of recycled packaging waste: 50% of municipal waste and 55% of packaging waste should be recycled / reused;	Related to% of packaging waste disposed of in landfills and% of the fraction of dry waste recovery: reduce packaging waste disposal in landfills by 22% recovery: reduce packaging waste disposal in landfills by 22% and increase the recovery of dry waste fraction by 20% compared to the situation in 2013.
Role of producers	Producers must organize and / or finance the collection, transport, and pre-treatment processes necessary for recycling packaging waste. For this, they pay fees defined for each material and calculated according to the quantity of packaging sold by each producer. They must also provide information for recycling on their packaging.	Producers provide financial support for technical assistance and for improving the infrastructure of WP co-ops to increase their productivity in the processes of collection, transportation and pre-treatment for recycling. They are also responsible for implementing points for voluntary packaging delivery (PEVs), in addition to providing consumers with information for packaging recycling.
Role of LA	LA are legally responsible for waste management and for organizing municipal selective collection. In most countries, they receive financial support from producers to send packaging for recycling.	LA are legally responsible for waste management, however, in most cities, there is no municipal selective collection. LA is not directly involved in the scheme, although in some cities WP co-ops have service provider contracts with LA or have agreements with them, receiving support such as a warehouse concession, for example.
Packaging waste collection operators	The collection service is done by public or private waste collection service providers hired by LA	In most cities, WP co-ops and autonomous WP do the collection. In the biggest cities, the selective collection is done by private service providers hired by LA.
Pre-treatment processes for packaging recycling	Pre-treatment processes for packaging recycling operated by private companies. Sometimes these processes are operated at facilities owned by LA.	In most cities, this service is operated by WP co-ops and there is an agreement between them and LA.
Role of PROs	PROs, organized in different EU's member states, are financed by fees paid by all or a large part of companies in an industry but have financial and administrative independence from them. PROS contract collection and pre-treatment services for packaging waste from LA and private companies and are also responsible for recording and controlling recycling data.	No PRO organized. Companies act individually, contracting NGOs for intermediating relations with WP co-ops and LA. Most of companies act collectively by mean of their industrial sector association or forming coalitions.
Reporting and monitoring	Each country has a third part responsible by monitoring recycling data reported by PROs to Government Environment Department.	Annual reports on goals and results are presented by companies or their coalitions to Government Environment Departments and published on websites.
Data traceability of the recycling targets	Not available to the public for alleged commercial reasons. The accuracy of the applied measurement methodologies is questioned, pointed out as not comparable between countries.	Not available to the public for alleged commercial reasons, records based on the sales invoice for waste packaging treated as raw material, easily controllable, but probably registered below the real due to an informal market and the lack of registration and control capacity in WP co-ops.
System financial flow and surveillance	PROs transfer part of the resources paid by producers to service providers, reducing expenses with waste management in the municipalities, but without covering 100% of the packaging collection costs. The scheme is also considered to encourage investments in the recycling chain. The fees paid by companies are defined by the PROs and therefore competition between the PROs is encouraged. The financial surveillance of the system is carried out by the stakeholders of the PROs. Cost- effectiveness not measured.	Expenses with waste management in the municipalities are indirectly reduced by the action of WP co-ops, which reduces the costs of collecting municipal waste and disposal. The scheme also encourages investments in the recycling chain since WP co-ops are an essential link to feed the chain. Companies define how much is transferred to NGOs that define values to be transferred to WP co-ops, but these values are not related to actual costs. No financial surveillance of the system is reported. Cost-effectiveness not measured.
Free riders' control	Most of the countries have specific legislation and ways of acting, none of them reported as 100% effective	Legislation oblige all producers to participate on the scheme whereas no enforcement mechanism is defined. Some state governments have proposed specific controls recently.

Source: Elaborated by the author

more cost-effective for recycling targets.

Municipal waste recycling targets differ considerably between the two schemes. First, the BR P-EPR is more recent than the EU P-EPR, as usual, the BR P-EPR starts with modest goals that should grow over time. Second, few municipalities practice selective waste collection and there are no information, education, and awareness programs for the separation of recyclable materials at the source in operation, either by the LA or by EPR schemes. Consequently, a small percentage of municipal waste is diverted from the landfill for recycling by formal WM systems. Even so, the industry registers considerable levels of recycling of papers, metals, and some types of plastics, whose responsibility it attributes to the action of the WP (Rutkowski&Rutkowski 2017; CEMPRE 2018).

In addition, the share of recyclable waste in the composition of urban waste is greater in the EU MS than in LMIC, as shown in Figure 7. However, although in LMIC, organic matter is the largest fraction of waste, the composition of recyclables collected and treated by WP co-ops (P WP) approximates the composition of EU MS packaging waste (PW EU), as shown in Figure 8. For comparison purposes, this Figure also shows the average percentage of recyclables that make up municipal waste in Brazil (W BR). Thus, concerning the CE perspective, the inclusive P-EPR scheme seems capable to add beneficial innovations to the P-EPR.

As shown in Table 3, the average price of plastics is 2.5 times that of paper but 64% of the materials marketed by the WP co-ops is paper, which makes up 13.1% of MSW; while 17% of the materials marketed by the WP co-ops are plastics, which makes up 13.5% of MSW, but provides 37% of WP co-ops' revenue (Figure 5). Despite of WP co-ops efforts to recycle plastics, they have been more effective in recycling paper. In fact, they are limited by householder separation, which is much more efficient for paper than for plastics due to a lack of environmental education. Notwithstanding, WP co-ops value a greater number of subcategories of materials, including plastics, responding to the specific needs of buyers in the recycling industry (Purshouse et al. 2017). They also know better than LAs how to work with the value chains and market the materials (GiZ 2018a).

Typical MRFs in the EU MS usually classify a variety of nine commingled collected materials: corrugated cardboard, newsprint, magazines, and mixed paper make up the paper fraction; the metal fraction is made up of aluminum and ferrous metal. The plastic fraction is composed mainly of HDPE (high-density polyethylene), bottles and pots of PP (polypropylene) and PET (WRAP 2015). For glass, in most EU countries, the main method of collection and classification is the single flow "bring bank" system (WRAP 2006). On the other hand, as described in Section 3.2, the output of the WP co-ops consists of 55 different recyclables subfractions (ANCAT 2020).

There are significant differences between the PW EU and P WP plastic waste subfractions, as shown in Figure 9. Inclusive P-EPR schemes recover different types of plastics but EU P-EPR prioritizes the valuable ones. In the EU MRFs, 3 types of most valuable plastics (PET, HDPE, and PP) compose 80% of the plastic treated and ¼ of the material is unidentified, which means that will be transformed in refused material to recycling. However, WP co-ops sort 11 different plastic subfractions to be recycled. PET, HDPE, and PP represents 56% of the plastic fraction treated by them.

The outputs of the EU MRF are defined by technological restrictions imposed by the automated process they employ, which, in turn, will influence the selective collection implemented by the LA. The capacity of contracted MRFs will also define the information provided to consumers on what should be separated in each region. The profit rates of the recycling businesses are an important factor for the organization of the system, as they directly impact the costs to be paid by packaging producers. In turn, WP normally collects all available recyclable materials, be it plastic, pa-

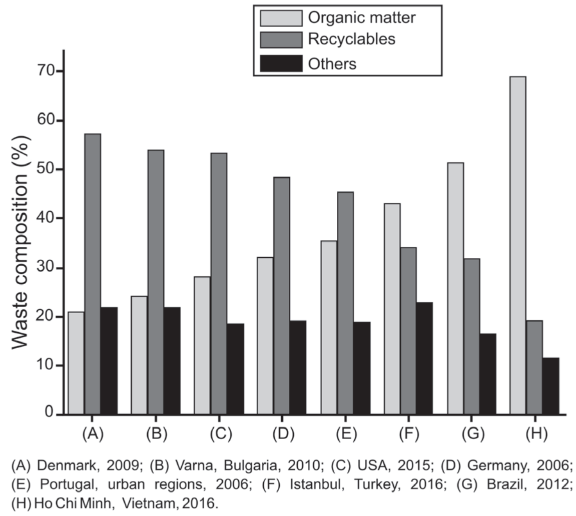


FIGURE 7: Waste composition in different countries. Source: Alfaia et al. 2017.

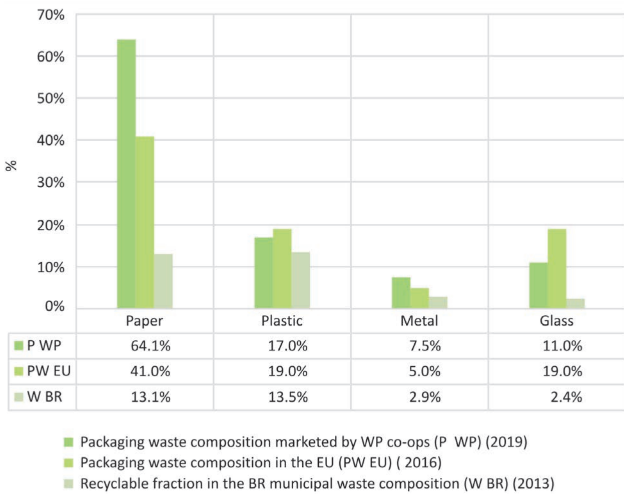


FIGURE 8: Comparison of packaging waste composition in the EU (PW EU), packaging waste marketed by WP co-ops (P WP), and the average percentage of recyclable materials that make up Brazilian municipal waste (W BR) for the main recyclable materials. Source: Elaborated by the author from ANCAT 2020, Eurostat 2019, CEMPRE 2018.

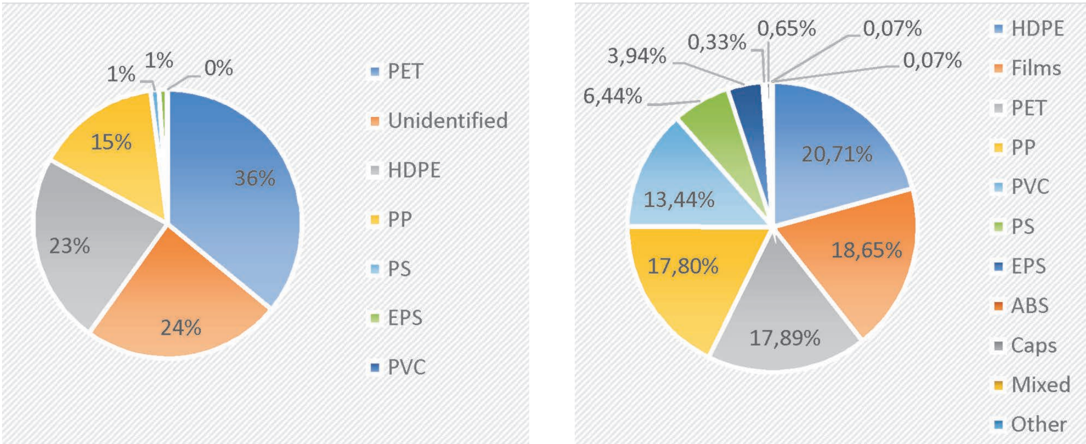


FIGURE 9: Plastic packaging sorting composition in EU MRFs (left) and WP MRFs (right). Source: Elaborated by the author from WRAP 2006; ANCAT 2020.

per, metals, electronics, furniture, clothing and even used frying oil. They are interested in collecting and preparing for sale as much material as possible, since their income comes from what they sell, usually at tiny prices. WP describe themselves as "environmental agents" committed to increasing waste recycling rates. Organized under SSE framework, WP enterprises are based on values such as cooperation, solidarity, trust, self-management, and the feeling of belonging to a common social condition (Rutkowski, 2008; Gutberlet 2015). WP do not just evaluate their businesses in terms of cost-benefit analysis.

In developed countries economic market concerns have been the main drivers of waste recycling rules (Rogoff and Ross, 2016). However, market-driven actions are usually not the most efficient in environmental and social terms; recycling can remain stagnant if supported only by market forces. To meet the objectives of the CE, as much of any recyclable material as possible must be separated from the waste and sent for recycling. In the inclusive model, a wide range of packaging is being effectively recycled due to the modus operandi of the WP.

The BR P-EPR system is operating independently from LA and without a centralized operator. In Europe, the system depends on the organization of PROs, often criticized for lack of transparency in their actions and for environmental results below expectations, such as those related to green innovation, DfE and waste flow treated (EASAC 2020; Lerpeniere & Cook 2018; EC-DGEnv 2014; Massaruto 2014; da Cruz, Simões & Marques 2014; Mayers&Butler 2013; Brouillat & Oltra 2012). On the other hand, in most developing countries, LA have not been able to implement WM efficiently. Therefore, the inclusive scheme is providing a simpler operation to P-EPR that can be considered an easier system for deployment and dissemination in similar situations.

Nevertheless, in the long run, the system cannot ignore the LA that are legally responsible for WM. As proposed in some EU MS, producers must reimburse LA for the cost of packaging collection. In turn, LA must not ignore that subcontracting the WP co-ops as providers of selective collection services benefits WM, reducing costs, amplifying the efficiency of municipal selective collection, and improving the quality of collected material. On the other hand, WP co-ops have also added benefits to the recycling value chain, amplifying resource recovery when performing MRF services. Therefore, the choice of producers to include WP co-ops as agents in P-EPR schemes should also be encouraged.

Finally, EPR is an environmental policy approach that should also develop Design for Environment (DfE) activities and innovation for reduce packaging waste generation. Neither of the two P-EPR models showed evidence of achieving these objectives. In the Brazilian scheme, DfE was not ever mentioned. On the other side, across EU, between 2013 and 2015, the amount of packaging waste generated grew by 6% suggesting that more effort on waste prevention is needed (EASAC, 2020). An eco-modulation fee to be paid by producers has been suggested as a lever to encourage eco-packaging design at EU-EPR. Regulations in this direction are in place in Germany and under regulation in Italy and France (Scharff 2018). Further studies on these experiences should indicate their validity to reduce waste generation and improve the use of resources in packaging.

4.2 Proposing an equitable and inclusive P-EPR model

Performance of EPR schemes are influenced by many factors as population density and country geography; historical development of the waste management infrastructure and existence of complementary waste policy instruments (pay-as-you-throw schemes and landfill taxes, for instance); the value of secondary materials on the national market; awareness and willingness of citizens to participate on the waste recycling effort (EC-DGEnv 2014).When the P-EPR was established in Europe, most municipalities already run well-organized waste management systems and the recycling market was incipient. In this context, the producer's financial resources were used to encouraged private sector's investment in the recycling industry and to modernize waste collection systems through the addition of technology, which resulted in increased operating costs

(OECD 2016). However, recent studies showed that the decision to implement EPR schemes in some EU countries without considering the IRS prevented the system from achieving its objectives (OECD 2016; Scheinberg et al. 2016; Mrkajić et al. 2018; Wiesmeth et al. 2018; Ferronato et al. 2019).

On the other hand, in LMIC, LA cannot afford expensive waste management systems (UN HABITAT 2010; Giz 2018a; Ferronato et al 2019). As waste management for recycling is poorly implemented by LA, the waste recycling system has been developed based on WP's social technology (Scheinberg 2012; Rutkowski & Rutkowski 2015). In addition, LMIC has an abundant workforce, which in the most can not be employed in modern technological industries and services. The inclusive recycling schemes then appear as an "appropriate technology" (Schumacher 1973), a technologically accessible, labour-intensive, autonomous, decentralized option for upgrading WM in these countries.

Another interesting innovation observed in the BR P-EPR scheme related to CE is providing mutual learning and support for both stakeholders in relation to recycling issues. Producers can better understand the challenges of recyclability because WP are able to clearly indicate which materials cannot be recycled due to restrictions from the market. This information has helped producers to take measures to improve the recyclability of their packaging (Demajorovic & Massote 2017; DANONE ECOSYSTEM, 2016). Waste pickers, in turn, improved their learning about the recycling industry, due to the opportunities created by packaging producers to overlap with intermediaries in the recycling value chain. Nationally organized into cooperative networks, Brazilian WP are transforming their position in the recycling sector. This type of learning and networking between different economic groups is an interesting innovation and support for new approaches required for the Circular and Green economies around the world.

Thus, the modernization of the WM in the LMIC must be made from the improvements in the conditions of the WP in operating the CSS. Figure 10 summarizes the benefits that the inclusion of WP co-ops could bring to cities towards an integrated and sustainable WM. Due to the improvement of the municipal WM and the livelihoods of a vulnerable urban population, the scheme results in an affordable and efficient alternative for the implementation of ISWM in emerging countries.

Figure 11 represents schematically a proposal for an equitable and inclusive P-EPR. The formalization of WP co-ops as a provider of municipal selective waste collection services for LA and as a provider of sorting and pre-treatment services for recyclables for packaging producers is the answer to the research question on how waste pickers should be invited to work with the formal WM system to achieve an ISWM and make cities more sustainable and inclusive. Based on the polluter pays principle, packaging producers should be responsible for financing this system in LMIC by paying fees that, together with the householders' WM fees, should cover the total costs of both services. Further research is needed to suggest how these costs can be fairly defined and to assess what is the best governance needed to collect and manage these fees. As well as to improve data collection and assessment on EPR and waste management. Such aspects are highly criticized in both EU and BR schemes.

Learning how to work in an inclusive operational mode

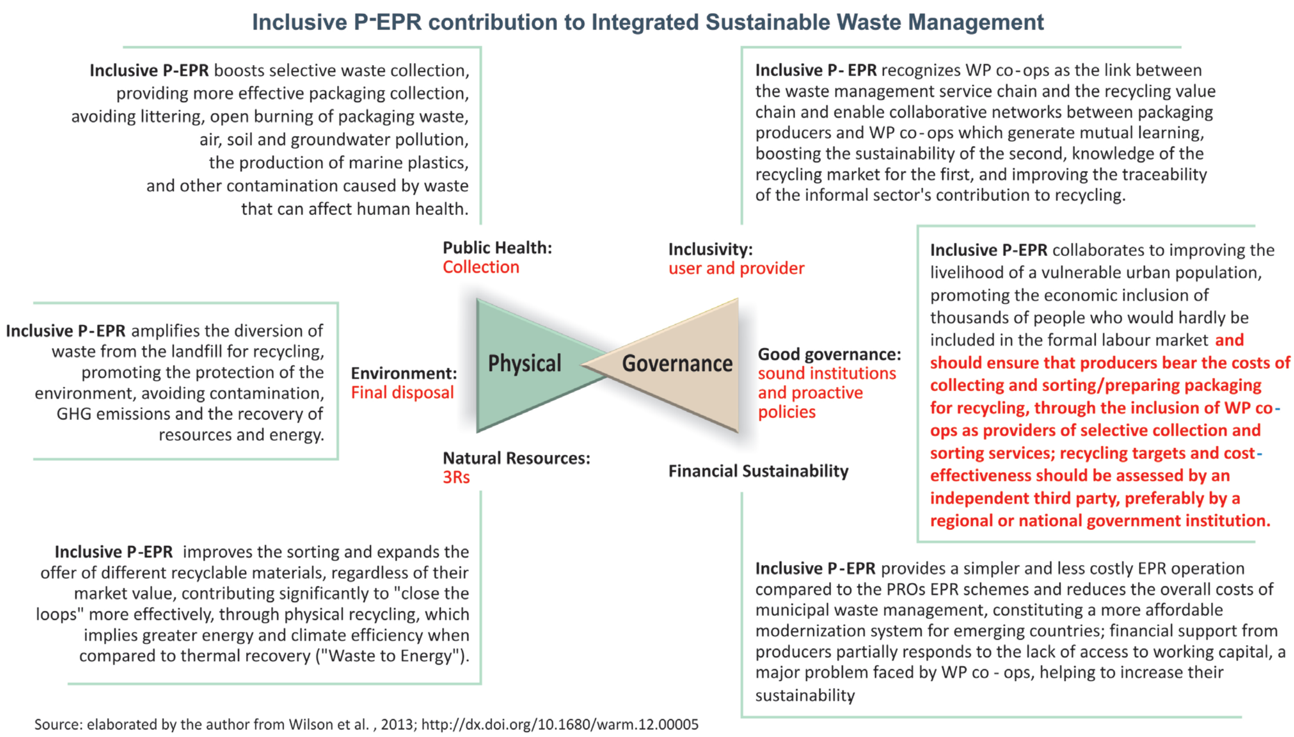


FIGURE 10: Inclusive EPR schemes contribution to the Integrated Sustainable Municipal Waste Management. Source: Elaborated by the author from Wilson et al. 2013.

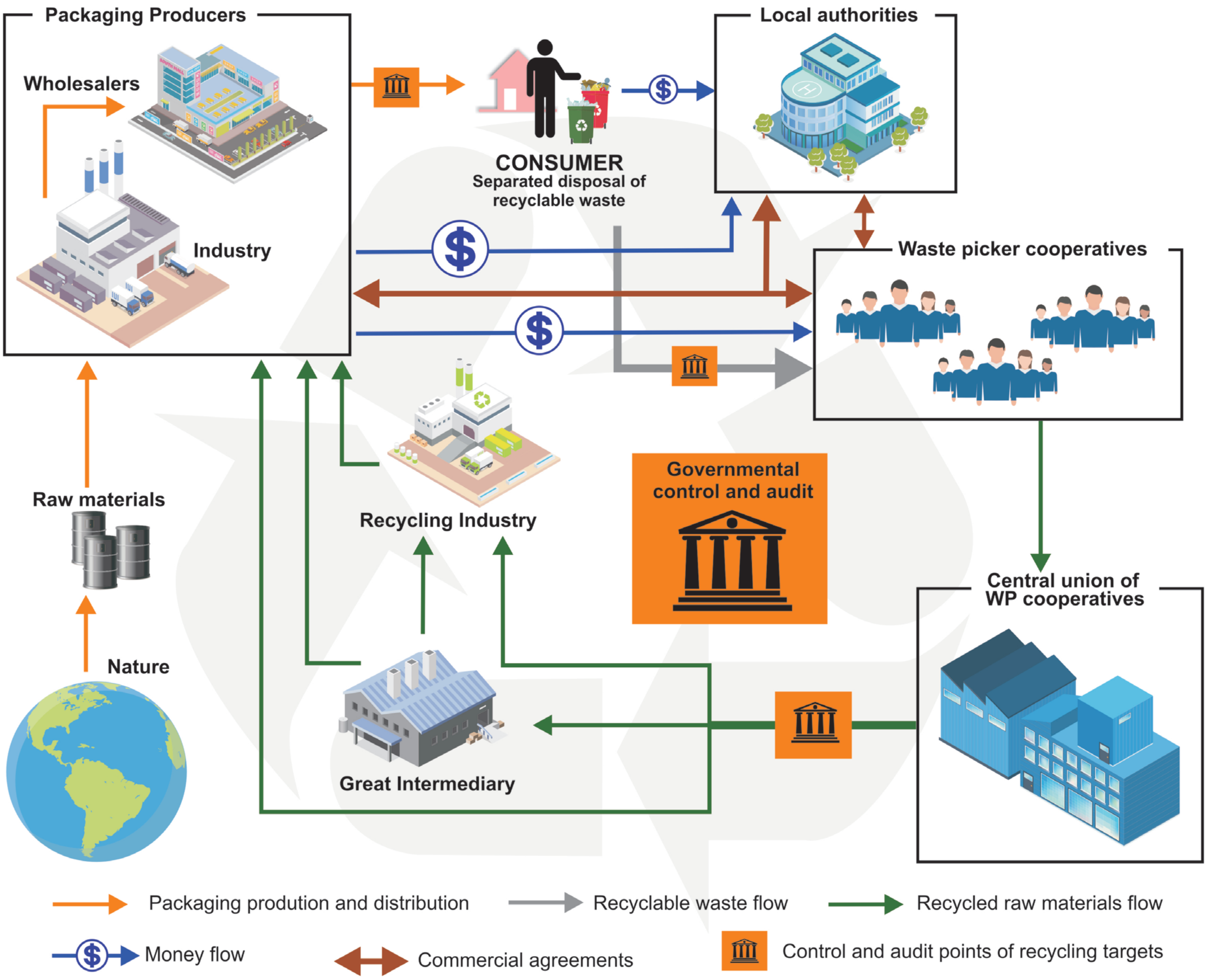


FIGURE 11: Schematical representation of an Equitable and Inclusive P-EPR scheme. Source: Elaborated by the author.

can be crucial to achieve increasing plastic recycling targets agreed by packaging producers worldwide and important to achieve the ambitious EU waste recovery and diversion targets listed in Table 2. Particularly in those EU countries where IRS activities have been imputed to meet, exceed, or at least be indispensable to achieve EU objectives for recycling and recovery, such as for some Eastern EU countries with a largely Roma IRS (Scheinberg et al. 2016; Mrkajić et al. 2018; Ferronato et al. 2019; da Cruz et al. 2014 ; GiZ 2018a). It would also be beneficial for improving the local recycling industry due to the increased availability of diverse secondary raw materials, and for creating green jobs capable of absorbing unskilled labour.

5. CONCLUSIONS

After a comparative analysis between the Brazilian and EU EPR schemes for packaging, the research highlighted some aspects and advantageous results of the contribution of the inclusive EPR scheme to increase the efficiency of the municipal WM in resource recovery. An equitable and inclusive P-EPR is proposed as a simpler and less expensive innovative solution for the implementation of packaging EPR, constituting a more accessible WM modernization system for emerging countries, including some in the EU.

The study also showed that improvements in the EPR schemes must be implemented. The lack of strong and structured initiatives to improve the separation of dry waste at the source and the involvement of LA as an interested party prevented the BR P-EPR scheme from rapidly increasing recycling rates. These actions can not only improve overall waste recycling rates, but also improve WPs' livelihoods, since a large part of their income depends on the sale of materials.

It is also observed that just defining and controlling waste recycling goals does not necessarily result in real improvements in the general waste management system. Both the EU and BR P-EPR systems do not have effective cost and cost-effectiviness control, which could guarantee efficiency improvements in WM policies.

Reporting and data control on EPR and WM also need to be improved and harmonized to ensure efficient implementation of the expected environmental objectives of EPR schemes. This is not an exclusive defect of the inclusive

P-EPR system, but it seems to be related to the inability of governments to make companies agree to pay real recycling costs, as well as to reduce the quantities of marketed packaging, which can be observed throughout the world.

However, neither these nor other aspects described seem to prevent the inclusive P-EPR model from being transposed to other countries where the presence of waste pickers is registered, despite the political and institutional issues that may arise and need to be addressed.

The inclusive and equitable P-EPR scheme is presented as a solution for the inclusion of the IRS in the WM, as well as to achieve the ISWM in LMIC. The scheme can make cities more sustainable, inclusive, and capable of improving global recycling rates and reaching the highest plastic recycling targets agreed by many packaging producers towards the Circular Economy.

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LIFE CYCLE ASSESSMENT OF BEVERAGE PACKAGING

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ABSTRACT
Global plastic production has been increasing annually since World War II and is currently at least 380 million tonnes. Plastic drinks packaging is ubiquitous; over 13 billion plastic bottles are used per year in the United Kingdom alone. Global concern about pollution from plastics in the seas and the environmental costs of plastics manufacture is rising. This study aimed to: i) review the costs, benefits, advantages and disadvantages of plastics as packaging materials and ii) use life cycle assessment to determine if there is less environmentally impactful beverage packaging than plastic bottles. As different beverages have different packaging needs, three categories were used: commonly used containers for milk, fruit juice and pressurised 'fizzy' drinks. The packaging types included in the assessment were glass bottles, aluminium cans, milk cartons, Tetra Pak, polyethylene terephthalate (PET) bottles and high-density polythene (HDPE) bottles. The ISO 14040:2006 and ISO 14044:2006 standards for life cycle assessment formed the basis of the methodology. The open source software openLCA was used to conduct the life cycle assessments. Data was assembled from free LCA databases such as the European reference Life Cycle Database of the Joint Research Center (ELCD), existing life cycle assessments, scientific reports and peer reviewed literature. The functional unit was set at a container that held one litre of fluid. The results found that in each category there was a less impactful beverage packaging than plastic bottles. In the Pressurised Beverage Category, it was found that 100% recycled aluminium cans would be the least impactful option, in the Fruit Juice Beverage Category it was found that Tetra Pak would be the least impactful option and in the Milk Beverage Category it was found milk cartons would be the least impactful option.

1. INTRODUCTION

Beverage containers have been in existence nearly as long as civilisation. Needing something in which to hold and store drinks, humans used a wide range of containers such as animal skins, stone, earthenware and glass. These containers were usually hardwearing and used repeatedly. In the modern era, many drinks containers are made of plastic and have become single use; used once then disposed of. This is due to changes in lifestyle; individuals no longer make much of their own food or buy products such as milk from local sources. The rise of convenience food and mass production has changed how beverages are purchased, stored and consumed. A reduction in food prices means that consumers are buying more beverage packaging because they are, in general, able to afford more food (Andreyeva et al., 2010; Epstein et al., 2012).

Plastics have become a fundamental global feature of everyday life. Plastic is an umbrella term for items manufactured with any synthetic or semi-synthetic organic polymers. Plastic may be shaped when soft and then hardened to retain a given shape. A polymer is a chain in which each link is the "mer," or monomer (single unit). The chain is manufactured by joining, or polymerizing, at least 1,000 links together. The most produced polymers are, in descending order: polyethylenes (low and high density) (PE), polypropylene (excluding fibres) (PP), fibres of acrylic, polyamide and polyester, then polyvinyl chloride (PVC), polyethylene terephthalate (PET) and polystyrene (PS) (Geyer et al. 2017).

Plastics have a number of benefits as a packaging material for beverages as they are light, durable and cheap to produce, able to withstand pressure and can contain many different fluids safely. Nevertheless, plastics have limited recyclability; they are made of polymer chains that shorten each time they are recycled and thus usually 'downcycled' into a lower quality product rather than back into packaging (La Mantia, 2004). Methods such as life cycle assessment (LCA) enable an evaluation of the true costs and benefits of plastic packaging materials.

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Plastic bottles are the world's most common beverage packaging. In the UK; in the year 2016, >13 billion plastic bottles were produced, with only 7.5 billion of those going for recycling (Environmental Audit Committee, 2017). The most common types of plastic bottles for beverages are made from PET or HDPE. Both plastics have been highlighted as a priority pollution problem (Environmental Audit Committee, 2017). Indeed, a ban on plastic bottles for beverages has been widely touted for both environmental and public health reasons (see e.g. Thompson et al, 2007). However, a ban on plastic beverage bottles would remove a practical option for water storage and dissemination during times when municipal tap water supplies are contaminated. Plastic bottles are versatile (can keep liquids hot or cold), relatively inexpensive, and can keep beverages healthy, safe, and convenient. In developing countries with poor infrastructure, bottled water offers a partial solution to unsafe drinking water. Clearly, there are well-developed arguments on both sides

The abundance of plastics, particularly disposable items, has generated a global public outcry over plastic pollution. Items such as plastic straws are being banned (The Ellen McArthur Foundation, 2016). There is a focus on plastic bottles as a waste product or plastics as a pollutant but the entire life cycle of a plastic bottle must be considered to understand its full environmental impacts. Plastic manufacture relies on the extraction of raw materials such as crude oil; this has environmental impacts which are not always noted with the same attention as, for example, marine plastic pollution (O'Rourke and Connolly, 2003). Adverse impacts can include; ozone depletion, petroleum hydrocarbon emissions to the atmosphere and a high generation of solid wastes (O'Rourke and Connolly, 2003). These impacts are purely for the extraction of the raw material, they do not account for the energy needs and emissions generated from manufacture or additional processes. The whole life cycle must be assessed to address all the impacts of plastic.

To minimise the impacts plastics have on the environment, avoidance or minimisation of use is key. However, vital items must be manufactured from plastics, such as personal protective equipment – so important during the Covid19 pandemic - in the health sector. Some products such as beverage packaging have a long history of manufacture from a variety of materials. Replacing plastics with a less impactful packaging type may help mitigate the many impacts of plastics.

Plastics are used for packaging various beverages. According to the UK Department for Environment, Food & Rural Affairs' (DEFRA) Family Food Statistics 2016/2017, the amount of milk purchased per person is comparable to the amount of soft drinks purchased per person. To understand the impacts of plastics used to package beverages, different beverages with different packaging needs should be examined.

Each type of packaging has different requirements, fizzy drinks require packaging that can be pressurised, milk packaging needs to aid refrigeration and fruit juices are often unrefrigerated so have specific hygiene needs. Knowing which packaging has the least impacts for each type of beverage could aid consumers, lawmakers, pressure groups and businesses to understand the true impacts of beverage packaging in relation to each.

2. LIFE CYCLE ASSESSMENT OF BEVERAGE PACKAGING

2.1 Previous studies

LCA is a suitable tool to compare different types of packaging that serve the same purpose as it compares the products assessed against only each other. LCA is based on product system results in relation to each other rather than their impacts overall; it can only show if something is 'better' or 'worse' than another. LCA is guided by two International Organization for Standardization (ISO) standards, ISO 14044:2006 and ISO 14040:2006, and for an LCA to be deemed valid by other practitioners, it should adhere to these standards (Bjørn et al., 2018c).

LCA is often considered when the sustainability of a product or process or measurement of how 'environmentally friendly' something is needed. A number of studies have used LCA to review the environmental impacts of drinks packaging, with some focusing just on plastics packaging, or on specific types of beverages such as carbonated drinks or milk (Amienyo et al., 2013; Romero-Hernández et al., 2009). Many LCAs have been conducted on beverage packaging, some by companies on behalf of beverage packaging producers and others by academics. Almost all follow the ISO standards for LCA and many assess glass and PET bottles due to their use across different beverages.

The majority of LCAs that have assessed beverage packaging concluded that glass is the most impactful beverage packaging regardless of the other packaging types involved (Amienyo et al., 2013; Franklin Associates, 2009; Jelse et al., 2009; Meyhoff Fry et al., 2010; Saleh, 2016). Amienyo et al, (2013), noted that glass had the highest global warming potential (GWP) compared with aluminium cans and PET bottles and concluded that the PET bottle was the least impactful of the three. PET and HDPE bottles' assessed impacts vary across recent LCAs, however they are consistently presented as less impactful than glass and more impactful than composite packaging such as milk cartons (Franklin Associates, 2009; Jelse et al., 2009; Meyhoff Fry et al., 2010).

A report by Franklin Associates (2009) concluded that aluminium cans are more impactful than PET bottles as they have higher energy demands, higher solid waste generation and greenhouse gas emissions. However, the report also noted that aluminium can manufacture uses less fossil fuels than PET bottle manufacture due to the widespread use of hydropower in primary aluminium smelters (Franklin Associates, 2009). This highlights the importance of correctly allocating energy sources within LCAs e.g. the work of Saleh (2016), based in Palestine, reported drastically different values than the Franklin Associates (2009) study based in America. Data must be suitable for the country of study.

Meyhoff et al, (2010) and Jelse et al, (2009) found product systems with plastic elements were the most impactful in the product systems compared and both advocated lessening plastic content. Jelse et al, (2009) found that Tetra Pak containers with plastic caps had ~30% higher GWP than other Tetra Pak product systems, indicating that even a comparatively small increase in plastic content can cause a considerable increase in impacts (Jelse et al., 2009). Both these studies included plastic bottles and products that contained some element of plastic, therefore were comparing similar products, and demonstrating how ubiquitous plastics are in beverage packaging.

Amienyo et al, (2013) compared PET packaging to packaging without plastic elements and found it less impactful, within the system boundaries of the study, than the glass bottles and aluminium cans in many categories. Different levels of reuse and recycling were included as PET plastics have a limited recyclability whereas both glass and aluminium can be recycled indefinitely (Amienyo et al., 2013). The study – unsurprisingly – found that improving recycling and reuse of all packaging types would lessen their impacts. Accorsi et al's (2015) conclusions seem different from that of Ameinyo et al, (2013), classing glass bottles for extra virgin olive oil (EVOO) as less impactful than PET bottles, though this was under the assumption that the glass was recycled at a higher rate and with differing transportation assessed.

Glass generally has the highest impacts out of beverage packaging options, followed by plastics and aluminium cans that are found to be more impactful than composite packaging. Many current LCAs only focus on one beverage type, or compare packaging without being concerned with beverage type (Cleary, 2013; Saleh, 2016).

Existing LCAs for beverage packaging are not without flaws. An LCA is a complex undertaking and at every stage, many decisions are made in terms of allocation, data quality, what will and will not be included in the scope of the assessment and the impact categories that will be assessed. Quality of data is essential in creating a relevant, reliable and authentic LCA. The information that goes into the life cycle inventory may not be perfect; some data may be hard to find or measure or just be too ambiguous to include (Bjørn et al., 2018a). As each LCA has a different scope and boundaries, there will be different data requirements and different conclusions drawn. Several LCAs include transport as a key variable, modelling different product systems with different transportation distances as part of the comparison (Amienyo et al., 2013; Fachverband and Kartonverpackungen, 2007; Jelse et al., 2009). For these studies, assumed distances are applied and modelled, for other studies transportation has been decided upon by experts or given an average value (Meyhoff Fry et al., 2010). Meyhoff Fry et al, (2010) did not provide transportation distance for all product systems causing an imbalance when it comes to accuracy within the LCA. Transport can have high impact contributions due to fuel usage and emissions so inaccuracy could alter the results significantly. In some LCAs, transport is scoped out of the system boundary entirely due to the difficulty of accurately quantifying the distances the packaging would have to travel (Bjørn et al., 2018b; Curran, 2017a). For such reasons, this study will not include transportation.

It is clearly important to identify a relevant functional unit for beverage packaging due to the dimensions of each packaging type, as they are hollow vessels. A factor indicated by Cleary (2013) was that the mass of the container per amount of beverage contained is important, hypothetically 1 kilogram of PET may have more impacts than a kilogram of glass but the amount of PET to hold 1 litre of a beverage is far below the amount of glass needed for the same purpose (Cleary, 2013). Therefore, this study will use a functional unit based on the volume of beverage contained, not packaging weight.

Weighting, alongside normalisation, is a controversial step used in some LCAs. It involves assigning certain impacts a higher value than others, for example human health may be considered to have a greater weight than marine ecotoxicty in certain weighting sets (Bare et al., 2008). Saleh (2016) created a weighting set using a survey of experts to assign values to each category. There are standard weighting sets produced by different organisations, but some LCAs only briefly mention they have used weighting and do not always explain the weight given to impact categories, e.g. the study by Cleary (2013).

As results are only for a specific functional unit, a LCA cannot show any potential runaway processes that might occur when certain levels of outputs are reached (Rosenbaum et al., 2018). LCA data comes from different geographical areas - even within relatively small countries like the UK - and timeframes and cannot account for any unique characteristics of specific areas, such as existing contamination, temperature or other local emissions or outputs from other activities and products that might interact with those from the product system (Rosenbaum, 2017).

2.2 Purpose of study

LCAs often compare a few packaging types without specific concerns for the beverage the packaging will contain or study a variety of packaging types, particularly many variations of certain products, such as Tetra Pak variants. This study will consider three categories of beverage containers; pressurised drinks, unrefrigerated fruit juice and fresh milk containers. Two hypothetical 100% recycled packaging types will be included for glass and aluminium to indicate their near infinite recyclability only, as packaging made of 100% recycled materials are not common these hypothetical containers are purely for comparative purposes. Plastics' overall costs and benefits as beverage packaging are critically evaluated for pressurised beverages, fruit juice and milk.

The aims of the study were to: i) critically evaluate and review the costs, benefits, advantages and disadvantages of plastics as a beverage packaging material; and ii) identify, using life cycle assessment, if there are suitable replacements for beverage packaging constructed from plastics which have lower environmental impacts.

3. METHODS

Each beverage packaging category assessed had at least one form of plastic packaging. The software package OpenLCA (http://www.openlca.org/) was utilised as it is a free, widely-used open source program that is compatible with numerous impact methods. Data was collected from reliable sources such as existing LCA databases, peer-reviewed literature and scientific reports, and collated in a Microsoft Excel spreadsheet. The results of the LCA were compared within each drinks category to identify if there is a packaging type that has fewer environmental impacts than plastics. The study used ISO 14044:2006 and ISO14040:2006 standards as a framework. The data used, and data sources can be found in Appendix 1.All impacts were equally weighted.

3.1 Life cycle assessment stages

The four stages for LCA are outlined below; whilst they are separate stages many inform the others and there can be adjustment throughout the process (Mathews et al, 2018).

Goal and Scope Definition: ISO 14044:2006 states that the goal must be clearly defined with four statements needed in key areas. 1) Intended application; 2) Reason for carrying out study; 3) Audience; 4) If the results are used in publicly released comparative assertions. The scope consists of several qualitative and quantitative pieces of information that define what is and is not included in the study, the parameters of the study and which product systems were studied. Information such as the functional unit is decided upon in this stage.

Inventory Analysis: Collection and documentation of data gathered in accordance to the needs of the goal and scope. Data is collected, validated, allocated to its associated processes and some data often has to be converted to the functional unit, it was aggregated for the analysis, in this study it was stored in Microsoft Excel. For the inventory analysis product systems were collated within openLCA, a product system includes all the gathered data involved in the product's life cycle organised in such a way that it can then be used in the LCA.

A product system includes the processes for the inputs and outputs of the system, for example for a plastic bottle petroleum must be extracted, so the process to extract the petroleum would be included in the product system with the petroleum as the 'flow' into the next process.

Impact Assessment: This is the stage where the study moved beyond individual flows and processes and assessed that the impacts of the product system were in accordance to the goal and scope. Impact categories were chosen that were relevant to the goal and scope of the study and the choices must be justified. ISO14040:2006 states that these impact categories must be listed explicitly in the study. Using openLCA the life cycle impact assessments were generated for each category, this stage was largely automated and involved ensuring that all data was correct, impact categories were correctly chosen and that there were no technological errors (Rosenbaum et al., 2018). It was in this stage that data was assessed for the impacts of each product system for each impact category.

It is important to understand through this stage and the interpretation stage that what the life cycle impact analysis shows is potential or theoretical impacts. To meet the ISO Standards for LCAs there were three mandatory steps for the life cycle impact assessment stage:

i. Selection of impact categories, indicators and characterisation modules, this step is completed by choosing from existing LCIA methods.
ii. Classification of the LCI results, assigning them to impact categories based on what their known impacts are, this is typically done by the software automatically.
iii. Characterisation of the results, the software will quantify how much each of the inventory flows are contributing to the impact categories.

Interpretation: The ISO standard gives less in terms of guidance on this stage, but the aim of the interpretation stage is to examine the results to be able to report any findings, recommendations or conclusions (see Discussion). The optional weighting and normalisation step of LCA was not preformed. Weighting is controversial because unless the LCA has a specific purpose, such as examining impacts on human health, it can be difficult to justify what weight impacts could have in relation to each other. This study will not use weighting, mainly as it limits a study's ability to be used as a comparative piece of work and justification for specific weighting is highly subjective (Bettens and Bagard, 2016; Jelse et al., 2009).

3.2 Data

Data was obtained from a range of authoritative sources, as shown in Appendix 1. Many datasets were examined but only the most relevant and robust were selected. Data was gathered or adapted for the functional unit. The weight of each packaging container was estimated by calculating an average of the weights of 10 different examples of each packaging type. For the aluminium can, ten 500 ml cans were weighed and a theoretical litre can was modelled. When data from different sources or datasets was used, it was carefully processed in order to avoid double counting of any materials, impacts or outputs in the product system. As datasets for processes in glass manufacture were not available, data from many sources had to be adapted to model these processes.

Data was assembled for three categories of beverage packaging each containing a different liquid: non-refrigerated fruit juice, fresh milk and pressurised drinks such as cola. Each of these three categories included the most commonly used plastic packaging used for the beverage as a baseline as well as other commonly used packaging types (Table 1). All categories included glass bottles so a

TABLE 1: Beverage packaging categories and types of packaging assessed.

Category	Plastic packaging	Other packaging	Other packaging	Hypothetical recycled packaging	Hypothetical recycled packaging
Fruit Juice	PET Bottle	Tetra Pak	Glass Bottle	100% Recycled Glass	
Milk	HDPE Bottle	Milk Carton	Glass Bottle	100% Recycled Glass	
Pressurised	PET Bottle	Aluminium Can	Glass Bottle	100% Recycled Glass	100% Recycled Aluminium Can

TABLE 2: Processes scoped in and scoped out of product systems.

Scoped In	Scoped Out
Extraction of virgin materials	Transport – for all stages of production including virgin material extraction and end of life treatment.
`Manufacture of packaging	Filling
End of life treatment -including landfill, burning and recycling for each product according to UK rates of disposal	Beverage manufacture

hypothetical 100% recycled glass bottle was included to demonstrate the ability of glass to be constantly recycled without degrading as plastic does. The same was completed for hypothetical 100% recycled aluminium cans in the Pressurised Beverage Category.

3.3 Functional unit, goal and scope

The functional unit was the packaging required to hold 1 litre of a specific beverage. This was modelled as one single container for each functional unit, so the aluminium can was modelled as a hypothetical can that could hold 1 litre. This is to keep consistency of scale across the LCA; if the PET bottle unit used was a 1 litre bottle against ten 100 ml glass bottles the glass would hold the same amount of fluid but the weight of glass would be far greater than for a single 1 litre container. In accordance with the requirements to present the scope of the study Table 2 shows the scoped in and out processes for product systems.

Cut-offs are points beyond which parts of the product system are considered too small or insignificant to be counted. In some LCAs this is when the material in question constitutes less than 5% of the finished product and in others it is 1% (Curran, 2017a). For other parts of the product system it might relate to how much energy they require or contribute; below a certain threshold they can be considered irrelevant to the goal and scope of the specific study (Curran, 2017a). For this study, any process that contributes less than 1% of material or energy to the product was not included in the scope of this study.

Although the methodology utilised may be universally applied, the scope of this study relates to UK practices and uses relevant data. We used data from 2010 onwards unless no reliable and robust data was available and older data was necessarily adapted. Two different product system 'types' were studied due to the inclusion of the two hypothetical 100% recycled product systems. Figure 1 shows the product system and scoped in and scoped out processes for the beverage packaging that is not 100% recycled. Figure 2 shows the product system and scoped in and scoped out processes for the 100% recycled beverage packaging product systems.

3.4 Allocation and impact categories

Allocation is the process by which each process and output is associated with the correct product, flow and the like. Correct allocation was achieved by consulting the literature closely for each product and ensuring that when data was taken from different sources there was no double counting. There are many different impact categories available; each uses indicators allowing a prediction of the impacts of the product system. The categories for this study are shown in Table 3.

3.5 Assumptions and limitations

Note that some data had to be adapted using reasonable assumptions made, particularly where complete process datasets could not be found or datasets from other countries had to be adapted with UK energy usage

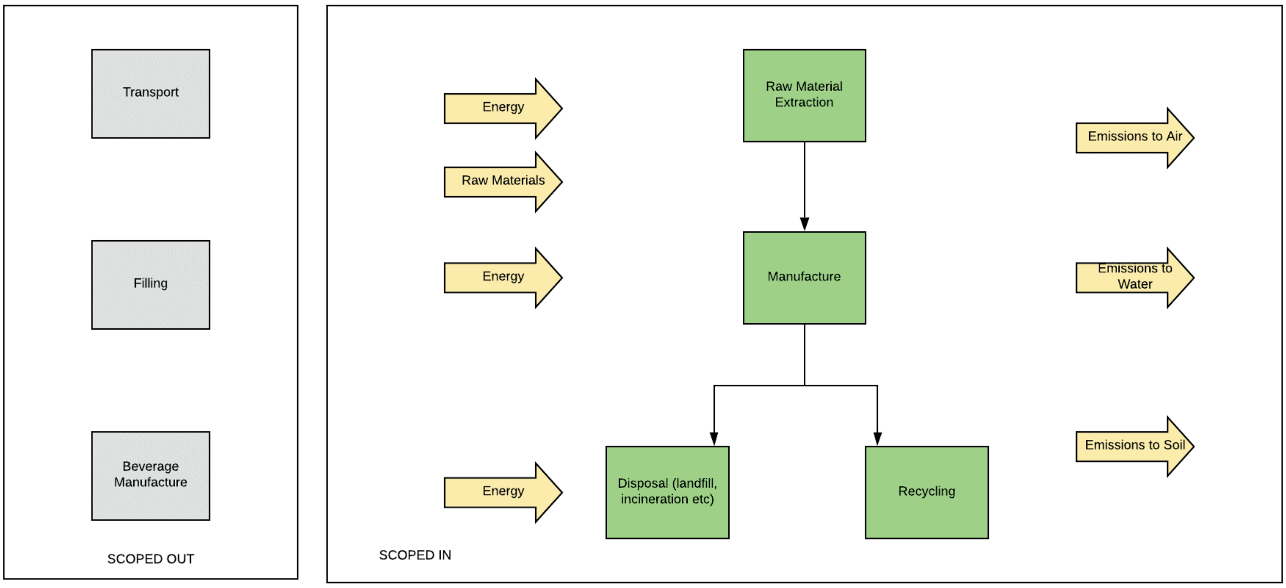

FIGURE 1: Product system and scoped out processes for beverage packaging made from virgin materials.

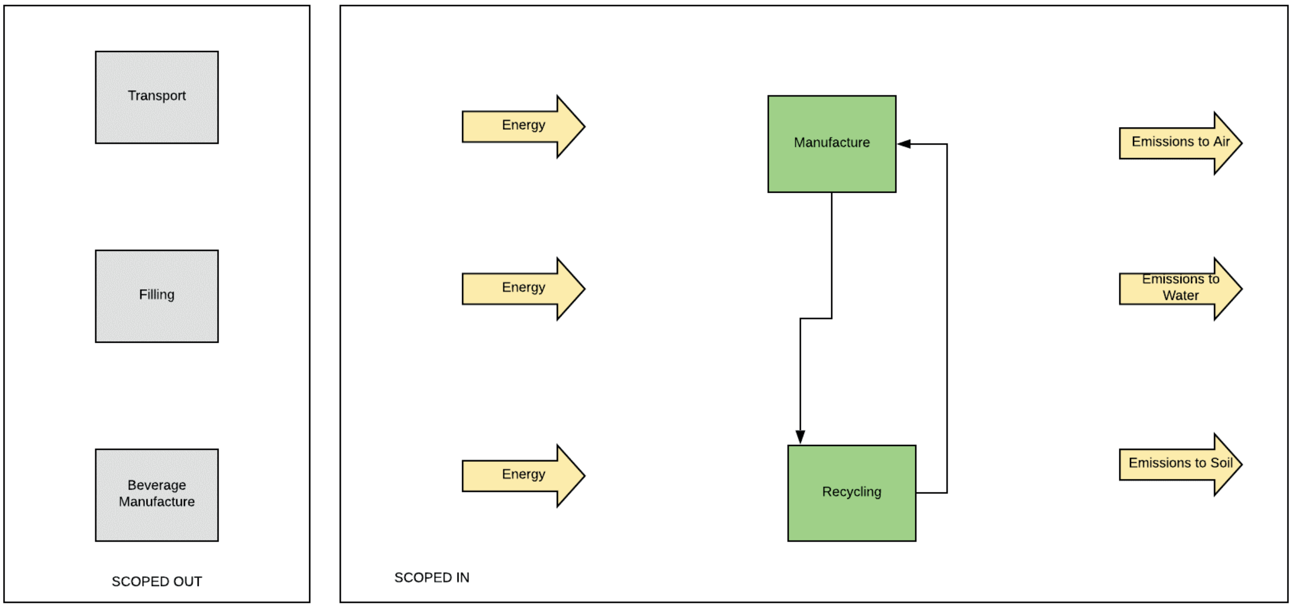

FIGURE 2: Hypothetical product system and scoped out processes for beverage packaging made from 100% recycled materials.

or flows. Assumptions had to be made for the Tetra Pak product system as accurate manufacturer's data could not found so data for milk carton production was adapted.

4. RESULTS

4.1 Introduction

The results of all three categories showed clear differences in each beverage packaging's impact within each CML impact category. All three drinks categories had glass bottles as one of the packaging types and in all three the virgin glass bottle had the highest impacts in most CML categories, with PET bottles showing maximum indicators in the two beverage categories in which it was present.

When showing the data in graphical form, the maximum indicator has a value of 100% and each other product's indicator is shown relative to the maximum indicator. To ascertain which beverage packaging types in each beverage category were the most impactful, each beverage packaging was ranked for each impact category, with the initial table showing the ranks and an additional table showing the collated results. The lowest scoring beverage packaging is the most impactful in that category. Whilst it is often easy to see from the maximum indicators that are the most impactful in a category, how the other categories relate to each other in a cumulative fashion can be harder to define.

Using these ranked scores, the most impactful and least impactful beverage packaging types overall for each category were quantified. These scores were used to identify which packaging types were most impactful across categories.

TABLE 3: CML (Institute of Environmental Sciences, Leiden University) impact categories and their descriptions.

CML Impact Category	Description of Impact Category
Acidification Potential – Average Europe	The potential of the product system to cause acidification
Climate Change - GWP 100	The potential of the product system to impact climate change through 'global warming potential'
Depletion of Abiotic Resources – elements, ultimate reserves	The loss of resources due to the product system such as chemical elements and overall reserves of resources
Depletion of Abiotic Resources – fossil fuels	The loss of fossil fuel resources due to the product system
Eutrophication – generic	The potential of the product system to cause eutrophication in all waters
Freshwater Aquatic Ecotoxicity	The potential of the product system to have toxic outputs into freshwater systems
Human Toxicity	The potential of the product system to have toxic impacts on human health
Ozone Layer Depletion	The potential of the product system to deplete the ozone layer in its current state
Photochemical Oxidisation	The potential of the product system to generate NOx and cause 'summer smog' due to air pollution
Terrestrial Ecotoxicity	The product system's potential to have toxic impacts on terrestrial environments
Marine Aquatic Ecotoxicity	The product system's potential to have toxic impacts on marine environments

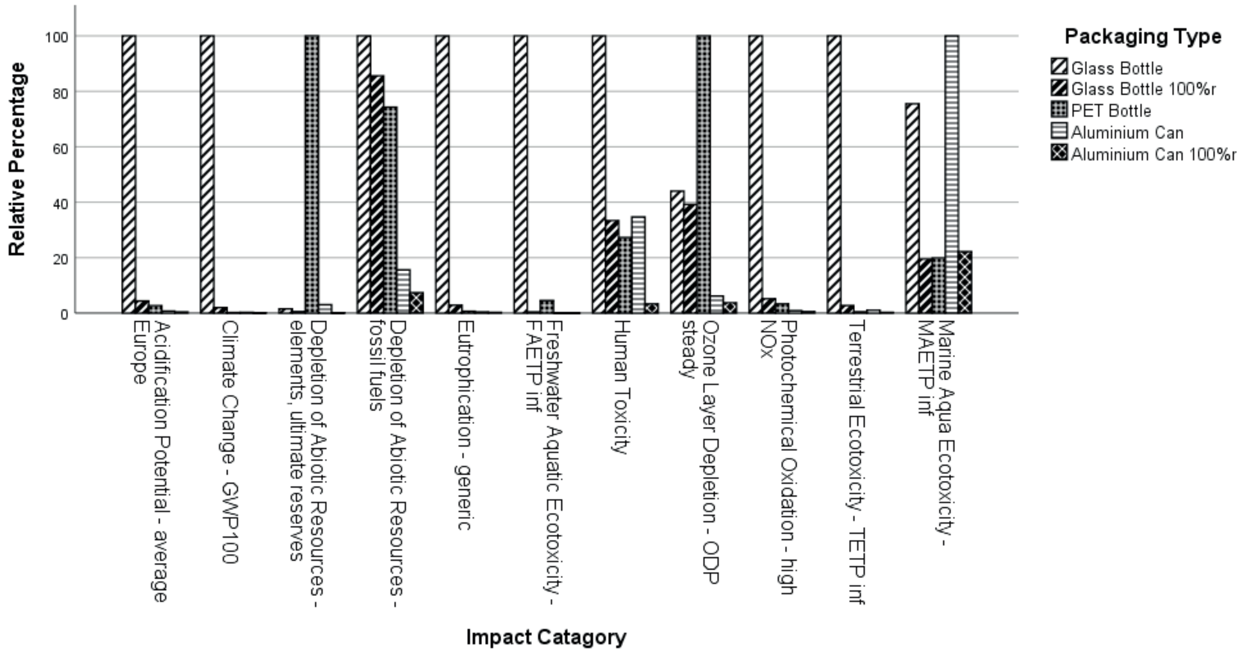

FIGURE 3: Relative results from pressurised beverage packaging category using CML impact methods. Product with the maximum indicator is set to 100% and other variants displayed in relation to this result indicating differences from maximum impact indicator.

4.2 Pressurised Beverage Packaging Life Cycle Assessment

The Pressurised Beverage Packaging Category LCA had the highest number of product systems assessed, including the PET plastic bottle (entirely plastic packaging) and a hypothetical 100% recycled glass bottle and a hypothetical 100% recycled aluminium can. Results from the life cycle assessment are presented in Figure 3.

The recycled containers were assessed to have lower impacts overall than their counterparts made of virgin materials. The 100% recycled aluminium had the lowest impacts overall in this category with the lowest impacts in all categories except 'Marine Aquatic Ecotoxicity' (lower than its virgin counterpart). The beverage packaging type with the most maximum indicators was the virgin glass bottle, the maximum in eight of the twelve categories.

The PET bottle scored the maximum indicator for two categories; 'Depletion of Abiotic Resources – elements, ultimate reserves' and 'Ozone Layer Depletion'. The virgin aluminium can was the maximum indicator for the 'Marine Aquatic Ecotoxicity' category.

Within the Pressurised Beverage Category (ranks shown in Table 4, combined results shown in Table 5), the glass bottle was the highest ranked overall for environmental impacts across all impact categories with the 100% recycled glass bottle second. The 100% recycled aluminium can is the lowest ranked gaining the lowest score of 5 in all categories bar one, Marine Aquatic Toxicity, where it

TABLE 4: Ranks for each beverage packaging in the Pressurised Beverage Packaging Category across all eleven CML impact categories; 1 is the highest rank for the most impactful packaging, 5 is the lowest for the least impactful..

Beverage Packaging Type	Acidification Potential - Average Europe	Climate Change - GWP 100	Depletion of Abiotic Resources elements, ultimate reserves	Depletion of Abiotic Resources - Fossil fuels	Eutrophication - generic	Freshwater Aquatic Ecotoxicity	Human Toxicity	Ozone Layer Depletion	Photochemical Oxidisation	Terrestrial Ecotoxicity	Marine Aquatic Ecotoxicity
Glass Bottle	1	1	3	1	1	1	1	2	1	1	2
Glass Bottle 100% R	2	2	4	2	2	3	3	3	2	2	5
PET Bottle	3	3	1	3	3	2	4	1	3	4	4
Aluminium Can	4	4	2	4	4	4	2	4	4	3	1
Aluminium Can 100% R	5	5	5	5	5	5	5	5	5	5	3

TABLE 5: Collated rank scores for each beverage packaging type in the Pressurised Beverage Category.

Beverage Packaging Type	Ranked Score (lowest value is most impactful)
Glass Bottle	15
Glass Bottle 100%R	29
PET Bottle	32
Aluminium Can	35
Aluminium Can 100%R	53

scored 3. Notably this is the impact category in which the virgin aluminium was identified as the most impactful beverage packaging. Both categories of aluminium cans were less impactful overall than the PET plastic bottle according to the ranked scores. The recycled versions of the glass bottle and the aluminium can both came second to their virgin counterpart.

4.3 Fruit Juice Beverage Packaging Life Cycle Assessment

The Fruit Juice Beverage Packaging Category was the only one to assess the impacts of Tetra Pak. Results from the LCA are shown in Figure 4. This category shows similar results to the Pressurised Beverage Category, with glass being assessed to have the highest impacts overall. The PET bottle has the highest impacts in the same two categories as in the 'Pressurised Beverage' category; 'Depletion of Abiotic Resources – elements, ultimate reserves' and 'Ozone Layer Depletion'. The key difference is the inclusion of the Tetra Pak container, which shows comparatively very low impacts compared to the other product systems even with no recycled content, the only categories where Tetra Pak has noticeable impacts are both the marine and freshwater toxicity and fossil fuel depletion.

Within the fruit juice beverage category, ranks shown in Table 6, the combined results shown in Table 7, the glass bottle was the highest ranked overall for environmental impacts across all impact categories with the 100% recycled glass bottle the second most impactful. The Tetra Pak carton is the lowest ranked gaining the lowest score of 4 in all but two categories, scoring 3 in Depletion of 'Abiotic Resources elements, ultimate reserves' and scoring 2 in 'Freshwater Aquatic Ecotoxicity'. The PET bottle was the third most impactful of the four ranked beverage packag-

TABLE 6: Ranks for each beverage packaging in the fruit juice beverage packaging category across all eleven CML impact categories; 1 is the highest rank for the most impactful packaging, 5 is the lowest for the least impactful.

Beverage Packaging Type	Acidification Potential – Average Europe	Climate Change - GWP 100	Depletion of Abiotic Resources elements, ultimate reserves	Depletion of Abiotic Resources – fossil fuels	Eutrophication - generic	Freshwater Aquatic Ecotoxicity	Human Toxicity	Ozone Layer Depletion	Photochemical Oxidisation	Terrestrial Ecotoxicity	Marine Aquatic Ecotoxicity
Glass Bottle	1	1	2	1	1	1	1	2	1	1	1
Glass Bottle 100% R	2	2	3	2	2	4	2	3	2	2	3
PET Bottle	3	3	1	3	3	3	3	1	3	3	2
Tetra Pak	4	4	3	4	4	2	4	4	4	4	4

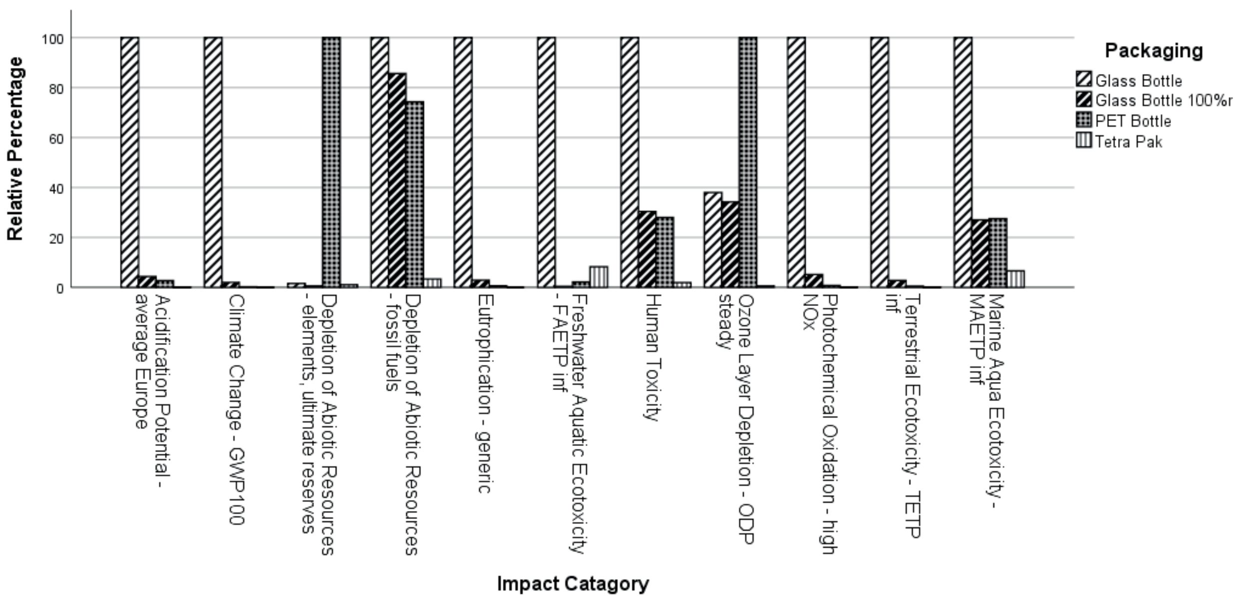

FIGURE 4: Relative results from Fruit Juice Packaging category CML impact methods. Maximum indicator is set to 100% and other variants displayed in relation to this result indicating differences from maximum impact indicator.

TABLE 7: Collated rank scores for each beverage packaging type in the Fruit Juice Beverage Category.

Beverage Packaging Type	Ranked Score (lowest value is most impactful)
Glass Bottle	13
Glass Bottle 100%R	27
PET Bottle	30
Tetra Pak	41

ing types in this category, and did score the highest ranks in two categories 'Abiotic Resources elements, ultimate reserves' and 'Ozone Layer Depletion' the same impact categories the PET bottle gained the highest rank for in the pressurised beverage packaging LCA rankings.

4.4 Milk Beverage Packaging Life Cycle Assessment

The milk beverage packaging category LCA was the only one that included the HDPE plastic bottle as the baseline plastic. It also was the only LCA to include the milk carton and HDPE plastic bottle; this LCA had the hypothetical 100% recycled glass bottle as a recycled option (Figure. 5). Both glass bottles were the most and second most impactful packaging type in all categories in this LCA, the HDPE bottle was the third most impactful packaging type in all the impact categories. Unlike the PET bottle in the other beverage categories the HDPE bottle did not have a higher impact in any impact category than the glass bottles. The milk carton has low overall impacts in all categories.

Within the Milk Beverage Category, ranks shown in Table 8, the combined results shown in Table 9, the glass bottle was the highest ranked overall for environmental impacts across all impact categories with the 100% recycled glass bottle the second most impactful. The milk carton is the lowest ranked in all categories.

The HDPE plastic bottle scored 3 in all categories aside from 'Depletion of Abiotic Resources elements, ultimate reserves' where it scored 4, it did not have the highest impact in any impact category unlike the PET plastic bottle in the two other categories.

4.5 Most Overall Impactful Beverage Packaging

Different processes within each product system had the highest contributions to the overall impacts in each

TABLE 8: Ranks for each beverage packing in the Milk Packaging Category across all eleven CML impact categories; 1 is the highest rank for the most impactful packaging, 5 is the lowest for the least impactful.

Beverage Packaging Type	Acidification Potential – Average Europe	Climate Change - GWP 100	Depletion of Abiotic Resources elements ultimate reserves	Depletion of Abiotic Resources – fossil fuels	Eutrophication - generic	Freshwater Aquatic Ecotoxicity	Human Toxicity	Ozone Layer Depletion	Photochemical Oxidisation	Terrestrial Ecotoxicity	Marine Aquatic Ecotoxicity
Glass Bottle	1	1	1	1	1	1	1	1	1	1	1
Glass Bottle 100% R	2	2	2	2	2	4	2	2	2	2	2
HDPE Bottle	3	3	3	3	3	3	3	3	3	3	3
Milk Carton	4	4	4	4	4	2	4	4	4	4	4

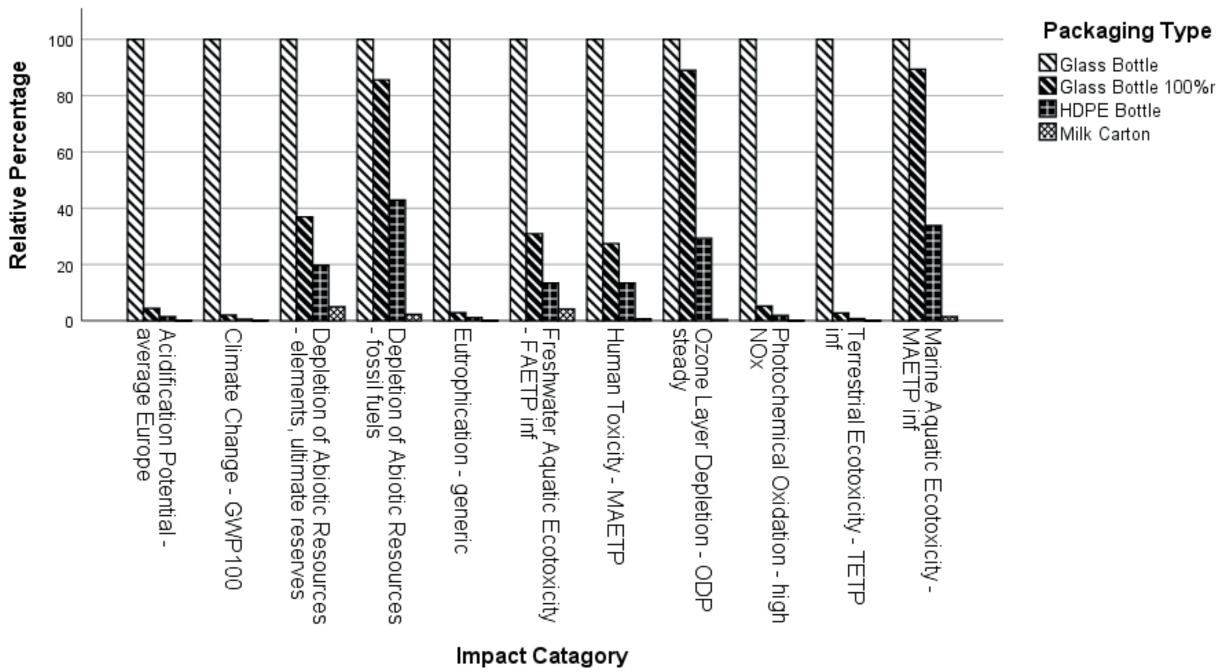

FIGURE 5: Relative results from the Milk Packaging category CML impact methods. Maximum indicator is set to 100% and other variants displayed in relation to this result indicating differences from maximum impact indicator.

TABLE 9: Collated rank scores for each beverage packaging type in the Milk Beverage Category.

Beverage Packaging Type	Ranked Score (lowest value is most impactful)
Glass Bottle	11
Glass Bottle 100%R	24
HDPE Bottle	33
Milk Carton	42

impact category. Understanding where the impacts come from may demonstrate which processes need improvement or mitigation to minimise impacts. In the ranked scores for all its beverage categories, virgin glass had the highest impacts across all categories, the two plastic bottles both came third in their categories. However, the pressurised beverage packaging category had five packaging types unlike the other two, but in all LCAs plastics ranked below the two glass packaging types and above the other alternatives to plastics.

As glass was overall the most impactful beverage packaging type, understanding exactly where the high impacts are originating could help in minimising them in future packaging. As virgin glass bottles had maximum impacts in all CML categories across the beverage categories all impact categories are included in Table 10.

The virgin glass bottle overall had the highest impacts coming from the glass melting process. Grouped in this process were the extraction of composite materials that made up the glass, the energy to melt the materials and all emissions from the melt. This process releases a high level of gases particularly carbon dioxide (and its equivalents) which are the indicators for global warming in the CML impact category 'Climate Change'. In both the beverage categories in which the PET bottle product system was assessed it was the maximum indicator for the same two CML impact categories in Table 11. In both the impact categories for which PET plastic was the maximum indicator in the pressurised beverage and fruit juice categories, the PET granule production was the highest contributor to the impact. PET granule production in these product systems included material extraction.

4.6 PET and HDPE Plastic Bottle Comparison

The PET plastic bottle would appear more impactful than the HDPE bottle. However, as life cycle assessment is comparative only to the other product systems in its specific assessment, this could be an incorrect assumption. To assess if there is a significant difference between the two plastic bottles modelled in this study a further LCA was conducted for the two plastic bottles.

The results of the LCA conducted for the HDPE and PET bottles (see Figure 6) indicate that PET plastic bottles have a higher impact overall and in every CML impact category than the HDPE bottle. This demonstrates that not all plastics have the same level of impacts and that even packaging choices within plastic bottle options can vary the potential impact of the packaging.

As PET is the more impactful of the two plastics, and the more abundant, understanding the sources of the impacts within the processes, as with the glass bottles, can help understand which processes may need improving to minimise impacts.

TABLE 10: Highest contributing processes to virgin glass bottles' product system impacts in CML impact categories, Table does not indicate percentage contributed compared to other product systems in categories.

Impact Category	Highest Contributor	% Contributed	Indicator
Acidification Potential	Glass Melting	97.1	kg SO_2 eq.
Climate Change	Glass Melting	99.0	kg CO_2 eq.
Depletion of Abiotic Resources -elements, ultimate reserves	Waste Incineration	50.6	kg antimony eq.
Depletion of Resources – fossil fuels	Electricity Demands	56.8	MJ
Eutrophication	Glass Melting	98.4	kg PO_4 eq.
Freshwater Aquatic ecotoxicity	Glass Melting	99.7	kg 1,4-dichlorobenzene eq.
Human toxicity	Glass Melting	78.4	kg 1,4-dichlorobenzene eq.
Marine Aquatic ecotoxicity	Glass Melting	85.6	kg 1,4-dichlorobenzene eq.
Ozone Layer Depletion	Electricity Demands	59.1	kg CFC-11 eq.
Photochemical oxidation	Glass Melting	96.6	kg ethelyne eq.
Terrestrial ecotoxicity	Glass Melting	98.8	kg 1,4-dichlorobenzene eq.

TABLE 11: Highest contributing processes to PET bottles' product system impacts for CML impact categories in which they had highest impacts. Table does not indicate percentage contributed compared to other product systems in categories.

Impact Category	Highest Contributor	% Contributed	Indicator
Depletion of Abiotic Resources -elements, ultimate reserves	PET Granule Production	99.6	kg antimony eq.
Ozone Layer Depletion	PET Granule Production	77.0	kg CFC-11 eq.
Depletion of Abiotic Resources -elements, ultimate reserves	Waste Incineration	50.6	kg antimony eq.

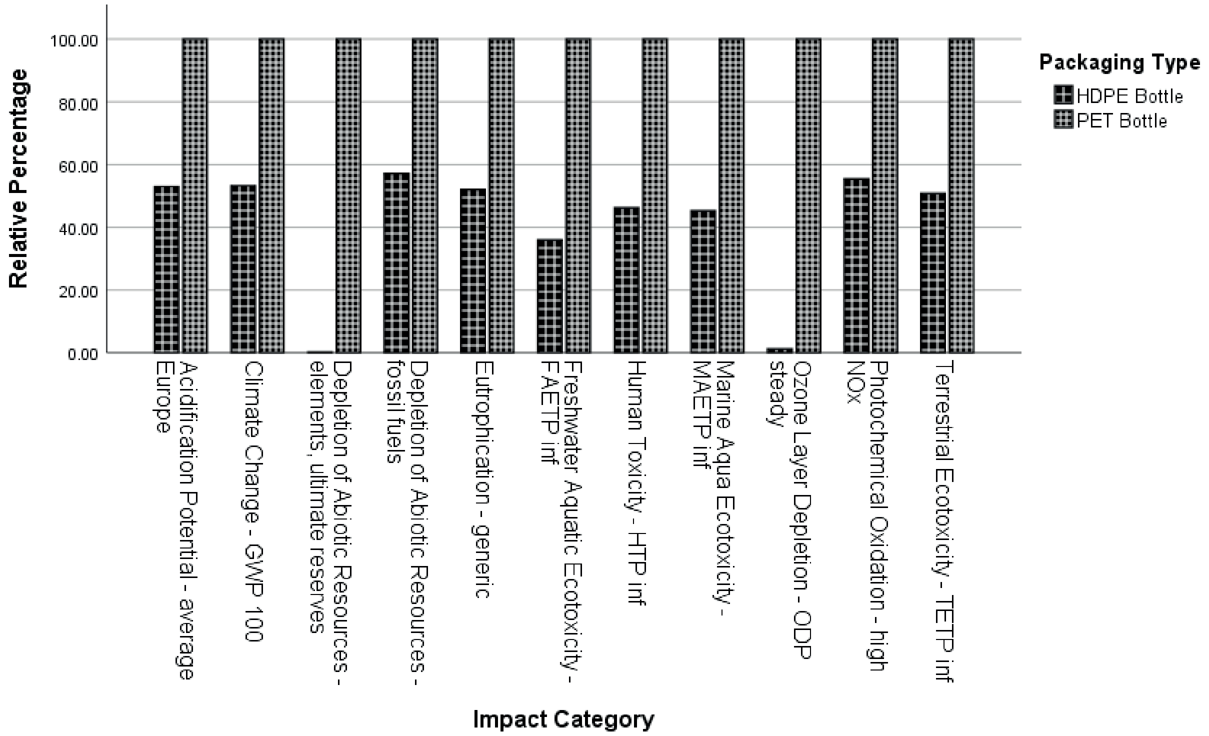

FIGURE 6: Results for HDPE bottle and PET bottle life cycle assessment using CML impact methods. Maximum indicator is set to 100% and other variants displayed in relation to this result indicating differences from maximum indicator.

5. DISCUSSION

5.1 Key Results

The results indicate that there are less impactful packaging types than the two modelled plastic bottles, PET and HDPE in all three beverage packaging categories. The replacement packaging types in each category are shown in Table 12.

In each category the glass and recycled glass bottle were always the most impactful packaging type. Both plastics always came below both glass packaging types but above the other modelled packaging types in each category. PET plastics showed maximum impacts in 'Depletion of Abiotic Resources -elements, ultimate reserves' and 'Ozone Layer Depletion' CML impact method categories, indicating there are still considerable costs to using plastics.

Whilst the weights of all the packaging types were fairly consistent, the glass bottles varied greatly in weight with the heaviest weighing 980g. This variation in weight would likely mean that the bottles at each end of the range would present different impact results (an average was taken for this study). Another factor was that energy had to be carefully allocated. Incorrect energy allocation could lead to certain product systems, such as a milk cartons, having far higher impacts in certain categories. This links back to the earlier discussion on allocation; every effort was made to ensure all data used was suitable for the UK, and if not from a UK source modified for UK energy mixes.

The decision to not use weighting allowed all impact categories to be given equal ranks so level of overall impact was clear to understand. If this study had a specific focus, such as finding packaging with lower water demands then weighting would have been appropriate, but this study sought least impactful overall packaging without contending with the complexities of each impact and the controversy of placing impacts above each other.

In the Pressurised Beverage Packaging category, packaging was chosen that was commonly used for pressurised beverages. PET plastic bottles are the most common packaging in this category (Hahladakis et al., 2018). Pressurised beverage packaging has to be able to withstand pressurisation meaning the Tetra Pak and milk carton types of packaging were unsuitable. It is likely weight was a factor in the results of the assessment, the lightest packaging types, the aluminium can, and recycled aluminium can were the least impactful, although the virgin aluminium can only ranked slightly better than the PET bottle.

This differs from results of other LCAs on PET plastic bottles and aluminium cans. Some have found aluminium to be more impactful than PET plastic, especially when weighting has taken place. Others found aluminium cans to have a far higher climate change (GWP impact) than PET plastics (Amienyo et al., 2013; Franklin Associates, 2009). This may be due to the scoping out of transport as much bauxite is mined in Australia and transport to and from that continent would have high fuel demands (Amienyo et al., 2013). Many LCAs base their measurement on 330 ml aluminium cans whereas we hypothesised one can that would hold 1 litre of the beverage, thus using significantly less material by comparison. The dataset used for the manufacture of the aluminium sheet for the aluminium can sources the majority of the energy used in production from hydropower, which is far less damaging to the climate, where the PET granule manufacture uses a majority of oil, coal and gas energies.

TABLE 12: Highest contributing processes to PET bottles' product system impacts for CML impact categories in which they had highest impacts. Table does not indicate percentage contributed compared to other product systems in categories.

Beverage Packaging Category	Packaging Types to Replace Plastics
Pressurised	Aluminium can or 100% recycled aluminium can
Fruit Juice	Tetra Pak
Milk	Milk Carton

The fruit juice category showed similar results to the pressurised beverage packaging category. The Tetra Pak's low impacts are due to the fact that around 75% of the Tetra Pak packaging is made of paper, which unlike the other materials used in beverage packaging, is renewable (Fachverband and Kartonverpackungen, 2007; Jelse et al., 2009).

The Tetra Pak may show low impacts due to the far lower energy demands and the highest proportion of the container being made of paper which does not require mining or other similar processes to extract. However, the recycling of Tetra Pak is problematic, whilst all the component parts can be recycled this is an energy intensive process as the composite material must be separated then the additional recycling processes must take place (TERI, 2010; Tetra Pak, 2012). The plastic and paper materials which make up most of the Tetra Pak are not indefinitely recyclable due to the shortening of the paper fibres and polymer chains, additionally this recycled material is often downcycled into less recyclable products (TERI, 2010).

The impacts on both water ecotoxicity categories for the Tetra Pak, the categories it has the most noticeable impacts on, is from the polythene and aluminium as characterised by openLCA, the paper content of the packaging seems to have little impact on any CML impact category. However if a different impact category set was used that considered water demands it may have greater impacts due to growing trees.

In the Milk Beverage Category both the HDPE bottle and milk carton products showed far lower impacts than both the virgin glass and recycled glass bottles. Both are over fifteen times lighter than the glass bottles modelled so use far less material. Both also require far lower temperatures to be manufactured, the milk carton only using heat for the LDPE plastic coating over the main paper body of the packaging. The milk carton has less environmental impacts than the HDPE bottle in this category as it is made predominantly of paper and is only 5% plastic.. The results of this study align with the findings of the extensive literature review of by Von Falkenstein et al., (2010), which found most LCAs assigned the lowest impacts to carton type packaging particularly for climate change and acidification potential.

The milk carton has similarly problematic recyclability issues to the Tetra Pak, whilst it does not include aluminium so separation of the composite materials includes less energy, there is still a limited number of councils that collect cartons, and only one UK carton recycling plant (WRAP, 2017). There are similar issues to the Tetra Pak with the materials having limited recyclability and will often be made into downcycled materials and products. Similarly to the Tetra Pak, the milk carton shows most impacts on the marine and freshwater ecotoxicity categories; this is due to the LDPE coating on the packaging. However, as the models do include end of life treatment these packaging types still rank far lower in impacts than other packaging in their categories even with their low recycling rate.

Whilst this study was not conducted using data directly obtained from the manufacturer or using the ecoinvent database, as many of the studies in this area do, instead using and adapting freely available data, similar results have been reached (Amienyo et al., 2013; Cleary, 2013; Eriksson et al., 2009; Meyhoff Fry et al., 2010; Saleh, 2016). This is important to note due to the complexity of LCAs and the general reliance of practitioners on these sources of information. Our results are consistent with the general outcomes from other LCAs obtained using information accessible to those without the means to access expensive or privileged sources of information.

When the PET plastic bottle and the HDPE plastic bottle were compared the HDPE's lower impact scores, and relative lightness compared to the PET suggests light weighting may be a solution to be explored for PET bottles, as reducing the material used will reduce the impact per container. This could be considered for all packaging types; lighter containers are generally less impactful and the heaviest packaging, glass, was always the most impactful overall in all packaging categories.

Other studies have found virgin and recycled glass to have a high level of impacts relative to other packaging (Accorsi et al., 2015; Cleary, 2013; Saleh, 2016). Other studies have also found plastics to have high comparative impacts to ozone depletion and abiotic resource depletion compared to other packaging types, as found in this study (Fachverband and Kartonverpackungen, 2007; Jelse et al., 2009; Meyhoff Fry et al., 2010). The other beverage containers tested showed similar results to previous LCAs that tested their product type, except for aluminium cans showing high Aquatic ecotoxicity in previous studies. Tetra Pak and milk cartons had lower impacts than other beverage containers except those that modelled containers not included in this study, such as pillow pouches for milk; (Fachverband and Kartonverpackungen, 2007; Jelse et al., 2009; Meyhoff Fry et al., 2010; Saleh, 2016). The only assessed beverage packaging type that did not show similar results, as previously discussed, was the aluminium can when compared to the PET bottle.

One of the study's aims was to critically evaluate the costs, benefits, advantages and disadvantages of plastic as a beverage packaging material. The results of the LCA yield some valuable results in this regard. Plastics are less impactful on a single use basis than glass (even 100% recycled glass), probably because it is considerably lighter and requires less energy to manufacture, so there are benefits from using plastics as packaging materials. The HDPE plastic showed no maximum indicators in any category, showing it could be a lower impact alternative to glass.

When the PET and HDPE bottles were compared, the results for impacts in the 'Depletion of Abiotic Resources -elements, ultimate reserves' and 'Ozone Layer Depletion' showed how high the impacts from PET bottles are in these categories. HDPE in the Milk Beverage Packaging Category did not show the lowest impacts in those categories. The HDPE bottle is lighter than the PET bottle and the granules require less energy to manufacture (see Appendix 1). The differences in the manufacturing processes of these two polythene-based thermoplastics may factor into the vast comparative difference between their impacts. A higher amount of crude oil is used to manufacture the PET granules than the HDPE granules per kilogram which likely contributes in part to the impact gap between the two (Plastics Europe, 2018).

Unlike the HDPE bottle, the PET bottle showed up as a maximum indicator in some CML impact categories. The categories in which PET was maximum indicator in the pressurised and fruit juice categories ('Depletion of Abiotic Resources -elements, ultimate reserves' and 'Ozone Layer Depletion') are in line with other studies (Accorsi et al., 2015; Morales-Méndez and Silva-Rodríguez, 2018). The biggest factor in both of these impacts is the production of plastic granules, which encompasses inputs and outputs of mining raw materials, processing, and preparation for manufacture. Ozone depletion has been linked to plastic production by several other studies, in some studies this is found to be due to chemicals used in the blowing process, however, this stage is not part of granule production (Morales-Méndez and Silva-Rodríguez, 2018). Therefore, the costs of PET plastics as a beverage packaging are high in these processes as both contribute to the depletion of resources, ultimate reserves and fossil fuels; this is unsustainable.

Plastics are widely recycled, unlike some of the other modelled packaging such as Tetra Pak, and rPET and rHDPE granules are now available to be used in bottle manufacture, however they do not alter the manufacture process which glass cullet does by lowering heat or energy demands (Meyhoff Fry et al., 2010). The only potential reduction in impacts from using recycled plastics granules is in relation to minimising extraction of raw materials and initial material production. However, as plastics are not indefinitely recyclable, raw materials will always need to be extracted to create new bottles.

Plastics have many benefits as a packaging material, being light, durable and less fragile than glass and needing less raw materials per bottle than some of their alternatives (Andrady and Neal, 2009). In some cases, such as bottled water supplied to areas with unsanitary water stress, plastic bottles are vital, and plastics can be used for many beverage types. However as demonstrated in this study plastic bottles can have high environmental impacts compared to some other beverage packaging, and in a few impact categories are worse than the far heavier glass.

Glass bottles, both virgin and recycled had high impacts compared to all other product systems, however this does not consider the potential of reusing the glass bottles. Many communities no longer have milk delivered, but previously this was common across the United Kingdom (Campbell, 1994). These glass bottles were reused multiple times, this is referred to as its trippage rate, for milk bottles this is between 20 and 40 cycles of reuse before the glass would have to be disposed of or recycled (Campbell, 1994). This would imply, roughly, that one glass bottle, when reused would be able to hold the equivalent of between 20 – 40 single use bottles. If this glass were then recycled it could be less impactful per use than the HDPE plastic bottles. The LCA by Mata and Costa, (2001) found that reused glass bottle schemes had far lower impacts in all tested impact categories scoped into that study, than non-returned glass systems. Whilst this study was undertaken under the former ISO standards, it still indicates that reuse of glass would be beneficial, especially when compared to single use glass bottles. This study is also supported by the findings of Simon et al., (2016), which found that if a glass bottle was reused it would reduce the environmental impacts of the container, however reuse only continued to net benefits for between 7 – 9 uses where it plateaus and no further significant benefits are generated. Changing to a reuse scheme would entail far more complex logistics, even if door to door delivery were not put into place, potentially a deposit scheme or personal refilling would have to be set up which would be less convenient than door to door delivery.

People could be encouraged to move away from buying beverages that could be made at home. Whilst generally people cannot produce animal milk or fruit juice at home, people can already make pressurised beverages with systems such as SodaStream's sparkling water maker to which flavoured syrups can be added to make at-home versions of popular sparkling drinks (Sodastream, 2019). This makes use of reusable, durable plastic bottles rather than single use beverage packaging. Alongside at-home solutions, common sparkling drinks such as cola, are already frequently provided in fast food restaurants on tap at 'free refill' stations, these could also be utilised 'on the go' by people bringing their own container, much as increasingly more people do with refillable coffee cups (Smithers, 2018). The United Kingdom's government has put forward the idea of a tax on disposable coffee cups which has, for some, been a driving force to switch to reusable alternatives, some coffee outlets also offer discounts for those using reusable cups, or reward schemes (Environmental Audit Committee, 2018; Smithers, 2018). Similar pressure on single use packaging, such as a tax, could also cause changes in behaviour and a move towards reusable beverage containers and refill systems.

Plastics are also considered to be less inert in the environment than glass, as plastics when broken down into smaller microplastics have many well documented detrimental impacts on the environment this study has not been able to entirely capture the complex impacts plastic particles can have (Vethaak and Leslie, 2016; Eriksen et al., 2014). Plastic waste in the environment is noted as a habitat for various bacteria and pathogens that can be detrimental to the environment, whilst there is less research on glass in this context it does not seem to have the same impact potential (Vethaak and Leslie, 2016).

Whilst there would still be packaging involved in the methods outlined above it would be far less, and more renewable solutions could be found for those, for example if a supermarket was providing refills, the containers used to transport the concentrate could also be refillable and reusable and taken back by the beverage manufacturer. Companies like Sodastream already have infrastructure in place for gas canisters for their products to be returned and refilled. Refill stations would be arguably more difficult for beverages such as fresh milk and fresh fruit juice that are more prone to spoiling and have a shorter shelf life but could be possible with appropriate management and changes in customer behaviour.

5.2 Limitations

LCA sometimes struggles to model the impacts of unpredictable factors. For example, the likelihood of incidents due to extraction processes or dangerous manufacturing processes are complicated to model because they do not occur regularly and cannot be scoped into a LCA. The various different ways materials might be extracted or obtained cannot all be placed into a single model and the potential for endless iterations of the models is difficult to manage. Similarly, the management of waste in ways that are unpredictable (such as upcycling or fly-tipping) can add a level of complexity that LCA cannot account for.

Transport was scoped out of the study due to the complexity and variability of transportation methods; each shipment of beverage packaging may come from different sources depending on the company producing the beverage. Some may have their packaging produced in the same factory others may ship them in from overseas. There is no standard that can be applied. However, this does remove a vital source of emissions from the assessment as vehicular pollutants can be highly impactful on the environment and human health. Alongside this some studies that have attempted to scope in transportation have found that heavier, bulkier packaging types require more fuel and energy to transport, which would likely impact the results of the assessment for this study (Accorsi et al., 2015).

6. CONCLUSIONS

This study has successfully reviewed the costs, benefits, advantages and disadvantages of plastics as packaging materials and used LCA to determine if there is less environmentally impactful beverage packaging than plastic bottles. It compares beverage packaging through usage, giving clear results within the scope of the LCA. In each category there are more environmentally-friendly alternatives to plastic bottles. For pressurised beverages, aluminium cans, particularly recycled aluminium cans, are less impactful. For fruit juice, Tetra Pak packaging is less impactful and for milk, cartons are less impactful. However, glass bottles even if they are made completely from recycled materials are more impactful than plastic bottles.

This shows that whilst there are single use beverage packaging replacements for plastics they themselves are not the most negatively impactful single use beverage packaging within the scope of this study.

Whilst this study supports the results of previous LCAs for beverage packaging, we have not used subjective weighting and we have used: i) free, open source software and ii) the comprehensive CML impact categories that allow for a wider analysis of the overall impacts of all packaging types than many previous studies. Due to datasets being frequently updated for the processes that make up product systems LCAs need to keep current with changes to production especially as new innovations and changes to energy provision can drastically change the environmental impacts of products.

It is recommended that the packaging types identified as the least impactful in each category are used in situations where single use packaging is required. However, there should be a move towards reusable beverage packaging to reduce environmental impacts and encourage more sustainable lifestyles. Changes in infrastructure and potential incentives to use reusable packaging should be implemented and policies such as the proposed coffee cup tax should be adapted for single use beverage packaging. All beverage packaging assessed showed some form of environmental impacts and both the milk carton and Tetra Pak, despite being less impactful than the plastic bottles still contain plastic elements.

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DETERMINATION OF THE COMPOSITION OF MULTILAYER PLASTIC PACKAGING WITH NIR SPECTROSCOPY

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ABSTRACT
In food and medical packaging, multiple layers of different polymers are combined in order to achieve optimal functional properties for various applications. Flexible multilayer plastic packaging achieves a reduction in weight compared to other packaging products with the same function, saving material and in transportation costs. Recycling of post-industrial multilayer packaging was achieved by some companies, but the available technologies are limited to specific polymer types. For post-consumer waste, recycling of multilayer packaging has not been achieved yet. One of the main challenges in plastic sorting is that the detection and separation of multilayer packaging from other materials is not possible yet. In this study, the possibility to detect and sort flexible multilayer plastic packaging was investigated with near-infrared spectroscopy, which is the state-of-the-art technology for plastic sorting. The results show that from a detection and classification point of view, sorting of monolayer, two- and three-layers samples under laboratory conditions is possible. According to the captured data, the sequence of layers has little influence on the spectra. In case of glossy samples, the spectra are influenced by printed surfaces. With an increase in thickness, the spectra get more characteristic, which makes the classification easier. Our results indicate that the sorting of post-consumer multilayer plastic packaging by main composition is theoretically achievable.

1. INTRODUCTION

In the European Union, the largest proportion of produced plastics is used as packaging material, with about 40 wt% of the total amount. Among the packaging materials produced, 17 wt% are multilayer packaging films [Mumladze et al., 2018]. Flexible multilayer plastic packaging is becoming more and more popular especially for food and medical packaging due to the low energy consumption in production, low material requirements and the reduction in packaging weight [Butler, Morris; 2016]. Besides, multilayer packaging enables an extension of functional properties by adding additional layers, for example, a barrier layer against oxygen and water vapor [Butler, Morris; 2016].

Multilayer packaging consists of two or more layers of different materials, which depending on the production process, are either bonded with diffusion forces between layers or with special adhesives. There are different kinds of materials available, depending on the application area and functional requirements [Emblem, Hardwidge, 2012]. Each kind of material has its own physical and functional properties; Therefore, there is no general composition of multilayer packaging [Dixon, 2011]. The most commonly used flexible packaging film materials are low-density polyethylene (LDPE), polypropylene (PP) and polyethylene terephthalate (PET) [Wagner, 2016].

Plastic films separated from sorting plants often end in incineration or in low-quality products through downcycling [Plasticseurope, 2019]. To achieve a high recycling rate, even low quality waste streams, such as flexible multilayer packaging and mixed film waste, need to be recycled. Several companies have achieved post-industrial multilayer packaging recycling with different processing treatments. For example, the company APK GmbH from Germany has successfully recycled polyamide (PA) / polyethylene (PE) multilayer materials by using special solvents [APK GmbH, 2020]. In case of post-consumer multilayer packaging waste, the recycling is not yet possible. One of the challenges is determining the composition and separating them from other packaging materials in post-consumer waste.

Near-Infrared (NIR) spectroscopy is the state-of-the-art technology in sorting plastic waste. It classifies materials according to their own characteristic spectrum [Siesler et al., 2002]. In this study, the composition determination of transparent post-consumer multilayer plastic packaging

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TABLE 1: Available monolayer materials and thickness.

Material	Typical thickness (µm) [Dixon, 2011]	Colour	Analysed thickness (µm)
OPP	15 - 50	transparent	15, 20, 35*
LDPE	25 - 100	transparent	30, 50, 70, 100*
PET	12 - 50	transparent	12*, 23, 45*, 50*

**printed sample available; OPP: oriented PP*

waste is investigated with NIR spectroscopy by comparing the spectra of monolayer with two and three-layers samples. Based on this, the composition of two-layer samples was analysed and determined. In addition, influences of layer thickness, material proportions, layer sequence and printed surfaces were investigated.

2. MATERIAL AND METHODS

In Germany, 2-dimensional (2D) plastic, e.g. films, and 3-dimensional plastics are collected in lightweight packaging waste. In most sorting plants, films are firstly separated from the input waste stream with air classifiers or ballistic separators for further processing or transportation [Pretz et al., 2020]. Therefore, multilayer packaging needs to be separated from other 2D plastics.

2.1 Material selection

As adhesives are not necessary in multilayer production and the thickness of adhesives layer is typically much lower than the plastic layers [Emblem, Hardwidge, 2012], samples without adhesives are used. The investigated material types are LDPE, PP and PET, since they are most often used in flexible packaging films [Wagner, 2016]. Typical thickness in packaging film plastics as well as available monolayer material, colours and thicknesses are shown in Table 1. Multilayer samples were two or more layers of monolayer samples. The samples are virgin material and were produced by the company Estiko-Plastar AS (Tartu, Estonia).

2.2 Spectral analysis

The detection of multilayer plastics and the sorting ability were determined through spectral analysis with NIR spectroscopy. The NIR sensor in this study is a Helios-G2-320 from EVK DI Kerschhaggl GmbH (Raaba, Austria) with a spectral range of 900 to 1700 nm and a spatial resolution of 0.8 mm/pixel. Halogen lamps are used as emitters. The reflection of radiation from the surface is captured by the NIR sensor, see Figure 1. The difference in NIR spectra is represented by changing of the intensities of the reflected radiation. Consequently, the first derivative of the reflected radiation was used for analysis and the samples were then pixel-based classified with support vector machine algorithms based on the characteristic peaks (position and level).

Usually in sorting plants, the background of a NIR-based sorter is a black conveyor belt. Black materials reflect less radiation and can thus be easily distinguished from non-black objects. However, since most multilayer packaging materials are transparent and have low thickness, the intensity of the reflected radiation is relatively low and the detection is limited. To demonstrate a general feasibility of the composition determination for flexible multilayer plastic packaging, the used background was a homogenous paper in dark green, which is NIR active and has its own characteristic spectrum. The detection and classification of materials are therefore based on the differences between the spectra of samples and of the used NIR-active paper.

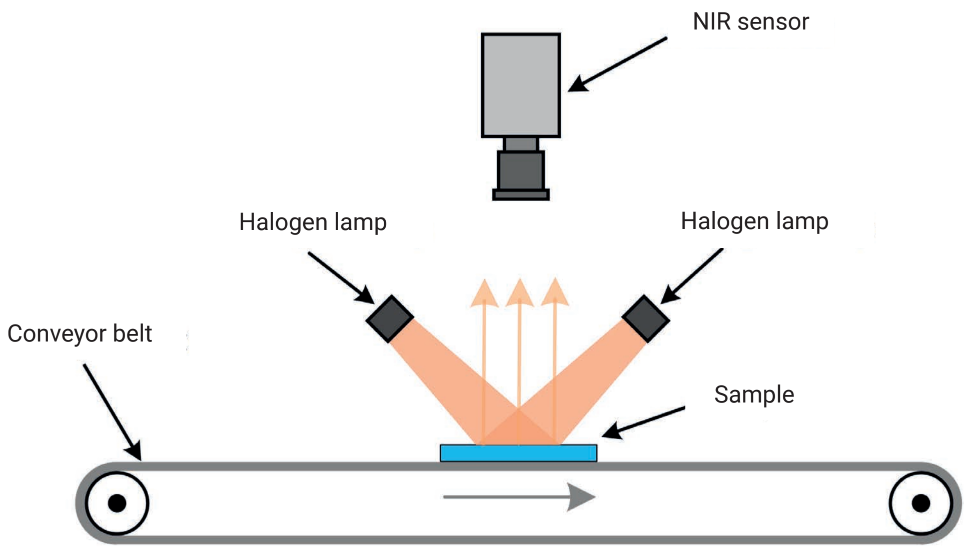

FIGURE 1: Schematic sketch of NIR sensor and halogen lamps arrangement.

3. RESULT AND DISCUSSION

In order to determine the detectability of different multilayer samples, spectra of background material and monolayers were captured. The spectra of multilayer samples were then compared with the ones of the background and corresponding monolayers to determine whether the difference is great enough to be detected and classified. Figure 2 shows mean spectra of background, transparent OPP with a thickness of 35 µm, transparent LDPE with 30 µm and two layers combinations as an example.

As Figure 2 shows, all layers could be detected, since their spectra differ from the spectrum of the background. The two-layer samples could be distinguished from the monolayer samples by the position of peaks, e.g. in the wavelength area 1200-1300 nm and about 1460-1550 nm. However, the spectra shown in Figure 2 are only the mean values of some selected pixels and classification process of all pixels is necessary for determining the sorting ability. The results in Figure 3 show that the samples were successfully classified into their right classes with an accuracy of more than 95%. The spectra of other mono- and two-layer materials were also compared with each other and the classification results are similar. Diverse compositions show spectra with slight differences and it was possible to classify more than 87% of all pixels to the right classes.

Additionally, three-layers samples exhibited a difference to mono- and two-layers ones in spectra (figure 2) and the classification according to the composition was also possible. The spectrum of three-layers PET/OPP/LDPE shows a difference at the wavelength from 1160 to 1300 nm and at about 1400 nm to the PET monolayer. At about 1680 nm, the negative peak differs this three-layers sample from OPP and LDPE. With these characteristics, the samples were possible to be classified to their classes with an accuracy of 89%.

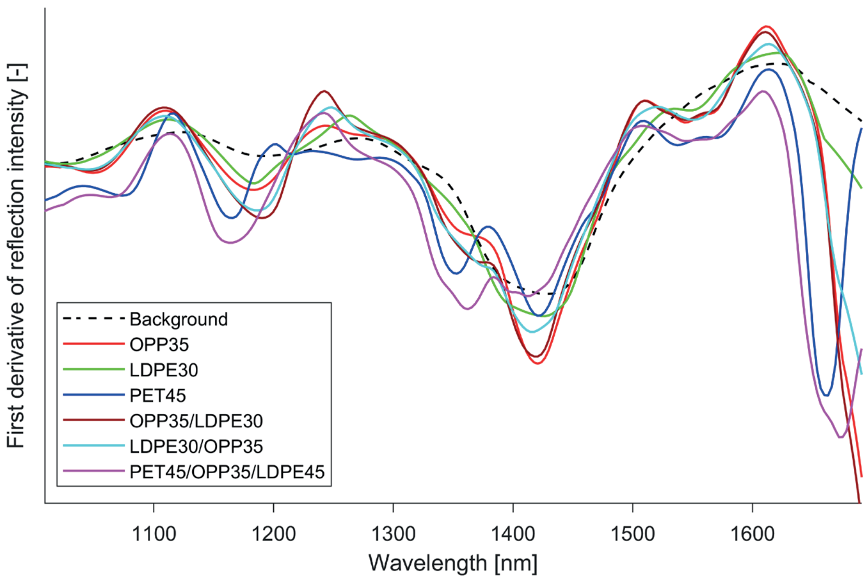

FIGURE 2: Mean spectra of background, OPP35, LDPE30, PET45, OPP35/LDPE30, LDPE30/OPP35 and PET45/OPP35/LDPE30. (Sample name is material with thickness, e.g. OPP35/LDPE30 is two-layer sample with one OPP layer with a thickness of 35 µm and one LDPE Layer with a thickness of 30 µm, the OPP layer is exposed to NIR sensor).

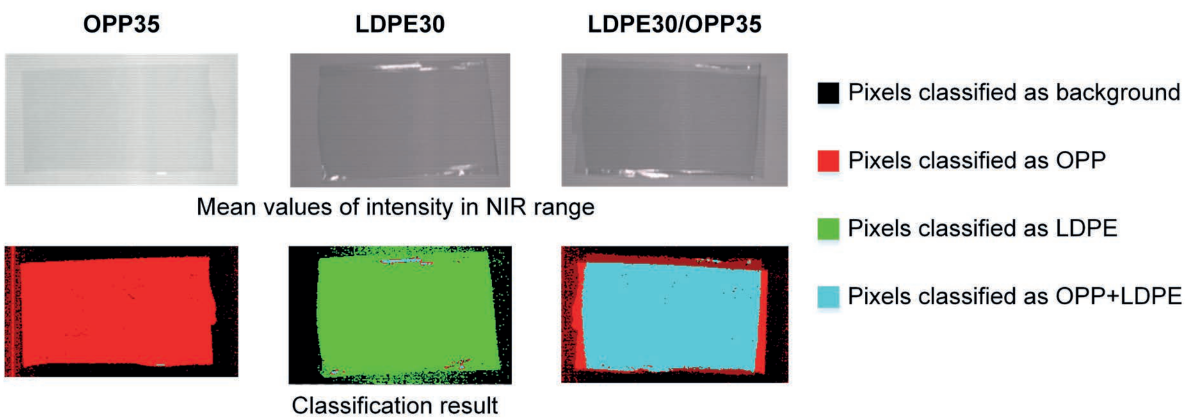

FIGURE 3: Classification results of OPP35, LDPE30 and OPP35/LDPE30.

3.1 Effects of individual layer thickness

To determine the influence of thickness and proportion of each material on classification, samples with one layer of OPP with a thickness of 35 µm and one layer of LDPE with 30, 50, 70 and 100 µm thickness each were analysed. Figure 4 shows that with increasing thickness of the LDPE layer, peaks and their position of the multilayer spectrum are getting more similar to the spectrum of LDPE monolayer. Analysis of other multilayer samples (e.g. PET with different thickness of LDPE) showed similar trends. Besides, the thicker the material, the greater the intensity of the peaks are. However, in this case, the proportion of LDPE increases with the overall thickness of the analysed sample, such that the influence of proportion on intensity could not be demonstrated. The classification results indicated that all the two-monolayer samples were able to be classified with an accuracy of 96%.

3.2 Effects of printing colour and layer sequence

For transparent samples, the sequence of layers did not influence the spectra, see Figure 2. The mean spectra of OPP35/LDPE30 in dark red and LDPE30/OPP35 in cyan in Figure 2 show a slight difference, but it was not sufficient to classify them into two classes, as almost half of the pixels were classified to wrong classes. Besides, there is no difference in the spectra and classification results regarding which side of the layer is exposed to the NIR sensor. For layers which are not glossy, the spectra of the samples with one layer printed are very similar to the transparent ones. This means that the detection and classification are independent on whether the samples are printed or not. Contrarily, in case of glossy samples, they could only be detected on the non-printed side, as the radiation is fully reflected from the glossy side and thus, the spectra could not be classified.

4. CONCLUSIONS

Through the analysis of monolayer, two- and three-layers samples with NIR spectroscopy, it has been demonstrated that different multilayer samples can be distinguished from each other by the classification of the captured spectra with support vector machine algorithms under laboratory conditions. Samples with different composition (OPP\LDPE, OPP\PET, LDPE\PET and OPP\PET\LDPE) have been analyzed and the captured spectra showed a difference to other multilayer and to the monolayer materials. The sequence of layers does not influence the spectra of samples, but there is a difference in spectra when glossy samples are printed. With increasing layer thickness, the intensity of peaks in the captured spectra is higher, which makes it easier to classify them. With the increase of proportion of one polymer, spectra are getting more similar to the spectrum of this material. Consequently, sorting materials according to their main composition might be possible for post-consumer multilayer plastic packaging.

However, all the tests in this study are in laboratory conditions and the sample materials were virgin production material and did not come from post-consumer waste streams. In practical sorting processes, post-consumer packaging plastics often have impurities on the surface, which, on the one hand, brings benefit as the samples are less glossy and they could be detected. On the other hand, NIR-active impurities can influence the spectra of the materials and make the classification more difficult. In further research, the impact of impurities and the detection and classification of post-consumer multilayer plastic packaging should be analyzed. In addition, further research is required to evaluate whether these results can be transferred to the NIR sorting of post-consumer waste streams.

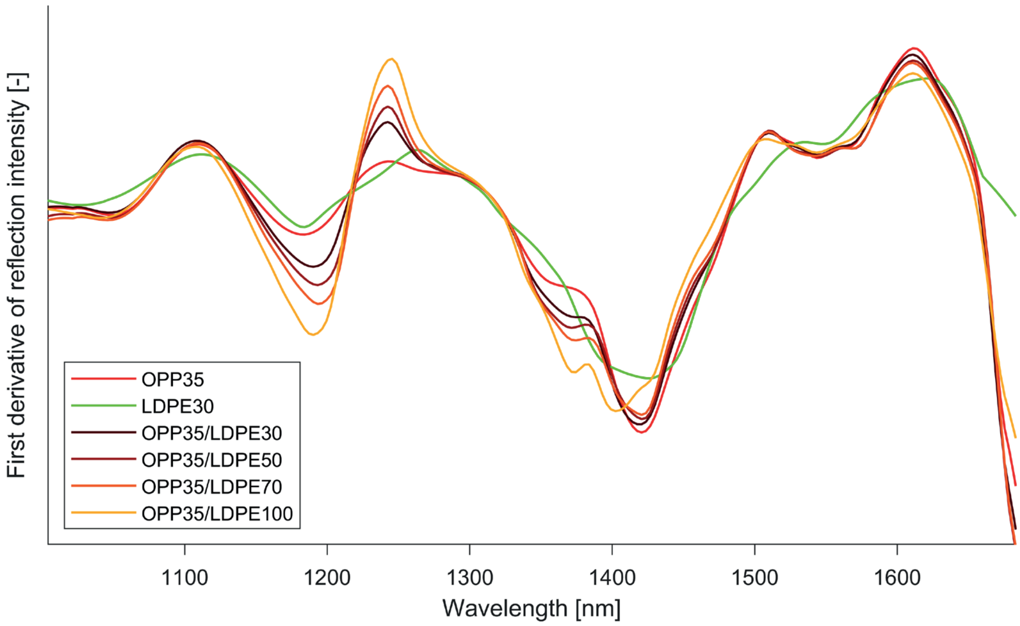

FIGURE 4: Mean spectra of samples with OPP 35 µm and LDPE 30, 50, 70 and 100 µm.

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REDUCING THE EFFECTS OF PLASTIC WASTE IN AGRICULTURAL APPLICATIONS BY DEVELOPING NEW OK SOIL BIODEGRADABLE PLASTICS

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ABSTRACT
Current intensive farming practices require the use of large quantities of mulching film and fruit protection bags since they prevent the growth of weeds, regulate soil temperature, retain water and nutrients and protect crops from insects. These practices use single-use conventional non-degradable polymers which create a serious problem of waste management since this management is time-consuming, expensive to recycle and, more importantly, non-environmentally friendly. By using biodegradable plastics, this problem could be solved by preventing the creation of waste. The objectives of this study were to produce an innovative biobased and biodegradable plastic film that entirely biodegrades in natural conditions on the field within a short time after its usage and to know its effects on tomato and peach crops. For this purpose, two types of films were produced: biomulching and biobags. Specific oligo elements have been added to the films in order to test the biomulching in tomato crops and to protect the fruits from insects and improve quality at harvest in peaches (biobags). Tests carried out on tomatoes showed that, these bioplastics improved soil quality by increasing (up to 13%) the concentration of oligo elements and by decreasing (65%) blossom end rot. By using biobags in peaches, a uniform colour (without red blush), required characteristic in this type of commodity (Protected Designation of Origin 'Calanda'), was obtained, with a decrease in both a* colour coordinate (more than 2 points) and carotenoid content (more than 3 μg g^{-1} fw). Moreover, bioplastics degrade completely after 6 months within the soil.

1. INTRODUCTION

For over a half a century farmers have been using plastic materials in agriculture because of their affordability and their easiness to be applied in the field. The main use of plastics in agriculture is for mulching, and in some Mediterranean areas, for fruit protection bags. The first ones prevent the growth of weeds, regulate soil temperature, and retain water and nutrients which means an increase in yields (Kader et al., 2017). The second ones are single use agricultural bags used in tree crops to protect the fruit from the Mediterranean fly (Ceratitis capitata), the climatic incidences and mainly from chemicals (Sharma et al., 2014).

The convenience of using this type of plastics has made the consumption of plastics grown rapidly in Europe (Mormile et al., 2007). The global market for agricultural plastic films, 4 million tonnes and approximately 10.6 million USD (2015), is projected to grow 5.6% per year through 2030 (Vitova, 2015). The total consumption in Europe exceeded 500.000 metric tons in 2013 being Spain and Italy the countries with highest consumption, due to their intensive horticulture activities. Together, they account for 40% of the demand and consume more than 120.000t per year (Plasteurope.com, 2017).

The main problem of agricultural films is that they have a lifespan of just one cultivation cycle, after which they need to be replaced, which is an intensive, expensive and time-consuming task (Malinconico et al., 2008). The conventional polymers used are non-degradable: LDPE (low density polyethylene) and HDPE (high density polyethylene). The use of this type of plastics create a serious waste management problem since it is time-consuming, expensive to recycle and, more importantly, it is non-environmentally friendly. Furthermore, films are increasingly thinner and often end up being damaged during the cultivation process so they fall apart into smaller pieces, which

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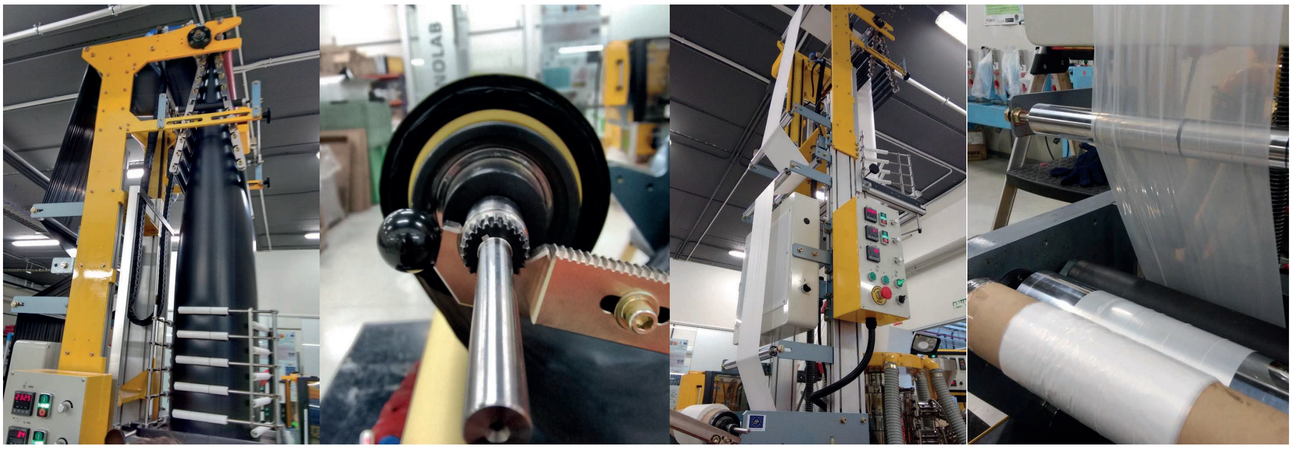

FIGURE 1: Biodegradable agricultural plastics production for mulching and fruit protection bags applications.

complicates the recycling and cleaning up. Approximately only 24% of mulching film in the European Union gets recycled, while about 50% ends up in landfills and the rest is incinerated or abandoned in the fields (European Commission, 2014).

By using biodegradable plastics, this problem could be solved by preventing the creation of waste. Biodegradation is a chemical process during which microorganisms that are available in the environment convert materials into natural substances such as water, carbon dioxide and compost within short time lapse. This process depends on the surrounding environmental conditions (e.g. location or temperature), of the material and on the application. According to the International Standard (EN13432), biodegradable plastics are those that are degraded either in 6 months at 58C (industrial), in 1 year at 28C (home composting) or in 2 years at 26C (soil environment).

Over the past few years, international research has carried out many tests to compare soil and crop quality as well as harvest yields between biodegradable films and polyethylene films and they show no differences between them. (Kasirajan & Ngouajio, 2012, Steinmetz et al., 2016).

Our research focuses on the development of an innovative biobased and biodegradable plastic film that entirely biodegrades in natural conditions on the soil in order to obtain the "OK Biodegradable SOIL" certification (provided by TÜV Austria). This certification ensures that the plastics are entirely biodegradable and not phytotoxic for soil and plants. Furthermore, this study has contemplated the innovation of the addition of certain specific oligo elements (Mn, Zn and B) to the mulching films in order to study its effects on the biodegradability and quality of the crops. Additionally, with the use of the biodegradable bags, aims to obtain a homogenous colour and a higher quality peaches which increase their selling price.

2. MATERIALS AND METHODS

2.1 Production of bioplastics

Biodegradable films were obtained by mixing selected natural biopolymers and additives in a conventional extrusion compounding process (Coperion ZSK26 co-rotating twin-screw extruder, semi-industrial) at AITIIP facilities (Zaragoza, Spain). The compounded materials were afterwards dried (Mini dryers Moretto X DRY AIR T) to ensure a low water level content that could negatively affect film properties. Finally, the blown extrusion machine (LABTECH LBM 125, semi-industrial) equipped with a film module (Type LF-400-COEX) was used to obtain all biodegradable film products (Figure 1).

Samples of mulching and fruit protection bags samples were based on Mater-Bi™ (corn thermoplastic starch, co-polyester, Novamont S.p.A.), Danimer™ (PHA/PLA, Meridian Holding Group) and BioPBS™ (bio-based polybutylene succinate, Japan Pulp & Paper GmbH). All the mulching films were 22 µm-thick and carbon black was used as a color additive using the masterbatch techniques. In addition, different percentages of oligoelements were added to the samples: Zn/Mn complex and Boron. Protection fruit bags were 40-50 µm-thick and white pigment (PW) was added as bleaching additive in the masterbatch processing.

Due to industrial secret, different concentrations of oligoelements or bleaching agents have been coded as A for the lowest concentration and B for the highest. Conventional LDPE mulching (Comercial Arnedo, Spain) and conventional waxed paper bags (Cooperative Calanda DO) were used as control samples.

The reference material for biodegradation test is microcrystalline cellulose, produced by ALDRICH, distributed by SIGMA ALDRICH SRL, Code number 310697-500G, Lot number MKBX5118V, Expiration date July 13th, 2022. The average dimension of the particles is 50 µm,

2.2 Experimental design

Three crop seasons (2016, 2017 and 2018) were analyzed for both mulching and fruit protection bags.

For mulching, in the first season 3 different plastics were tested (M11, M21 and M31) with two different concentrations of oligoelements for each plastic (Manganese/Zinc (codified as A for the lowest concentration and B for the highest)). The mulching films were placed in the field with a separation of 1 m between lines. In the second season, the sample with lower mechanical performance was discarded and different amounts of oligoelements were added (two percentages of Mn/Zn and other two percentages of Boron (codified as A for the lowest concentration and B for the highest). Finally, in season 2018, after analysing of the previous results, the best combination selected was material M13 with Boron as oligoelement. A randomized complete block experiment was used to evaluate bioplastics, using three blocks which contain a complete set of bioplastics.

For fruit protection bags, in 2016, three different bioplastics were assayed (B11, B21 and B31) with two levels of added white pigment (coded as A for the lowest concentration and B for the highest). 300 bags per batch were tested and randomly distributed in six blocks. Each block consisted of one tree.

In the second season (2017), the best performance plastic (B12) and a new one (B42) were tested with a reduction in the amount of pigment content (codified as A for the lowest concentration and B for the highest). 500 bags per batch were tested and randomly distributed in six blocks.

In 2018, the best combinations of materials for mulching and fruit protection bags were tested as definite formulations. Table 1 shows the different samples compositions.

2.3 Vegetables and fruit samples

For mulching, tomatoes (Solanum lycopersicum 'Manitu') were manually planted (25 May, 2016; 2, June 2017, 4 May, 2018) with a separation between plants of 50 cm. They were harvested randomly from a commercial orchard located in the Mid-Ebro Valley (Zaragoza, Spain) at the time of optimum commercial harvest (25 August, 2016; 31 August, 2017; 28 August, 2018).

For fruit protection bags in peaches (Prunus persica '58GC'), the bags were randomly placed in a commercial orchard located in the Ebro Valley (Calanda, Spain) in the middle of the season (14 July, 2016; 17 July, 2017, 16 July, 2018). Fruits were harvested at optimum commercial harvest (13 September, 2016; 6 September, 2017; 20 September, 2018). Both crops were grown under drip irrigation and following the agronomic practices of the area. All samples were transferred immediately to the laboratory to carry out the fruit quality analysis.

2.4 Characterization Analyses

2.4.1 Mechanical properties

The modulus of elasticity (E) was determined using ISO 604 "Plastics – Determination of Compressive Properties". Elongation at break (ε) and tensile strength at break (σ) were determined by tensile testing in accordance with ASTM D 882 – 12 "Standard Test Method for Tensile Properties of Thin Plastic Sheeting".

2.4.2 Heavy metals and fluorine concentration of the biofilms

The concentration of heavy metals was quantified using the EPA 3052 1996 "Microwave assisted acid digestion of siliceous and organically based matrices" and EPA 6010C 2007 "Inductively coupled plasma-atomic emission spectrometry". Heavy metals are defined in the Standard EN 13432:2000 "Packaging: requirements for packaging recoverable through composting and biodegradation".

The concentration of fluorine was quantified using EN 14582:2016 "Characterization of waste - Halogen and sulfur content - Oxygen combustion in closed systems and determination methods" and EN ISO 10304-1:2009 "Water quality - Determination of dissolved anions by liquid chromatography of ions - Part 1: Determination of bromide, chloride, fluoride, nitrate, nitrite, phosphate and sulfate (ISO 10304-1:2007)".

2.4.3 Biodegradation tests

Biodegradation tests were carried out according to ASTM D 5988-12 "Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials in Soil". The tests were performed adding a mixture of three different

TABLE 1: Sample composition of mulching samples (M) and for the fruit protection bags (B).

Year	Mulching		Bags	
	Material	Oligoelement Level	Material	Colour Level
2016	M11	M11A	B11	B11A
		M11B		B11B
	M21	M21A	B21	B21A
		M21B		B21B
	M31	M31A	B31	B31A
		M31B		B31B
	Control (LDPE)	-	Control (waxed paper)	-
2017	M12	M12A	B12	B12A
		M12B		B12B
	M42	M42A	B42	B42A
		M42B		B42B
	Control (LDPE)	-	Control (waxed paper)	-
2018	M13	M13B	B43	B43B
	Control (LDPE)	-	Control (waxed paper)	-

soils (agricultural field, meadow area and forest area), to a compost (as inoculum) produced by Gardea, in the best experimental conditions (1:23 inoculum/soil and 300 mg of cellulose as reference material) to obtain the most effective substrate for the test. Samples for biodegradability were tested after milling with liquid N_2. The test was carried out in glass desiccators of 2 - 3 L inner volume, airtight (diameter 200 mm) in an overshadowed incubation chamber, with temperature monitoring at 26±2°C.

2.4.4 Soil collection and chemical analyses

Soil samples (only for mulching), were collected from the upper layer (20 cm) of the areas where the plastics were placed in two dates: before their colocation, and after 4 months of harvest and incorporation of plastic into the soil using a tractor. Samples were collected randomly from each repetition of each block of plastics. Each sample included 8 sub-samples taken throughout the entire line. Total carbon and total N were determined on a LECO TruSpec C/N/S automatic elemental analyzer (St. Joseph MI, USA). Samples of soil were analysed for total macro- and micronutrient in the samples, after microwave digestion in 65% HNO_3 using inductively coupled plasma-optical emission spectrometry (ICP-OES, model ICAP 6500 DUO THERMO).

2.4.5 Quality parameters

To evaluate the quality of tomatoes and peaches, destructive and non-destructive methods were used. 150 fruits per experimental unit were analyzed. Colour coordinates were determined using the CIELab colour space instrument with the aid of a spectrophotometer (Konica Minolta mod. CMS 700; Tokyo, Japan). Firmness was measured using non-destructive Acoustic Firmness Sensor (AWETA; Netherlands) for peaches and Durofel (Agrosta; Forges Les Eaux, France) for tomatoes. The firmness was also measured by destructive Magness-Taylor test using a digital penetrometer (Agrosta) with a tip diameter of 8 mm for peaches and of 4 mm for tomatoes and expressing the results as kg. Soluble solid content (SSC) as Brix degrees was determined in 10 samples by crushing the flesh and transferring the intact juice of the 10 samples to a digital refractometer (Atago mod. PR-101; Tokyo, Japan). Titratable acidity (TA) was measured by an automatic titrator (Mettler Toledo mod. G20 Compact Titrator; New York, NY, USA). Ten grams of juice from 10 fruits were diluted into 60 mL of distilled H_2O and titrated with 0.1 mol L^{-1} NaOH solution up to pH 8.1, expressing the results as g of malic acid per kg.

2.4.6 Statistical analysis

All samples were analysed at least in triplicate each year. Statistical analyses were performed using a one-way ANOVA test and the significance of the difference between means was determined by Duncan's multiple range test ($p<0.05$). Statistical analysis was performed using the Statistical Package for the Social Science (SPSS) software version 23.0.

3. RESULTS AND DISCUSSION

3.1 Mechanical properties

The mechanical properties of bioplastics for mulching are showed in Table 2. In some cases, oligoelements made plastic processing more difficult, and consequently, it was necessary to increase the thickness. Sample M11 is much more elastic (ε=552-615%) than the other ones tested, meanwhile mulching sample M31 was difficult to process due their low value of ε, ranging from 62% with oligoelements to 154% without them. Moreover, σ is higher in biobased samples than in control sample and decreases with oligoelements in all cases. In the second year, M12 was the best bioplastic. Consequently, in general, M11 and M12 without oligoelements showed the best mechanical properties due to the high values of E, σ and ε.

TABLE 2: Quality parameters of mulching plastic films.

Year	BATCH		Thickness (μm)	E (Mpa)	σ (Mpa)	ε (%)
	Material	Oligoelement Level				
2016	M11	M11A	20 (0)	183 (69)	24.3 (2)	552 (194)
		M11B	38 (4.1)	55 (10)	8.3 (2)	615 (117)
	M21	M21A	20 (0)	166 (35)	6 (3)	235 (118)
		M21B	30 (0)	108 (29)	5.4 (1)	214 (73)
	M31	M31A	30 (0)	245 (35)	17.3 (3)	154 (10)
		M31B	30 (0)	127 (25)	12 (4)	62 (54)
	Control (LDPE)	-	42 (8)	300 (14)	4.5 (1)	600 (20)
2017	M12	M12A	31 (1.5)	190 (55)	25.5 (1.8)	430 (90)
		M12B	33 (1.2)	160 (63)	22.1 (2.2)	583 (129)
	M42	M42A	51 (4.9)	137 (60)	6.4 (2.9)	247 (88)
		M42B	40 (3.3)	122 (55)	4.9 (3.3)	226 (61)
2018	Control (LDPE)	-	12 (2.6)	187 (20)	26 (3.8)	280 (39)
	M13	M13B	22 (2.0)	220 (42)	34 (7.3)	310 (40)

[1]*The values between parentheses are the standard deviation.*

TABLE 3: Heavy metals (mg kg^{-1} dm) in mulching and bags samples (2016-2018).

Metal	Control	M11A	M11B	M21A	M21B	M31A	M31B	M12A	B42A	B43B	M13B	DL (mg kg^{-1} dm)	EN 13432 (mg kg^{-1} dm)
Arsenic	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	< DL	< DL	2.5	5
Cadmium	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	< DL	< DL	0.19	0.5
Chromium	<DL	<DL	0.70	1.33	1.5	<DL	0.77	1.88	< DL	< DL	< DL	0.5	50
Mercury	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	0,358	0,426	0.3	0.5
Molybdenum	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	< DL	< DL	0.5	1
Nickel	1.15	<DL	<DL	<DL	<DL	<DL	<DL	1.41	< DL	< DL	< DL	1	25
Lead	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	0,514	0,86	2	50
Copper	6.63	<DL	1.61	<DL	1.70	<DL	2.39	< DL	< DL	< DL	0,437	1	50
Selenium	<DL	<DL	<DL	<DL	<DL	<DL	<DL	< DL	< DL	< DL	1,72	0.5	0.75
Zinc	<DL	5.88	1360	7.18	1700	10.5	2010	< DL	< DL	< DL	< DL	5	150
Fluorine	< DL	< DL	< DL	< DL	< DL	< DL	< DL	< DL	< DL	5,33	5,16	0.05	100

For fruit protection bags the values obtained were also appropriate for their agriculture use (data not showed). It is remarkable that σ and ε were higher for the bags than for the mulching films. It means that loading the biodegradable matrix with carbon black and oligoelements decreased elongation and tensile strength at break values of the samples.

In addition, thickness was more variable, possibly due to processing problems in adjusting parameters for film blowing.

3.2 Heavy metals and fluorine concentration of the biofilms

The concentration of heavy metals and fluorine has been quantified in order to verify the compliance with the limits defined in the Standard EN 13432 for compostable packaging. The regulation limits in mg kg^{-1} (dry mass) the quantity of certain heavy metals (Arsenic, Cadmium, Chromium, Mercury, Molybdenum, Nickel, Lead, Copper, Selenium and Zinc) and fluorine. The results showed that all metals and fluorine were below the detection limits of the technique with the exception of the Zinc element (Table 3). In 2016, the mulching samples that were additivated with higher concentration of Zn, presented a value above the regulation limit of 150 mg kg^{-1} (dm). The amount of oligoelement added to the mulching was calculated for fertilization purposes, but in order for the plastics to be labelled as "OK biodegradable SOIL" the percentage had to be lower. Therefore, in 2017 the concentration of this element in plastic was reduced (data not showed), but once again, the values were over 150 mg kg^{-1} (dm). For fruit protection bags, all samples are below the allowed limit (data not showed).

3.3 Biodegradation results

Regarding biodegradation behaviour, in the soil selected for the tests, high percentages of biodegradability in soil were achieved. In 2016, 98-100% of biodegradation was observed after 176 days. These values are much ap-

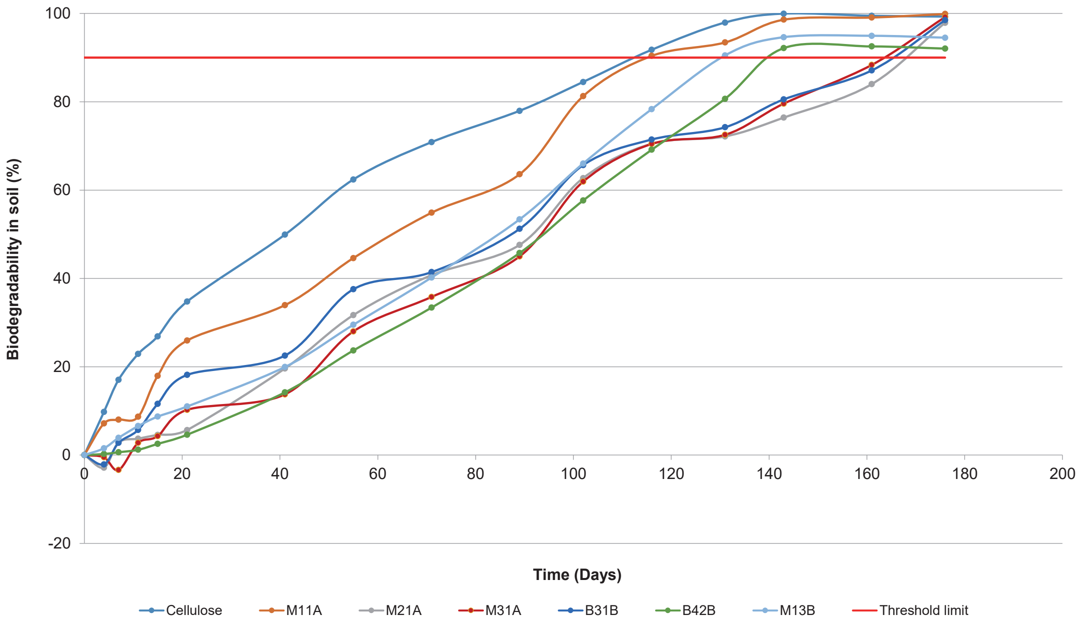

FIGURE 2: Percentages of aerobic biodegradation for each plastic material tested and the positive reference material.

propriate considering that the "OK biodegradable SOIL" certification forces to obtain a degradation of more than 90% within two years. In Figure 2 shows the percentages of aerobic biodegradation for each plastic material tested and the positive reference material (Cellulose).

3.4 Soil collection and chemical results

For the soil analysis, the results for seasons 2016, 2017 and 2018 are shown in Figures 3 to 5. In 2016 season (Figure 3), an increase in the concentration of Mn and Zn was observed due to the use of bioplastics which contained the highest concentrations of these elements. This result shows that the oligolements are present in the soil after plastic degradation. For the macronutrients, the concentration of P and K in soil was higher when using our bioplastics than when using the control one, thus being more interesting to use bioplastics with added oligoelements.

In the second year (Figure 4), the oligoelement concentration showed an irregular behaviour. An increase in the concentration of Mn and Zn was observed in the bioplastics with these additives, but also in the control ones. Similar results were observed for Boron. This could be due to cross contamination when the plastic degrades in the soil. No effect was observed in the concentration of macronutrients. Finally, the relation C/N decreased during the season, showing a positive effect of the incorporation of the bioplastics within the soil.

In the last season, in 2018 (Figure 5), several changes were observed regarding oligoelement concentrations.

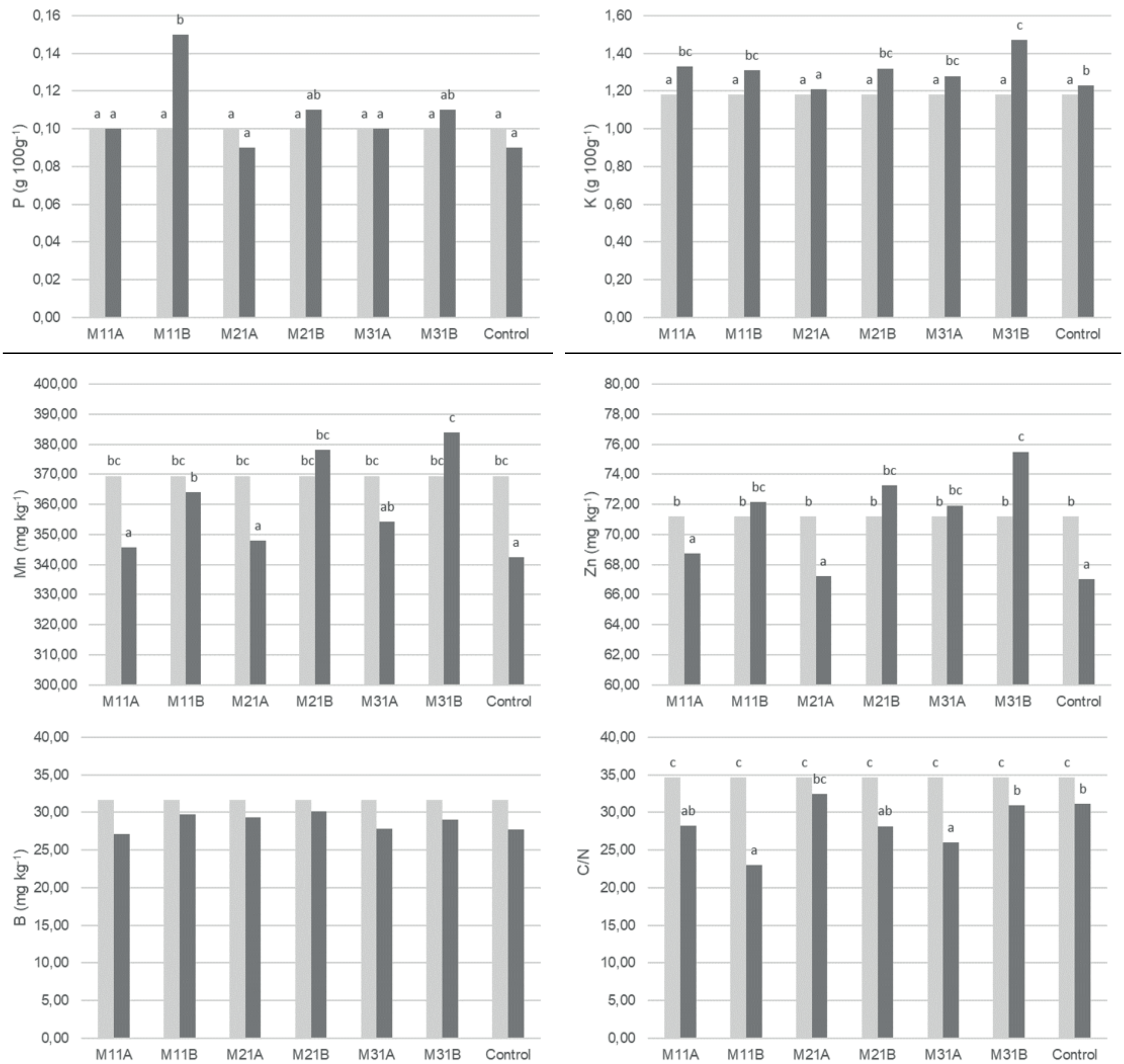

FIGURE 3: Concentrations of nutrients and oligoelements and relation C/N before (light grey) and after (dark grey) incorporation into the soil of 6 bioplastics (3 different materials with 2 different concentrations of oligoelements each one) in season 2016. Different letters indicate significant differences ($p \leq 0.05$) between treatments.

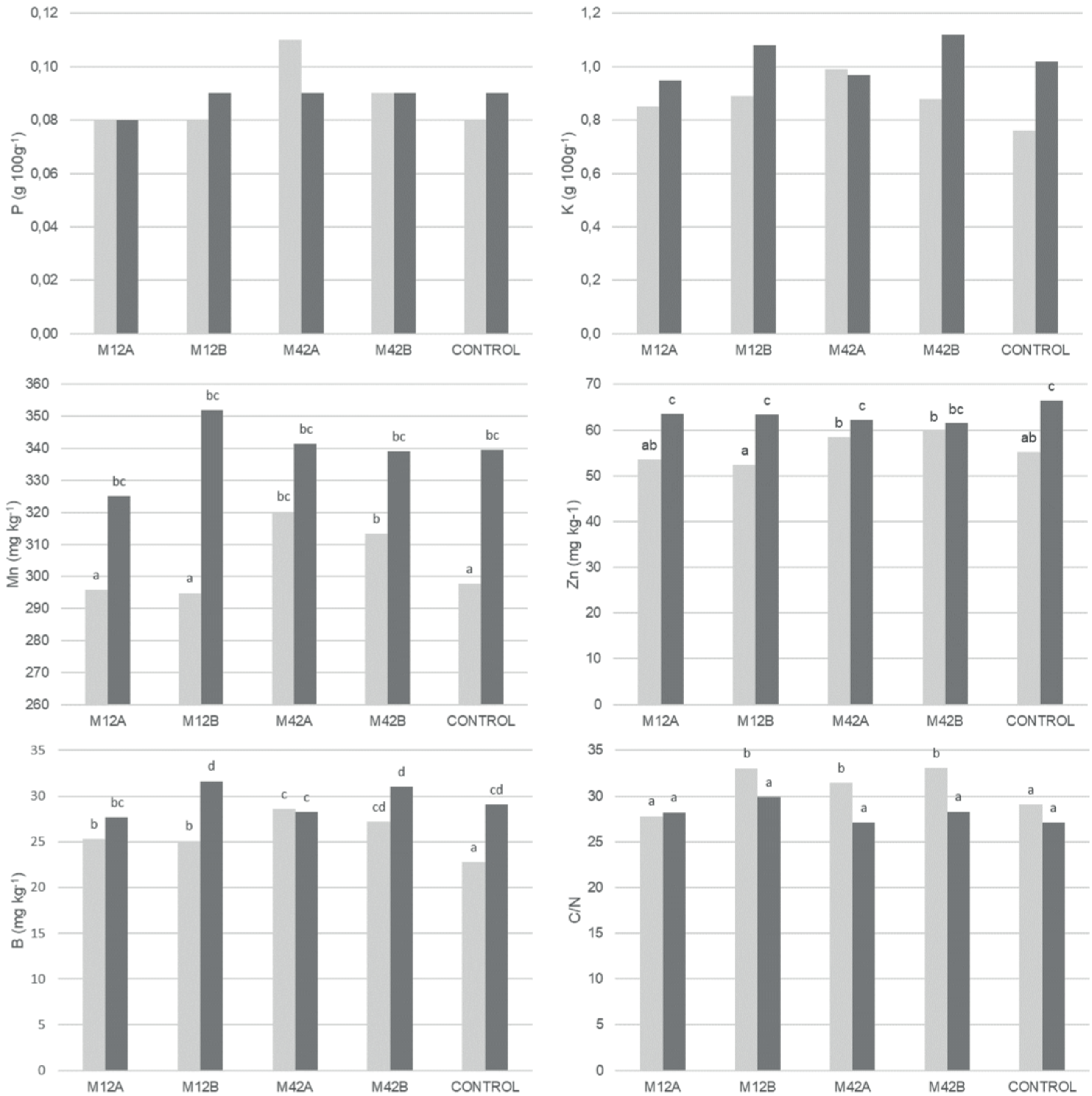

FIGURE 4: Concentrations of nutrients and oligoelements and relation C/N before (light grey) and after (dark grey) incorporation into the soil of 4 bioplastics (2 different materials with 2 different concentrations of oligoelements each one) in season 2017. Different letters indicate significant differences ($p \leq 0.05$) between treatments.

First, an increase in the Boron's concentration was observed in both control and biodegradable plastic rows. This indicates that the Boron incorporated within the bioplastics is properly liberated. However, the increase that was also observed in the control rows could also be due to a cross contamination or the bioplastic being present from previous seasons. Such cross contamination could also be the cause of the increase in the amounts of Mn and Zn. Moreover, the C/N relation decreased with the time. This is a positive result because the recommended values for C/N relation is around 15 and all the actions implemented in the field should aim to obtain these quantities.

3.5 Quality parameters

In 2016 season, although significant differences were observed, in the quality parameters of the crops, there was no clear pattern in the use of different plastics for tomatoes (Table 4). Therefore, the differences were due more to the intrinsic variability of the sample instead of the effect of the plastics on the crop.

In 2017 season, no differences were observed. The plastics did not have an effect on these quality parameters. The incidence of blossom end rot, a water-soaked spot located at the blossom end of tomato fruits, was higher in the control (18%) than in the bioplastics M12 (7%) and M42 (8%). This result could be related to a different temperature

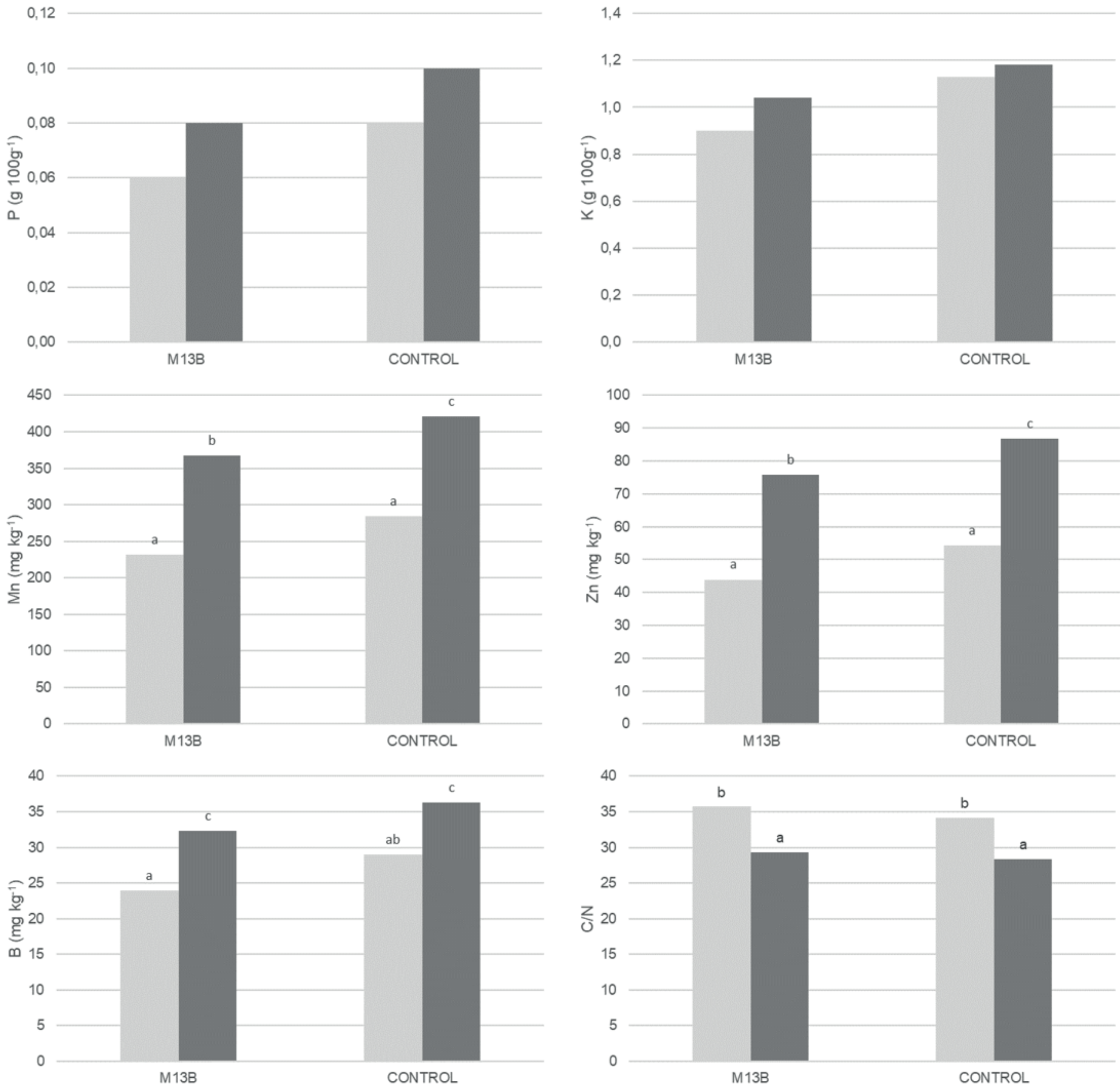

FIGURE 5: Concentrations of nutrients and oligoelements and relation C/N before (light grey) and after (dark grey) incorporation into the soil of 2 bioplastics (1 material with 2 different concentrations of oligoelements) in season 2018. Different letters indicate significant differences (p≤0.05) between treatments.

in the soil for each plastic or to different amount of reflected sunlight. More assays are necessary to confirm this hypothesis. Other tests carried out on tomatoes and peppers have also showed that fruit growth and quality gave very similar results when using biodegradable plastic or control mulch (Martín-Closas et al., 2008; Cowan et al., 2013), especially concerning total dry weight, soluble solids, colour and shape.

In 2018, statistically significant differences were observed in the weight and color parameters (coordinate a*). Fruits appeared larger and more reddish when cultivated on the new biodegradable plastic. As in the previous season, a greater incidence of physiopathies was observed in the plastic control of physiopathies, similar to the blossom end rot, specifically a cracking of the calicine zone of the fruit. Again, the biodegradable plastic showed a better performance, reaching a higher quality of the fruits. Figure 6 an example of the observed physiopathies can be seen. On the left, the tomatoes obtained with biodegradable mulching plastics can be seen and on the right, tomatos obtained with control mulching.

During the development of the fruit protection biobags, on one hand, we searched for fruits with homogeneous colour and, on the other hand, for fruits with quality parameters not affected by the use of biobags. In order to achive this, PW was incorporated to protect crops against UV rays and laser micro-perforations were done to the bottom of the bag to allow for the necessary elimination of water va-

TABLE 4: Quality parameters in 'Manitu' tomato at harvest.).

Year	BATCH		Firmness (kg)	Durofel	Weight (g)	SSC (°Brix)	a* (D65)	Blossom end rot (%)
	Material	Oligoelement Level						
2016	M11	M11A	0.32 a	65.04 a	102.11 ab	6.73 c	32.76 ab	<1
		M11B	0.39 d	68.18 b	107.84 abc	6.27 ab	34.17 c	<1
	M21	M21A	0.38 cd	70.26 b	97.97 a	6.60 bc	31.69 a	<1
		M21B	0.38 bcd	68.90 b	105.5 abc	6.23 ab	33.12 bc	<1
	M31	M31A	0.39 cd	63.62 a	113.42 c	5.93 a	31.99 ab	<1
		M31B	0.34 abc	68.76 b	102.54 ab	6.70 c	32.75 ab	<1
	Control	-	0.33 ab	69.02 b	110.42 bc	6.73 c	33.22 bc	<1
2017	M12	M12A	0.44 bc	68.06	143.75	6.47	32.24	7 a
		M12B	0.48 c	70.72	140.33	6.3	32.51	7 a
	M42	M42A	0.43 bc	69.17	146.99	6.5	32.48	8 a
		M42B	0.37 a	69.21	128.58	6.53	31.23	8 a
	Control	-	0.41 ab	70.88	141.48	6.33	32.04	18 b
2018	M13	M13B	0.57	78.12	101.05 b	6.67	31.91 b	12 b
	Control	-	0.59	79.56	74.17 a	6.21	29.48 a	4 a

[1]*different letters in the same column indicate significant differences (p≤0.05) between treatments for the same year.*

pour created during fruit ripening on trees. The use of this type of biobags did not affect the quality parameters of the peaches except for the colour (Table 5).

The use of biobags with PW caused a lower red coloration in the fruit and resulted in a lower coordinate a* value (from 14.88 to 16.05 without PW and from 12.39 to 14.49 with PW) and a more homogeneous orange colour. These values were also lower than the control ones (16.40 and 15.37 for 2016 and 2017 season, respectively). The visual differences in colour can be observed in Figure 7. The differences observed in the rest of parameters may be due to intrinsic differences in crops instead of the difference in composition of the bioplastics and conventional plastics.

4. CONCLUSIONS

The following conclusions can be drawn from the study. In general, the biodegradable mulch showed appropiate mechanical properties for its placement as the conventional mulch, and the ability to resist all crop season. The bioplastic degradation into the soil increased the concentration of Manganese, Zinc and Boron. High biodegradation in soil was observed for all the bioplastics, althought the addition of Zn was not adequate to obtain the "OK biodegradable SOIL" certification. The use of biomulching in tomatoes decreased the incidence of blossom end rot and did not affect the rest of quality parameters. For peaches, the colour was more uniform in those grown with biodegradable bags when compared to those grown with tradicional bags, this being an important feature for fruit producers. Final biodegradable formulations for mulch and fruit protection bags have obtained the "OK Biodegradable SOIL" certification.

FIGURE 6: Tomatos obtained with Biodegradable mulching M31B (a) and tomatos produced with conventional plastic, control (b).

TABLE 5: Quality parameters in '58GC' peach at harvest.

Year	BATCH Material	PW	Firmness (kg)	Aweta	Weight (g)	T.A.(g.malic L-1)	SSC (°Brix)	a* (D65)
2016	B11	B11A	3.19	10.03	220.83	5.64 ab	14.03 c	14.88 bc
		B11B	3.23	9.55	235.17	5.85 b	14.58 c	14.49 ab
	B21	B21A	3.24	12.79	232.27	5.14 a	12.93 ab	15.76 bcd
		B21B	3.03	13.23	230.00	5.14 a	12.35 a	13.34 a
	B31	B31A	3.20	8.81	207.53	6.02 b	14.70 c	16.05 cd
		B31B	3.23	12.87	221.30	5.51 ab	13.08 b	13.16 a
	Control	-	3.21	15.09	233.77	5.18 a	12.78 ab	16.40 d
2017	B12	B12A	2.56 b	9.32 a	215.46	6.75 a	13.43	15.45 c
		B12B	2.21 ab	9.64 ab	221.92	6.87 a	13	14.14 b
	B42	B42A	2.06 a	11.63 abc	211.58	7.26 a	12.63	15.62 c
		B42B	3.62 c	13.9 c	233.19	7.77 a	12.6	12.39 a
	Control	-	2.11 ab	11.88 bc	217.69	9.38 b	13.13	15.37 c
2018	B43	B43B	2.49	13.77	299.59	5.47	13.48	16.77
	Control	-	2.69	12.51	281.40	6.01	12.85	17.01

[1] *different letters in the same column indicate significant differences (p≤0.05) between treatments for the same year.*

FIGURE 7: In first row, (a) peaches obtained with control bags and in the row below (b) peaches obtained with the biodegradable bags (B43B).

AKNOWLEDGEMENTS

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TEX2MAT – NEXT LEVEL TEXTILE RECYCLING WITH BIOCATALYSTS

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ABSTRACT

Achieving a circular economy for end-of-life textiles is one of the large challenges in the textile industry. Currently, after disposal, textiles often end up in landfills or incineration plants. In recent years, the textile industry has exhibited high growth rates, with the annual global fibre production reaching 100 million t. It also has to be considered that textile products are increasingly becoming more complex to fulfil special functions, resulting in the use of multi-material textiles. However, these textiles are hard to recycle. The TEX2MAT project is a FFG (Austrian Research Promotion Agency)-promoted project conducted by a consortium of 13 research institutions and private businesses that offers a solution for material recycling. The goal of TEX2MAT is to develop an innovative process for the material recycling of selected multi-material textile streams. In multiple case studies, pre- and post-consumer cotton/polyester textiles from the Austrian SME sector were investigated to close the material cycle from raw material back to raw material. The case studies used a new approach involving the enzymatic hydrolysis of cellulose. Using this approach, cotton can be converted into glucose and polyester remains as the only polymer and is thus accessible for a rather easy recycling process. The obtained glucose can be used as a raw material for different platform chemicals. The project team successfully demonstrated the functionality of the entire processing chain by the complete removal of cotton from the textile and the weaving of new towels with the recycled polyester.

1. INTRODUCTION

The textile industry is growing worldwide, and soon, the world's fibre production will pass the 100 million ton per year mark (Cirfs, 2016). However, this growth is accompanied by a reduction in the average lifetime of textiles before disposal (Korolkow, 2015), which is probably the result of today's fast changes in fashion trends. Both of these factors contribute to the ever-growing amount of textile waste. Aggravating this issue is that many textiles consist of more than one material, which makes them impossible to recycle without significantly reducing the quality of the new product produced from recycled material (Piribauer & Bartl, 2019). These textiles are also a significant source of microplastic emission into the environment through the washing of clothing, etc. (Piribauer, Laminger, Ipsmiller, Koch, & Bartl, 2019).

An easy solution to the problem in regards to using multiple fibre types in one product would be for the producers to stop doing these types of fibre. However, this is often not a viable solution since the different materials are used for a reason. For example the use of spandex fibres is needed to fulfil product requirements regarding elasticity. Carbon fibres are blended into textiles to combat static electricity, which is often needed in medical applications. Cotton/polyester blends are broadly used since they are cheaper than pure cotton fabrics and have high durability while also retaining the moisture absorbing properties of pure cotton textiles. While the reduction in the use of these blended fabrics should be a goal, it is very unlikely that they will vanish from the market anytime soon.

To recycle these blended fabrics, the fibre materials would have to be separated or at least all but one type of material has to be removed from the fabric. There are chemical reactions that can remove certain fibre materials; however, there is always the potential for unwanted side re-

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actions, and many of the needed chemicals are potentially harmful to the environment. A possible solution can be the use of enzymes.

Enzymes are biocatalysts; they are very specific in their function and therefore only catalyse specific reactions (Palmer, 2001). While some enzymes can catalyse a biological degradation reaction for many different kinds of fibre materials, only enzymes that use polymers, which occur in nature, as a substrate reach an appropriate reaction speed for a recycling process. One of the most popular fibre material blends (Kunal & Amit, 2017), which contains both a natural polymer material and a thermoplastic synthetic polymer material, is the blend of cotton with polyethylene terephthalate (PET). The natural polymer in cotton is cellulose, and therefore, a group of enzymes, cellulases, exist that degrade cellulose into its monomeric component. On the other hand, PET is a synthetic thermoplastic material, and thus, if cotton is removed from the blend, PET remains and can be melted and re-spun into new fibres of high quality.

This idea of a circular way of removing cotton from blends and recreating new fibres and subsequently new textiles is what led to the start of the TEX2MAT project. The goal of this project is to create a practical circular economy solution that can handle the recycling of cotton/PET mixed textiles from both pre- and post-consumer sources. The recycled product has to be of high enough quality to be able to truly "close the loop", which means the creation of new textiles has to occur at the end of the process. To achieve this goal, three universities and multiple companies along the textile processing chain have collaborated.

Two distinct case studies were defined. One case study handles pre-consumer textiles from cutting rests of towel production. The other case study handles post-consumer bed sheets from a textile rental company. Both of these case studies are challenging in their own way. Pre-consumer textiles can still have waxes and esters on the fibres, which are needed for the weaving process. On the other hand, post-consumer textiles have been washed many times, which may damage the fibre materials. In the case of PET, damage to the fibre materials would be suboptimal, but for cotton, it could be beneficial for hydrolysis, resulting in easier removal.

As already mentioned, to remove cotton, cellulase enzymes are used. These enzymes are commercially available and have been optimized for maximum efficiency by the bio-refinery sector, where these enzymes are extensively used, making them ideal for use for this project. This enzyme class consists of different enzymes with three main functions. Endoglucanase breaks cellulose chains randomly along the chain, exoglucanase removes cellobiose units from the chain ends of the cellulose polymer, and finally, β-glucosidase splits the generated cellobiose molecules into glucose. These enzymes work synergistically, and high reaction speeds can be achieved (Singhania, Adsul, Pandey, & Patel, 2017).

After the removal of cotton with the cellulase treatment, only pure PET should remain. This PET can then be regranulated and, depending on the quality, can either be directly reused or recondensed to reach a spinnable viscosity. Then, fibres can be spun and yarns can be produced. These yarns can then be used for the production of new towels, and the cycle starts again.

This work consist of two parts. First, the core part of the tex2mat recycling process, the evolution of the enzymatic hydrolysis process with a pre-treatment step, is described in detail, and the experimental data are shown. The second part of this work describes the workflows of the case studies in the tex2mat process and highlights the quality of the recycling product by comparing it with the virgin material product.

2. PART ONE: FEASABILITY STUDY

The most crucial part of the project was the development of the technology needed for the full degradation of cotton in PET/cotton blends. Early experiments showed that just using cellulase enzymes without further treatment did not result in acceptable degradation speeds and material purities. Therefore, pre-treatments were tested, and an acceptable process flow was found. In this chapter, the steps taken from lab-scale tests to until the production of the first kilograms of pure PET material are shown by describing the biggest breakthrough experiments that contributed to the success of the project.

2.1 Methods

2.1.1 Identification of the material composition

To check the purity of the used cotton/PET mixed fabrics before and after enzymatic treatment, the sulphuric acid procedure according to DIN 1833-11:2010 (Din, 2010) was used. In the procedure, 75% sulphuric acid was heated to 50°C in a beaker. After adding approximately 1 g of the textile to be tested to the acid, the beaker was periodically shaken for one hour. Then, the textile was strained, neutralized, dried and subsequently weighed. Since only cotton was removed with this procedure, the PET content could be determined.

2.1.2 Mechanical preparation of fibre materials

For the maximum surface contact between the fibres and the fluid in the enzymatic hydrolysis step, the textile structure of the raw material had to be disintegrated. To achieve this disintegration, it was to be necessary to grind the raw materials in a standard cutting mill using a trapezoid screen with a screen size of 0.5 mm. Using larger screen sizes only reduced the size of the textile structures without breaking them apart into single fibres.

2.1.3 Pre-treatment of fibre materials

To significantly reduce the time needed to completely hydrolyse cotton, pre-treatment steps were necessary. Different pre-treatments were tried in the course of the project. Alkaline pre-treatments were deemed the most effective, the most promising of which was soaking in sodium hydroxide solutions of approximately 20%. As a result, the hydrolysis speed was accelerated from needing more than a month to merely needing a few hours for complete degradation in the hydrolysis step. However, it should be noted that sodium hydroxide also damages the PET fibres.

For the actual pre-treatment procedure, it was therefore important to keep the contact time between the textiles and the NaOH as short as possible. The pre-treatment was performed at room temperature simply by soaking the textiles in a large enough amount of NaOH solution to completely cover them. After one hour, the treatment was quickly stopped by pressing the NaOH out of the textiles and, in the case of the lab-scale experiments, thoroughly washing them with tap water until a neutral pH value was achieved.

In the case of the scaled-up experiments, the wash was only repeated two times, leaving a varying amount of NaOH in the textiles, which was used for the in situ preparation of the buffer solution needed for the enzymatic treatment.

2.1.4 Cellulase activity assay

The filter paper assay (FPA) was used as recommended by IUPAC (Ghose, 1987). Rolled filter paper (7.5-75 mm2, approximately 50 mg each) was submerged in a glass tube with 1 mL of 50 mM sodium citrate buffer at pH 4.8. Therefore, 100 L of enzyme (diluted 1:1000) was added to the substrate. The reaction was stopped at different time points (0, 5, 10, 20, 40, and 60 min) by adding 500 L of 1 M NaOH. Specifically, for the first time point at 0 min, NaOH was added before the enzyme, which was considered to be the reaction blank. Then, 3,5-dinitrosalicylic acid (DNS) reagent was added, and the sample was boiled for 5 min, followed by the addition of 1 mL of mQ-H2O (Miller, 1959). Two hundred litres of each sample was transferred into a 96 well-plate, and the absorbance was measured at 540 nm. All measurements were conducted in triplicate.

2.1.5 Enzymatic hydrolysis

For enzymatic hydrolysis, a constant pH value of 5 and temperature of 55°C needed to be ensured. These parameters were determined in pre-experiments not described in this work, but looking at the manual of a commercially available cellulase mixture, these parameters seem to be a good fit for various cellulases (Novozymes A/S, 2019).

To provide a constant pH throughout the entire process, in the case of the lab-scale experiments, a 50 mmol/l citric acid buffer was made with citric acid monohydrate, water and sodium hydroxide to adjust the pH to the exact value. These lab-scale experiments were performed in one-litre glass bottles that were heated to 55°C in a water bath. After reaching the desired temperature, the fibres and the cellulase enzyme mixture were added to each bottle. Each bottle was periodically shaken for the duration of the experiment.

For the larger scale experiments, the buffer solution was made by first dissolving the citric acid monohydrate in water directly in a 60 l reaction vessel and then adding the unwashed pre-treated fibre material, which resulted in an increase of the pH value since the material was still alkaline from the pre-treatment. Adjustments to reach pH 5 were subsequently made by adding more monohydrate or sodium hydroxide as needed. The enzyme was only added after reaching the correct pH value; otherwise, the enzyme could have been damaged. The reactor was heated with a stainless steel pipe spiral that was connected to a pump and a temperature controlled water bath. The reactor was continuously stirred throughout the experiments.

2.1.6 Washing and drying of fibre materials

For the one litre laboratory-scale experiment, drying was simply a matter of rinsing the material on a very fine mesh with tap water, followed by drying on a piece of paper towel without any heating. The mass of the remaining solid fraction could then be easily determined with a laboratory scale.

Washing by hand and subsequent air-drying was, however, not sufficient for the material in the scaled-up experiments and resulted in the growth of mould while drying. Therefore, the material was placed inside a 100% cotton pillow cover and was washed inside a conventional top loading washing machine using a custom detergent. Immediately afterwards, the material, still inside the pillow cover, was dried in a conventional tumble drier for multiple hours. Not only did this process result in very clean and white material, it was also the first step in the direction of industrial application.

2.1.7 Analysis of PET (DSC IV)

Differential scanning calorimetry (DSC) analyses were performed on several material samples. The measurements were performed according to the standard DIN 11357-1:2017 (Din, 2017). In polymer science, DSC analyses are an important method to determine the thermal properties and crystallinity of polymers. In the testing, two samples, the actual sample and the reference sample, sealed in separate crucibles were heated using a special temperature profile. The maximum test temperature Tmax was chosen to be 280°C, which is the standard test temperature for PET and allows for covering all the important thermal effects without damaging the material. The heating rate was constant at 10 K/min, which is a standard value.

Due to the specific heat and endothermal and exothermal processes, a difference in the temperature for the sample and the reference occurred. From this temperature difference, a heat flow from the sample was calculated, and certain material characteristic thermal properties (glass transition temperature T_G, melting temperature T_M, melting enthalpy, etc.) could be determined.

2.2 Proof of Concept

The hydrolysis trials started at the laboratory scale with reaction vessels of approximately one litre. To provide a large contacting surface, both the pre- and post-consumer cotton/PET mixed textiles were milled to a small particle size, as described in the methods section.

Subsequently, hydrolysis was performed with the specimen according to the method described in the previous section. Approximately 10 g/l cellulase enzyme with an activity of 1850 U/ml was used (results to approximately ~20.000 U/l in the reaction liquid). In addition to the milling of the fibres, no further pre-treatment was performed in these first trials. It was immediately evident that the speed of hydrolysis was far too slow for commercial use, as seen in the untreated cotton line in Figure 1. The untreated materials lost only approximately 17% of their weight in 24 h with a reduction of speed over time, suggesting treatment times of well over a month.

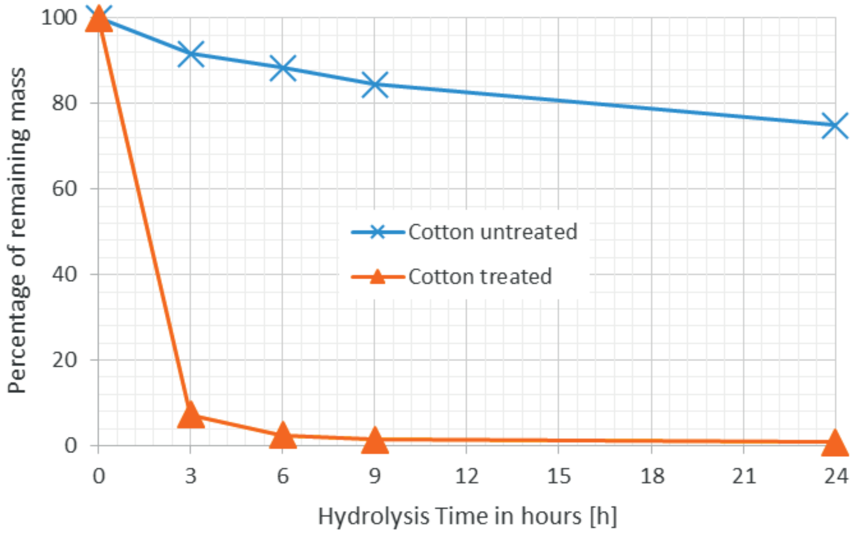

FIGURE 1: Residual mass of textile after hydrolysis for both treated and untreated materials.

Due to the insufficient grade of degradation in the experimental timeframe, various pre-treatments of the textile materials were tested. Alkaline pre-treatments turned out to be the superior choice due to their low cost and high effectiveness. The best choice after multiple tests seemed to be 20% NaOH, and an exemplary comparison between treated and untreated fibres can be seen Figure 1. After 3 hours, less than 10% of the material remained, and after 24 h, there was less than 1% left, which mainly consisted of seed capsules, etc. The next step was optimization of the enzyme concentration. The concentration was step-wise lowered from 10 g/l until a significant drop in the hydrolysis speed was observed. In Figure 2, it can be seen that even with concentrations as low as 1 g/l, 99% degradation was achieved. While for the larger scale experiments, high concentrations above 5 g/l were used as a safety precaution.

Due to these data confirming that the process could work in practice, the next step was a scale-up to generate enough material for the production of fibres. For this scale-up step, a 60 l barrel was modified with a heat exchanger and a stirrer (Figure 4 left). With this reaction vessel, 3 kg of the cotton/PET textile substrate could be hydrolysed at the same time. Over the course of the project, a total of approximately 30 kg of pure PET was created using this procedure. This amount was necessary to create a sufficient amount of PET for spinning trials and the creation of new textiles in the next step.

Before the full amount needed for textile production was produced, however, the suitability of the created rPET (recycled PET) for spinning fibres needed to be checked. The DSC method was chosen for this check because thermal values allow conclusions to be drawn regarding differences between samples of different origins and what has influenced or damaged the material during processing and usage. The DSC method also hints at potential substantial (molecular) degradation. The material used in this project originated from different sources in terms of the raw material, application and lifetime and went through a highly alkaline pre-treatment. Therefore, the aim was to determine whether there was a significant difference or decrease in

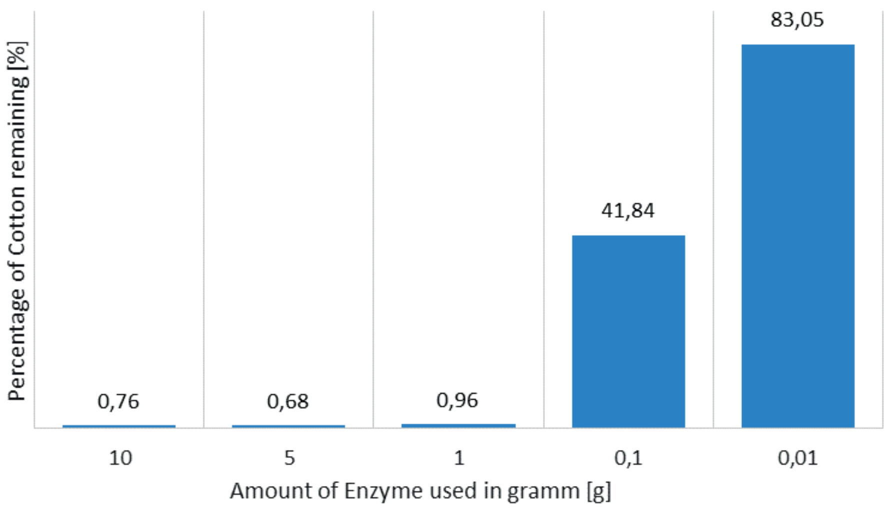

FIGURE 2: Residual mass in percent after 24h of hydrolysis for different enzyme conecentrations.

the molecular weight of the various samples. A shift in the TM to lower temperatures is an indicator that the molecular weight has been reduced (Berger et al., 1993).

Samples were chosen along the complete value chain to observe any changes or influence due to the processing steps and usage, which included virgin fibre grade PE and the rPET created in the enzymatic process, and the fibres spun from said rPET were also included as a reference, as shown in Figure 3.

As a result, no significant differences in the T_M (peak value) or melting enthalpy (normalized integral value) were observed between all of the samples. Thus, it can be assumed that the various samples did not suffer significant molecular degradation (molecular weight) and spinning should be possible. Further spinning trials confirmed that despite the greyish colour of the once white PET after regranulation, the material was still suitable for spinning into new fibres, as was proven by the first round of spinning trials. The re-spun fibres can be seen on the right side of Figure 4. With these results, the proof-of-concept was finished, and the case studies were started with the developed Tex2Mat workflow.

3. THE TEX2MAT WORKFLOW

After successfully hydrolysing all of the cotton in the blend at the 60-l scale (netting approximately 1.5 kg PET per batch) on a reasonable timescale, the case studies could start in their entirety. In the sections chapters, all of the steps in the procedure are described, and it is discussed why these steps are necessary and what the goal of each step is. Additional methods used for the full workflow are discussed directly in the corresponding sections.

3.1 Shredding and milling

One of the most important components of the enzymatic hydrolysis is ensuring the contact between the single textile fibres and the liquid containing the enzymes. Therefore, the textiles were first shredded to a size of approximately 5 cm squares and were then milled in a cutting mill with a screen size of 0.5 mm. This process resulted in the complete disintegration of the textile structure, resulting in single fibres only. In total, approximately 50 kg of textiles was processed in this way over the duration of the case study.

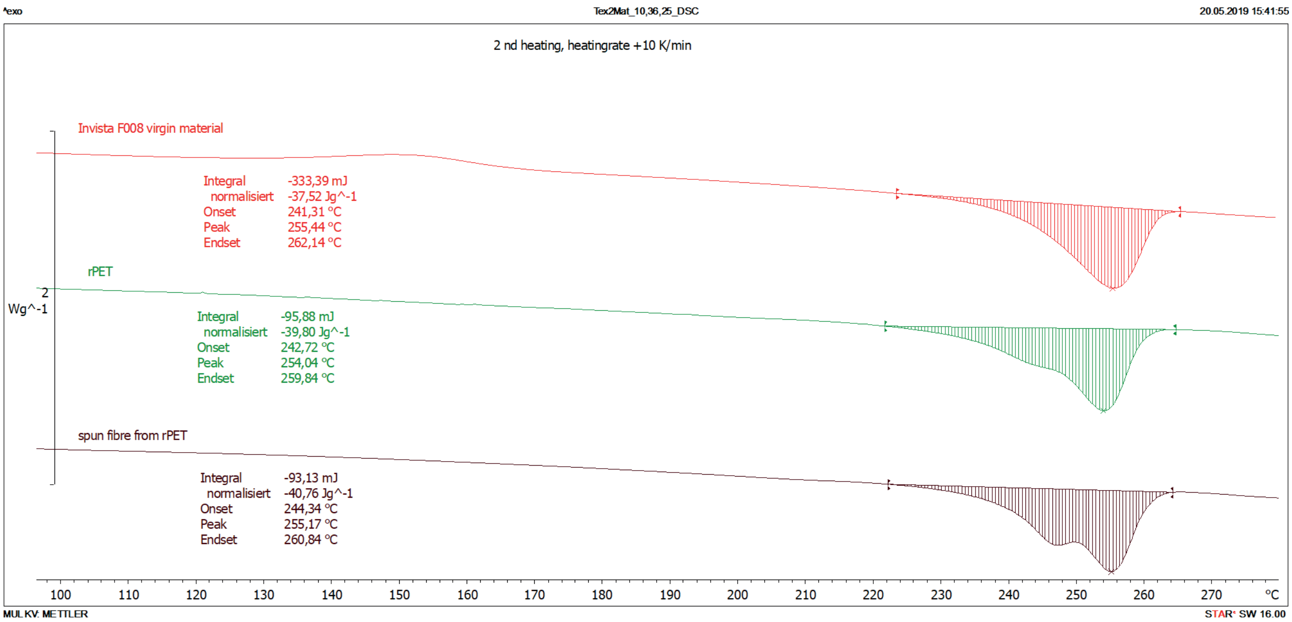

FIGURE 3: DSC-Analysis of PET virgin material (top), rPET (center) and rPET spun fibre (bottom).

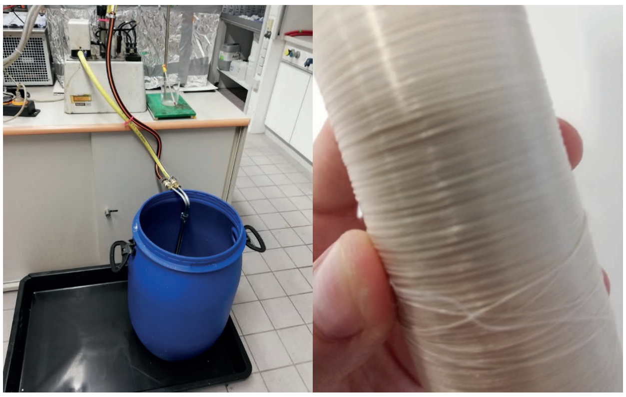

FIGURE 4: Residual mass of textile after hydrolysis for both treated and untreated materials.

3.2 Pre-treatment

Per batch, approximately 3 kg of the PET/cotton blend could be used in the 54 l reactor, and therefore, that amount was pre-treated. The pre-treatment itself was performed identical to the description of the pre-treatment for the 54-l scale in the Materials and Methods section. The treatment was performed to greatly enhance the speed of hydrolysis, as described in the previous section.

3.3 Enzymatic hydrolysis

Hydrolysis was performed in the 60 l reactor, as described in the Materials and Methods section. A hydrolysis time of 48 h was chosen along with approximately 1 kg of enzyme per batch to absolutely ensure that only a 100% pure product was generated, as a single faulty batch would have had fatal consequences for the further steps. A total of over 30 kg of PET was produced in the course of the project (approximately 25 batches total).

3.4 Washing/Drying

As described in the Materials and Methods section, the washing and drying of PET after successful hydrolysis had to be performed with the help of a conventional top loading washing machine and a tumble dryer. Since the fibres were very fine, they were placed inside a pillow cover, which allowed water and detergent to reach the fibres but did not let the fibres escape, which would have resulted in a major loss of material.

3.5 Granulation/recondensation

Before the material was regranulated, first the intrinsic viscosity was measured with a LMI5000 melt flow indexer to obtain a reference point. Then, the material was dried with a Piovan hot air dryer at 150°C for 4 hours since it is essential that no humidity remains. The material was then granulated with a Dr. Collin E25P, and the IV was measured once again. Since the IV was usually below 0.6 dl/g, the material needed to be recondensed, for which a recoSTAR SSP 1800 was used.

Multiple PET batches were processed with this scheme. To give an example, one of the first batches behaved as follows: the IV at the beginning was way too low, with a value of 0.47 dl/g, and after regranulation, it increased to 0.542 dl/g, which was still on the lower end for spinning fibres (MacDonald, 2003). After an additional six hours of treatment in the SSP, the desired value of 0.602 dl/g (over 0.6 dl/g) was reached, and the material was sent to the next production step.

3.6 Spinning

After the PET was properly adjusted, spinning trials were performed. All the experiments were conducted on a pilot spinning line "TS-32" with a 32 mm screw diameter. The spinning temperature was set to 265°C, and the final drawing process speed was 100 m min-1. The experiments started with pure virgin PET to clean the machine; then, spinning with a 25% recycling content was started, and the spinning proceeded for over one hour. After a cleaning procedure with virgin material, a 50% recycled content fibre was obtained.

Additionally, fibres with 100% virgin material were also produced to have a point of reference for further tests. Enough of these monofilament materials could be produced to proceed with weaving trials.

3.7 Weaving

As a product for the weaving trials, a standard bath towel, as is usually produced at the facilities of one of the project partners, was chosen as the desired product for the test. Due to the constraints in regard the amount of available material, it was decided to use the produced recycled PET fibres only in the weft since a too large amount would have been needed for use in the warp.

For weaving of the desired towel design scheme in industrial production, a 50/50 PET/cotton yarn is used, consisting of twisted staple fibres. Creation of such a yarn was unfortunately not feasible at such a small scale; therefore, a 100% PET yarn consisting only of the newly produced monofilament fibres and commercially available 100% cotton yarn were used in the weft together to reach the original mixture. The polyester fibres had to first be twisted with themselves to create a pure polyester yarn. In this way, yarns were created for all three material fractions (0%, 25%, and 50% recycled content, respectively). Since the difference in the yarn structure could make a major difference in the mechanical properties, it was decided to use our 0% recycled content material (100% virgin material) as our point of reference for any comparisons instead of a commercial 50/50 blended yarn.

Without any mayor complications, towels could be produced from all three materials, and there were no significant differences in the processing between the 50% recycled content fraction and the pure virgin material fraction. In Figure 5, the weaving of the 50% recycled content towels is shown.

3.8 Mechanical Tests

To test the properties of the produced towels, a multitude of test procedures were performed. Two of these procedures are described in this work. The first test procedure presented is the washing trial, in which the material was industrially washed multiple times and the change in size of the towels was observed.

The other test procedure investigated the maximum strength the towels were able to withstand with the help of a tensile strength machine. In such a machine, the towels were pulled apart until mechanical failure.

3.8.1 Washing test

The washing experiments were conducted with a Senking P50-16K industrial washing setup consisting of a washing machine and a tumble dryer, as seen in Figure 6.

The washing temperature was above 60°C with a peracetic acid-based disinfectant. The drying temperature was 170°C at the entry and 110°C at the outlet.

For the procedure, every towel was marked, and the length and width were noted. The washing/drying procedure was then performed 36 times, and after each full cycle, the lengths and widths were again recorded. On average, the towels showed shrinkage of approximately 12% in length and 13% in width, as seen in Table 1. All the materials seemed to be approximately the same regarding the length shrinkage, and only small differences could be observed regarding the width shrinkage. Compared to the

FIGURE 5: Weaving of the 50% recycled content towels.

FIGURE 6: Industrial washing machine Senking P50-16K.

usual shrinkage values of approximately 7% for towels according to the experience of the washing company, these numbers may seem high. These high numbers can, however, be attributed to the low stretching that was achieved in the laboratory spinning machine compared to industrially produced fibres, which is especially evident considering that the virgin material fibres produced showed the same large shrinkage.

3.8.2 Tear resistance (cross)

To check the mechanical stress that a towel is able to withstand, the tear resistance was measured. With a tensile strength testing machine, the towel was pulled cross to the direction of the longer side until the towel ripped apart. Every measurement was repeated two times, for a total of three tests per towel. On average, the tear resistance of the virgin material towels was 185 N/cm, and that of the 50% recycled content towels was 189 N/cm. Considering the calculated standard deviations (1 N/cm and 4 N/cm, respectively), these values should not be considered to be significantly different.

4. CONCLUSION AND OUTLOOK

The results described in this article impressively show that the new TEX2MAT process is suitable for recycling PET/cotton blends. This material mixture, which is very common on the market, cannot currently be recycled, but in the best case, this material ends up in an energy recovery process. With the process presented here, cotton, the fibres of which are often already damaged, can be processed into a platform chemical.

After the removal of cotton, PET remains as a pure material, making recycling possible, similar to PET bottles. The difficulty lies in the re-granulation of the fluffy PET fi-

TABLE 1: Lengths and widths of the tested towels after the corresponding washing cycles.

	Virgin Material		Tex2Mat RPET	
Cycle	Length [cm]	Width [cm]	Length [cm]	Width [cm]
0	92	50	92	50
1	85	47	85	47
2	84	46	84	46
3	84	46	84	46
4	83	46	83	46
5	83	46	84	45
6	83	45	83	45
11	83	44	82	44
16	81	43	82	43
26	81	43	82	43
36	81	44	82	43
Shrinkage [%]	11,96	12,00	10,87	14,00

bre material, and hydrolysis cannot be completely avoided. However, it has been shown that the IV can be brought to a value suitable for spinning fibres by post-condensation. Although the process could not yet be carried out at the industrial scale, the possibility for fibre to fibre recycling was demonstrated. Within the project, the reaction volume per batch could be increased from 1 l to 60 l.

However, to operate the process at the industrial scale, there still is work left to do in regards to optimization. For example, the chemicals used in the pre-treatment and hydrolysis have to be re-used multiple times to create a truly environmentally friendly process. This is also where the main limitations to scale-up are found. For the hydrolysis process itself, only a slightly agitated low temperature vessel is needed, in which the textile hydrolysis will take approximately a day. The investment and operation cost for enzymatic hydrolysis can therefore be estimated to be quite low compared to membrane separation for removal and re-use of the enzyme. The same is true for the purification of the used pre-treatment solutions or the mechanical presses used in the pre-treatment steps.

To evaluate the required procedures, it would be very important to use life cycle assessment tools right now when the process development is still in an early stage because at the end of the development process, changes are difficult to implement. These tools could also be crucial to find the correct dimensions for an optimal scale for the process (Koch, Paul, Beisl, Friedl, & Mihalyi, 2020).

While the development of the industrial process is not finished yet, it can be said with confidence that the process finally opens the door for a significant change in textile waste management by providing an ecological solution for one of the most important textile waste fractions. Furthermore, while the cotton/PET waste stream is very important, it is not the only stream in which enzymatic recycling can be used. Other cellulose-based fibres, such as the man-made regenerated cellulose fibres rayon and lyocell, are gaining market share (Cirfs, 2016). In addition, non-cellulose-based fibres, such as protein fibres, have the potential to be able to be hydrolysed with similar processes.

A major factor in a possible application of all of these processes is the separate collection of textile waste. If the required cotton/PET mixture is contaminated with other organic substances, the processes will potentially not work. Furthermore, the fabrics themselves have to be sorted incredibly well since, e.g., a nylon item in a cotton/PET batch will remain in the PET fraction and will have negative impact on the re-spinning of new fibres.

However, the best chance for this novel process to be used is with industrial textile waste. If the waste supply from a company only includes cotton/PET mixtures, the room for error shrinks. For example, in the case of textile rental companies, the composition of discarded textiles is well known, making them an ideal candidate as a supplier of secondary raw materials.

The interest in textile recycling technologies is currently growing. Recently, the European Commission amended the waste framework directive (European Parliament, 2018), which now takes textile waste into account. Amongst others, by 2025, a separate collection of textile waste will be compulsory within the EU. This legislation will increase the amount of separately collected textile waste and will put pressure on municipalities to improve the collection, sorting and recycling of textiles. This pressure will also give novel technologies such as the one described in this work a chance. While enzymatic recycling cannot and should not be the sole solution to the textile waste problem, it could become an integral part of the bigger picture.

The great interest in this topic was demonstrated by the fact that the TEX2MAT project was awarded the largest Austrian prize for industry cooperation, the "Clusterland Award 2019" (Ramsl, 2019).

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BENCHMARK ANALYSIS FOR RECYCLED GLASS IN AUSTRIAN WASTE MANAGEMENT

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ABSTRACT

The amendment of the Waste Framework Directive of the European Commission has set a new target for the use of recyclates. It is one of the most significant findings of this strategy that recyclates are currently too infrequently integrated into new products. Glass recycling, however, is widely accepted by the society. The chemical and physical properties of glass enable an almost 100% rate of recycling. Furthermore, the society is experienced in separating glass by colour, resulting in high-quality recycling glass for the production of new glass. Cullets are significant here. Evidently, the price of recyclates is linked to the price of primary material. Practical experience has shown, however, that pricing also correlates with different quality parameters such as degree of mixing, degree of degradation and presence of impurities. This paper examines the correlation between different quality features, how they affect the price of cullets and which quality is seen as benchmark quality for the Austrian glass industry. Experts from the Austrian processing and recycling business were interviewed about the most important parameters of their quality inspection and how they affect the pricing policy. Additionally, quality parameters for input and output material are included. Besides the interviews, specific questions on correlations between the price and quality of cullets were e-mailed to several stakeholders in the glass industry. The main purpose of this paper was to identify the effects of different quality parameters on the pricing policy and how a benchmark is or can be defined. Experts from the glass processing industry did not confirm a correlation between price and quality, however: higher quality does not necessarily mean higher prices. Glassworks are ready to pay higher prices for higher qualities to meet their sustainability objectives or to expand their production capacities.

1. INTRODUCTION

Glass recycling is considered to be a vital step to the desired circular economy due to the special properties of glass. Those enable almost full recovery in recycling processes without any loss of material properties or quality (Aldrian et al., 2018).

Glass is an indispensable and well-integrated part of everyday life, whether in packaging or architecture and construction industry. In terms of sustainability, glass packaging is often preferred to plastic packaging. A study conducted by Neill and Williams in 2016 suggests that consumers are willing to pay more for reusable glass packaging (specifically milk bottles) if they believed them more environmentally friendly, whether they truly have a lower environmental impact or not. (Neill and Williams, 2016). The increased recycling rate required by the European Commission's circular economy package, up to 75% for glass packaging by 2030, is no challenge for the local waste management as the current recycling rate in Austria is already well above this target (European Commission, 2019). A little more than half of Austria's glass production today (about 320.000 t) consists of glass packaging which translates to more than 257,000 t substituted and therefore conserved primary materials. In 2017, the Austrian glass industry produced about 420,000 t of container glass, about 239,000 tonnes of which had been collected separated. The difference of 181,000 t can not be defined exactly. The Austrian glass industry is strongly based on export of container glass. Additionally, waste glass, which is disposed in the residual waste and not in the separate waste collection is also part of this difference (Austria Glas Recycling, 2020b). The collection volume of 239,000 t allows glass packaging produced in Austria to include an average of 2/3 of waste glass. Furthermore, the production of glass in Austria mainly concerns packaging glass. A collection quota of >80% is achieved, with respect to the

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data provided by 'Austria Glas Recycling' a company by ARA (Austria Glas Recycling, 2020a; BMLFUW, 2017; WKO, 2017). The recycling quota for glass packaging has been rising the last years to 84,1% in 2018 as well as the recovery quota to 88,4%. These numbers are based on recycling and recycling in addition to energetic utilization, respectively (BMK, 2020). Meanwhile the average recycling quota of glass in the EU was around 70% in 2017 (WKO, 2017).

Advantages of glass recycling are the reduced energy costs and emissions which lead to a more sustainable overall approach to packaging. For every 10% waste glass in glass packaging production the energy costs decrease by 3% and the CO_2 emissions by 7%. Pursuant to Art. 2 (4) of the Austrian Abfallwirtschaftsgesetz (AWG 2002), glass packaging is classified as waste requiring separate collection. In Austria, about 80,600 containers for both white and stained glass are provided for separate collection and since Austrian citizens have separated glass for over 40 years now the system is well integrated into society (BMNT, 2017; WKO, 2017).

In contrast to the limited use of plastic recyclates in new products, recycled glass is one of the most important components in producing glass packaging. Today, about 60% of new container glasses are made of recycled glass. Green container glass even contains up to 90% of recycled glass (Initiative der Glasrecycler im Aktionsforum Glasverpackung, 2019).

Recycling of separately collected waste, with glass being no exception, depends on the willingness of citizens to separate right at the source. The purer waste glass is separated, i. e. the less its colours are mixed by neglect or on purpose, the more cullets may be integrated into new container glass (Aldrian et al., 2018). The need for high quality is evident. Austrian legislation, however, does not stipulate any quality standards for waste glass or minimum requirements for oven-ready cullets.

With this paper the requirements for waste glass are being examined, the correlation between price and quality of waste glass and the quality benchmark is evaluated. In addition, quality requirements for input material and treated glass as well as minimum requirements for oven-ready cullets are included. The importance of quality control and its implementation is underlined, focussing on container glass as bottles, glass packaging and cosmetics glass fed into separate collection systems.

2. METHODS

A market analysis of glass cullets was conducted by the means of telephone calls or via e-mail to identify the quality benchmark in glass recyclates.

To analyse the correlation between price and quality, glass processing plants, glassworks, manufacturers of preparation equipment and associations dealing with glass recycling received a specially designed assessment guide as well as a request to provide any product specification sheets that are given to suppliers for i.e. flint, amber or green glass - preferably container glass.

For the telephone interviews, a specific questionnaire with open questions was designed to address the correlation between price and quality of waste glass cullets, targeting the following questions:

- Which types of waste glass are processed or used?
- Is there a way how to assess the composition of input and output flows?
- Which contaminants have to be removed and how is waste glass processed? What is the amount of impurities and its upper limit beyond which product quality and cost-effectiveness of the treatment are impaired?
- What are the quality requirements for the products and how is quality control performed?
- Which developments/trends can be identified and derived in the market for cullets?
- Which unexploited potentials are currently identified in the field of waste glass processing and recycling?
- How could the quality of the waste glass collection system be improved?

Stakeholders who were not reached by phone received compressed e-mail editions of the above questions, followed by additional questions or brief telephone calls if any uncertainties or significant new findings occurred.

In addition to the then following personal discussions with representatives of the companies listed above, the remaining German glass industry was approached with the following shortened version of the questionnaire above consisting of largely the same topics:

- Contaminants which are particularly troubling for the production process and their limits
- Quality control of the inputstream (How? Where? Which parameters? Limits?)
- Quality control of products and important parameters
- Correlation between price and quality of the waste glass cullet

Altogether, 15 different stakeholders of the glass industry responded whether via e-mail or telephone. Three phone calls were made, reaching two glassworks and one manufacturer of preparation equipment. About 30 e-mails were sent to glass-processing plants, glassworks, manufacturers of preparation equipment and glass recycling associations in the process, resulting in a return rate of approx. 45%. In the end two glassworks, six glass-processing companies, one manufacturer of preparation equipment and three associations responded. Figure 1 shows the distribution of the consulted companies by stakeholder affiliation in the cullet industry. 40% of glass processing companies, 27% of glassworks, 20% of associations and 13% of manufacturers of preparation equipment participated in the survey. In the Appendix A the results of two telephone responses are listed, in Appendix B the results of six e-mailed answers are shown. For the other received answers, the participants have not given their permission to publish their answers neither in public listing of the company, nor in anonymous way.

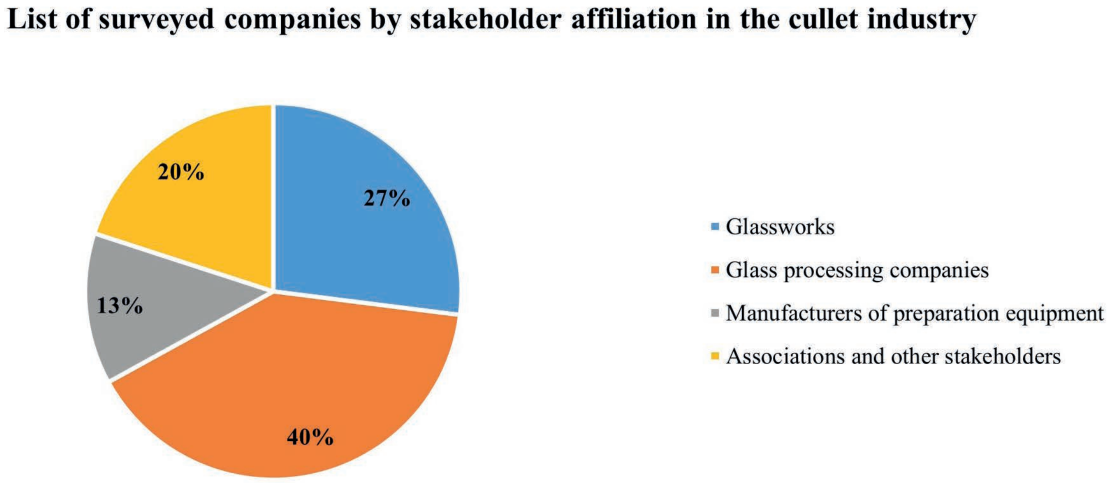

FIGURE 1: The 15 surveyed companies by stakeholder affiliation in the cullet industry from Austria as well as Germany.

3. RESULTS AND DISCUSSION

3.1 Quality requirements

Currently, Austrian law does not stipulate any minimum quality specifications for waste glass to be used in the container glass industry or for oven-ready cullets. The Bundesverband Glasindustrie e.V. (BV Glas), the Bundesverband der Deutschen Entsorgungs-, Wasser- und Rohstoffwirtschaft e.V. (BDE) and the Bundesverband Sekundärrohstoffe und Entsorgung e.V. (BVSE) have jointly developed general specifications regarding the quality of waste glass and oven-ready cullets which are observed by all European glassworks. These associations have also tackled waste glass fed to processing plants and developed a technical specification for hollow glass, enforcing standardisation in glass recycling and glass processing. Additional agreements are made between waste glass processors and glassworks generally exacerbating the values defined in the 'Quality requirements for cullets to be used in the container glass industry' guideline T 120 or adding additional parameters. These strict requirements allow the glass industry to produce new products of consistent quality.

Below, the quality requirements for waste glass are described in more detail. Furthermore, the product specifications of hollow glass developed by the associations mentioned above are examined. Finally, the minimum quality specifications for oven-ready cullets are specified. The following descriptions refer to the source 'BDE (2019)' in the References.

3.1.1 Product specifications for hollow glass

Since this paper discusses container glass, the product specifications of hollow glass are given first. Contaminant limits and purities of the collected fractions are particularly emphasised because a certain purity of the fraction is essential for its use by glassworks. The following product specifications developed by the above mentioned associations apply:

- Product specification of collected hollow glass, type flint glass TR 201 (BVSE & BDE, 2013a)
- Product specification of collected hollow glass, type green glass TR 202 (BVSE & BDE, 2013b)
- Product specification of collected hollow glass, type amber glass TR 203 (BVSE & BDE, 2013c)
- Product specification of collected hollow glass, type stained glass TR 204 (BVSE & BDE, 2013d)

The product specification sheets refer to container glass such as bottles, jars, pharmaceutical and cosmetics glass from glass packaging collection points. A comparison of the listed product specification sheets is shown in Table 1.

3.1.2 Quality requirements for cullets used in the container glass industry

The following refers to the 'Quality requirements for cullets to be used in the container glass industry' guideline T 120 (BV Glas et. al., 2014).

Container glass from households, industry and production such as bottles, jars, pharmaceutical and cosmetics glass (soda-lime glass) are included. With regard to the state of the art the following substances should not be included: lead glass, non-processed safety glass, glass ceramics, lighting glass, TV glass, quartz glass, borosilicate glass, other leaded glasses and all substances that may interfere with the melting and production process.

The quality of the processed cullets must be maintained during transport. For storage and delivery, ensuring that the processed cullets are not exposed to contamination is vital, including but not limited to suitably cleaned means of transport (e. g. lorries) and clean collection bins.

The 'Quality requirements for cullets to be used in the container glass industry' guideline T 120 (BV Glas et. al., 2014) is seen as the quality benchmark for cullets in the container glass industry by the surveyed companies.

Maximum content of impurities

The limits given in Table 2 represent monthly averages or averages of at least 250 tons of glass which should not be exceeded. All values are upper limits as stated in the T120 'Quality requirements for cullets to be used in the

TABLE 1: Products specifications of hollow glass (BDE, 2019).

	Flint glass[1]	Green glass[2]	Amber glass[3]	Stained glass[4]
Purity in wt.% Glass according to the specified amounts of contamination with other types of glass	Min. 97.00% Max. 1.00% Max. 2.00%	Min. 97.00% Max. 1.00% Max. 2.00%	Min. 97.00% Max. 1.00% Max. 2.00%	Min. 97.00% Max. 1.00% Max. 2.00%
Amount of other coloured glass	Max. 3.00% Green: 1.00% Amber: 2.00%	Max. 15.00%	Max. 15.00%	Max. 10.00%
Impurities				
Maximum amount of impurities (max. 1% in total)				
Ceramics, stone, porcelain	0.15%	0.15%	0.15%	0.15%
Other waste (cans, plastics, cardboards)	0.10%	0.10%	0.10%	0.10%
Maximum amount of other types of glass (max. 2% in total)				
Quartz glass	-	-	-	-
Borocsilicate glass	-	-	-	-
Glass ceramics	0.01%	0.01%	0.01%	0.01%
Glass from electronic equipment	0.005%	0.005%	0.005%	0.005%
Lead crystal glass	0.01%	0.01%	0.01%	0.01%
Wired glass	0.20%	0.20%	0.20%	0.20%
Car glass	0.20%	0.20%	0.20%	0.20%
Flat glass	2.00%	2.00%	2.00%	2.00%
Ampoules, injection glass	2.00%	2.00%	2.00%	2.00%

[1] Product specification of collected hollow glass, type flint glass TR 201 (BVSE & BDE, 2013a)
[2] Product specification of collected hollow glass, type green glass TR 202 (BVSE & BDE, 2013b)
[3] Product specification of collected hollow glass, type amber glass TR 203 (BVSE & BDE, 2013c)
[4] Product specification of collected hollow glass, type stained glass TR 204 (BVSE & BDE, 2013d)

TABLE 2: Maximum content of impurities.

Ceramics	20.0g/t
Non-ferrous metals	3.0g/t
Ferrous metals	2.0g/t
Glass ceramics > 10 mm	5.0g/t
Loose organics	300.0g/t
Moisture	2.0%
Heavy metals (Pb, Cd, Cr(VI), Hg) (considered as guidline)	200.00ppm

container glass industry'. Limits refer to flint, green, amber and stained container glass, respectively. (BDE, 2019)

Ceramics, stone, porcelain (CSP)

Avoiding a high CSP Value decreases the risk of explosions due to reactions of ceramics in the melt. The melting tank and process are very sensitive and precisely tailored to the mixture, fed via special feeding systems. Each tank is customised and adjusted to the requirements of the glassworks. However, due to their higher melting point, ceramics, stones and porcelain do not completely melt in the tank therefore they remain in the processed glass, creating inclusions.

These inclusions may cause a number of issues including breakage, defects and/or deformation of container glass. In addition, these visible defects cause rejection of the finished product. Therefore, the amount of reject increases at what is called the 'cold end' where each glass product is precisely measured and checked for faults using special inspection systems. If a defect is not detected, it may break at the bottler's, during transport or even at the end-consumers. With carbonated beverages, risk of breakage is particularly high (e. g. champagne bottles). Thus, inclusions can be a safety-related defect since their degree of hardness will differ from that of glass, resulting in voltage differences which may ultimately cause the glass to burst.

Metals (ferrous and non-ferrous)

Due to the comparatively high melting temperatures of around 1,500 – 1,600 °C metals can cause various problems in the melting process. Long-term damage of the melting tanks is is one of them as the metals sink to the bottom of the tank and rotate there due to the flow of the refractory material. Additionally, damage to the bottom of the melting furnace may even be due to metals melting at lower temperatures, such as lead. Furthermore, metals that are hydrophobic to the refractory material, such as ferrous metals, may aggregate into a ball which can lead to a rotational movement causing holes in the bottom of the tank. Metal-stopping layers are used to prevent this kind of damage.

Loose organics

In general, loose organics cause carbon input. The presence of carbon is basically not a problem, but varying contents of carbon may constitute one. If the content is too high, colours may begin to drift, causing large-scale production losses or deviations from the colour specifications of the customers. Carbon triggers an oxidative process in the melt that may cause the colour of the glass to change, especially with flint glass that may then be tainted grey or yellow. On the other hand, excessive or irregular accumulation of organics on the surface of the melt may cause foaming or blistering. When the molten material is surface-fired, too little heat is induced by the foam barrier. The result is irregular melting, higher energy input and/or longer residence times of the material in the melting tank. In extreme cases, nests from slightly molten glass material may arise.

The limits set to loose organics have several arguments. For energetic and environmental reasons, melting tanks are now operated with very small excessive air. Organic content influences this residual oxygen content. Therefore, in practical application, recycled waste glass is stored for about six weeks before it is mixed with molten glass, to avoid any influence of organics (fermentation).

Moreover, loose organics have a tendency of fine pieces of broken glass adhering to them. Due to their high organic content, these fine shards often cannot be completely used as secondary raw materials so that the recycling rate for waste glass will fall short of 100% (Aldrian et. al., 2018).

Moisture

Moisture is of particular importance for the sorting and preparation of waste glass. If the moisture content is too high, the cullets are difficult to screen and the quality of sensor-based sorting will suffer significantly. As a result, a constant transport speed on slides can no longer be guaranteed. If a cullet is slowed down during the sliding process by an 'adhesive' water film on the chute in the detection unit, it can no longer be blown away by compressed air in the separator on time.

On the one hand the limit to moisture is set to avoid excessive variations, on the other hand for the treatment process, a certain amount of moisture is beneficial in preventing dusting and segregation. However, this moisture must be evaporated and therefore should be kept at a minimum.

Heavy metals

Since glass can be melted again and again, there is still a lot of 'vintage' glass participating in the circulation, melted before lead-sorting equipment had been installed and the limits had been in place. This 'vintage' glass still contains a lot of lead which is slowly reduced by dilution processes. Lead was once used to increase the gloss of the glass (up to 35% of lead oxide (PbO)). During melting, evaporation may be significant in regard to environmental limits. However, exceptions are sometimes made to permit higher emission limits. Heavy metals basically do not affect material properties or consistency.

This limit is rather politically motivated, driven by the limitation of heavy metals established in the plastics industry. In fact, there is no elution from glass, but consumers prefer glass without heavy metals. Of course, accumulation of heavy metals in glass containers is undesirable. For example, basic contaminations of approx. 200 ppm of lead oxide (PbO) are already present in new bottles imported from Asia according to one of our interviewees. In addition, the glassworks are subject to legal stipulations which should not be violated by the use of recycling cullets. For example, Art. 4 (1) of the Austrian Verpackungsverordnung of 2014 states that packaging with a concentration exceeding 100 ppm by weight of lead, cadmium, mercury and chromium (VI) is prohibited unless lead crystal is concerned (Verpackungsverordnung, 2014).

Experts from waste-glass processing have supported restrictions concerning lead in packaging i.e.

For the discharge of lead-containing materials, a lot of equipment is already installed. Cullets containing lead oxide are detected by optoelectronic devices, such as UV cameras.

Heat-resistant glass

Contamination of the ember stream with heat-resistant glass prevents the cutter from working properly. The finished conditioned glass is shaped into gobs by the feeder machine and then cut by a scissors mechanism at the end of the drop-forming process. If larger melting relics – pieces of CSP or refractory materials - should occur here, the droplet cannot be properly cut, disrupting therefore the production process. Heat-resistant glass and glass ceramics are hard to detect at the input controls of glassworks, requiring appropriate sorting equipment in waste-glass processing. Glassworks and waste-glass processors are well equipped with sorting units detecting and separating heat-resistant glass.

A limit value for heat-resistant glass is given in the specification sheet 'T120' under the item 'Glass-ceramic'. A separation quality of at least 90% on the part of the system manufacturer is required to avoid the presence of these materials in the ember stream. Most European glassworks tolerate a maximum concentration of 25 g/t of refractory cullets in the ready-to-melt product.

Content of other coloured glass (incoming)

The colour is determined in the grain band > 8 mm square mesh. The maximum contents of other colours for the glass types flint, green, amber and stained are listed in Table 3.

3.1.3 Requirement for oven-ready cullets

The 'Minimum quality specification of oven-ready cullets' guideline TR 310 regulates the standards for the glass industry (BVSE & BDE, 2013e).

Container glass from households, trade and production is considered. The processing plant must be operated with appropriate technology such as sensor based sorting equipment for dealing with contaminants to comply with the quality criteria of oven-ready cullets as defined in Table 4.

The 'Minimum quality specification of oven-ready cullets' guideline TR 310 is seen as the quality benchmark for over-ready cullets by the surveyed companies.

3.2 Quality control

The applying quality requirements for separately collected waste glass have already been mentioned in the previous section. The Austrian and German industry com-

TABLE 3: Maximum contents of other colours for specific glass types (BDE, 2019).

	Flint glass[1]	Green glass[2]	Amber glass[3]	Stained glass
Colour amber	≤ 0.3%	Max. 10.0%	Min. 80.0%	Min 80.0%
Colour green	≤ 0.2%	Min. 75.0%	Max. 10.0%	Min 80.0%
Other colours	≤ 0.2%	-	-	-

[1] Transition colours from white to green, acid-green and half-white, are white.
[2] All shades of green including the reduced shades of green are considered green.
Cullets in the range of 568nm to 573nm are considered reduced green hues.
[3] All shades of amber are considered amber.

TABLE 4: Requirements for oven-ready cullets (BDE, 2019).

Ferrous metals	≤ 50ppm
Non-ferrous metals	≤ 60ppm
Inorganic non-metallic, non-glass materials like ceramics, stones, porcelain or pyroceramics: • Cullets > 1mm • Cullets ≤ 1mm	 ≤ 100ppm ≤ 1,500ppm
Organic impurities like paper, rubber, plstics, fabric or wood	< 2,000ppm

ply with the guidelines of the Bundesverband Glasindustrie e.V. (BV Glas), the Bundesverband der Deutschen Entsorgungs-, Wasser- und Rohstoffwirtschaft e.V. (BDE) and the Bundesverband Sekundärrohstoffe und Entsorgung e.V. (BVSE) since there are no uniform quality standards. The parameters CSP (ceramics, stones, porcelain), ferrous and non-ferrous metals as well as heat-resistant glasses are essential when assessing the quality of cullets. As these materials have different melting temperatures to glass, they do not (completely) melt in the melting tank, causing inclusions and, hence, production problems. In addition to these parameters, the content of lead-containing glass, the organic content and the fine-grain content are important for assessing the quality of the cullet.

Input material is visually controlled by material acceptance staff. Based on their experience and expertise, the quality of the glass fragments can be assessed by paying attention to the load of undesired impurities such as residual waste, wood and CSP. In case of suspicion, lead glass is checked with an ultraviolet lamp. Quality control in laboratories is not an option for real input quantities, owing to the high amounts passing per day. Hollow glass usually derives from colour-separated collection. The main focus is on the degree of refraction of the cullets and contamination of the loading surface by subcharge, such as gravel. An essential part of input control is the colour distribution of the bulk material since mono-coloured waste glass is a much better secondary raw material than multi-coloured glass. (BVSE, 2019). Stained glass would lead to discoloration in white glass (BMNT, 2017). In addition, empirical data show that waste glass fractions in certain regions (for example, in large cities) have a lower quality meaning lower purity.

Basically, the following stages of quality control are mentioned by the industry partners participating in this study after the processing of waste glass:

- The plant operator trusts his plant and no quality control is performed.
- There is a continuous sampling in the output stream of the treatment plant. The following parameters are checked in the laboratory:
  - colour purity,
  - impurities and
  - chemical composition.
- Semi-automated sampling is performed. Continuously sampled material enters an analyser running semi-automated analyses or protocols. A plausibility check made by laboratory staff is not rendered obsolete that way.

Quality control by sensor-based machines is still rarely met, though demand by industry is rising.

3.3 Sampling of recycled glass

Sampling may be performed before delivery of the cullets or sent by the supplier beforehand. Consequently, a representative sample volume is taken, analysed and if it complies with stipulated limits the total amount will be delivered. No further sampling is then required upon receiving the goods, thus saving time. The Bundesverband Sekundärrohstoffe und Entsorgung e.V. (BVSE) and the Bundesverband der Deutschen Entsorgungs-, Wasser- und Rohstoffwirtschaft e.V. (BDE) have jointly developed a guideline for standardised sampling of cullets to ensure reproducible measurement results: 'Sampling for the use in the container glass industry' TR 101 (BVSE & BDE, 2013f).

3.4 Market study

3.4.1 Pricing structure

The market price for waste glass is primarily based on geographic conditions as mentioned by some of the interview partners. The following questions are significant: Where does the cullet come from? Where does the cullet go? Furthermore, the market price includes collection, processing and transport and is based on the primary raw material and the amount of energy consumed by processing. There are no parameters defining the purity of waste glass.

The following factors are mentioned as essential by the surveyed companies:

- Price for quartz sand/soda/lime,

TABLE 5: Prices for recycled cullets (BDE, 2019).

Glass type	Price range	Price variation
Flint glass	€ 85,- to 90,./t	+/- € 10,-/t
Amber glass	€ 85,- to 90,./t	+/- € 10,-/t
Green glass	€ 65,- to 70,./t	+/- € 10,-/t
Stained glass	€ 40,- to 50,./t	+/- € 5,-/t

- transport of the cullets to the treatment plant,
- treatment efforts,
- energy drawback from using primary raw materials in the melt and
- transport of cullets to the glassworks.

Usually, the glassworks negotiate prices with the suppliers 'delivered to the delivery point' so that the material price for waste glass will vary with the site of the processing plant and the distance to the glassworks. The prices presented in Table 5 are currently observed in Austria (recycled waste glass) by two industry interview partners although it was mentioned more than once that those can vary and are therefore to be taken with a grain of salt. Others would not want to mention numbers precisely because of these concerns.

The interviewed container-glass processing experts have claimed that prices indicated in the table above are currently tolerable. They mentioned, however, that the pricing may also turn unfavourable; for example, the price for a tonne of waste glass may drop to € 20,-. The experts agreed on the other hand that any price above € 100,-/t was hard to achieve. Moreover, prices for waste glass are generally declining due to decreasing expenses for energy investments into production.

Observations in Germany show that the price of waste glass can vary due to geographical conditions. In the south there is an excess of white glass, but fewer glassworks are established. On the contrary in the north there are many glassworks and a high demand for white glass. Therefore, the price of white glass is higher in northern Germany than in the south. Prices for green glass are distributed exactly the other way round: In the north, there is an excess of green glass, in the south there is a great demand for it. One reason for this discrepancy may be found in the brewing industry located in Bavaria. Hence, the price of green glass is higher in southern Germany than in the north.

Figure 2 shows the development of the price in euros pro ton for processed cullets. It can be seen that the price for class cullets per ton since 2002 until 2019 has continuously risen as proven by the data from EUROSTAT (2020) and WIFO (2016), although a slight drop can be observed between 2008 and 2011 which is probably caused by the world economy crisis. The price for each individual color of cullets is decreasing over the years. These data are specifically from the UK (WRAP.ORG.UK, 2008) and show a different trend than the data from EUROSTAT (2020) and WIFO (2020).

In Figure 3 the price development of the primary resources for glass production is shown. For the composition of glass, a mixture of 20% soda ash and limestone respectively and 60% sand has been used to calculate the price per ton using individual statistics for each primary re-

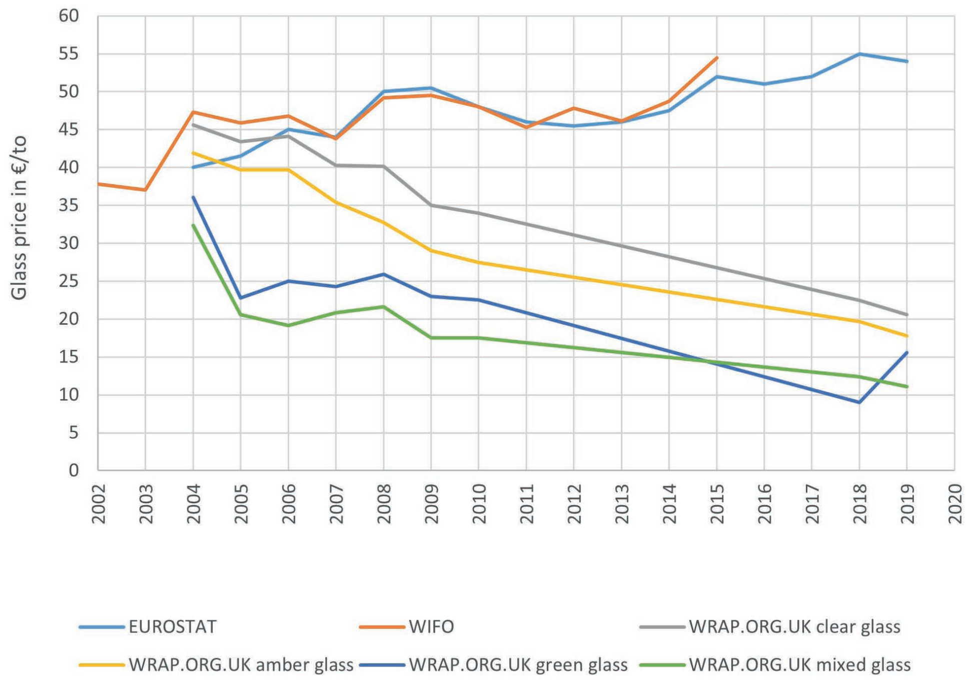

FIGURE 2: Price development for processed cullets.

source. However, no reliable data could be found for the primary resources other than sand. The price for sand is stable throughout the years starting from 2008. For the price per ton of limestone a drastic increase can be observed starting from 2014. The price per ton for soda ash is the highest out of the mixture ranging from 220 €/t to 280 €/t and peaking in 2018. The calculated price per ton of glass produced from primary resources starting from 2016 shows a slight increase of about 10 €/t through the time frame 2016 until 2019 while being in the range of 60-70 €/t, compared to 50-60 €/t for class cullets in the same time frame.

3.4.2 Correlation: price and quality

Quality does not really affect pricing policies. All suppliers face the same specifications (usually 'Quality requirements for cullets to be used in the container glass industry' T 120). This quality must be achieved in any case. Basically, higher quality of the input material does not result in a higher price: a delivery of qualities below the specified limit values is accepted but not separately priced. Exceeding the limits will cause rejection of the delivery. Waste glass recyclers will pay more for waste glass that is less contaminated because subsequent treatment will be less expensive. However, the main part of recycled waste glass from licensed collection systems is paid a standard price.

The surveyed companies observe the following relationships between price and quality:

- One is described like this: If the glassworks expands its processing and requires more waste glass, it is likely to pay a better price.
- Another is described like this: Companies are endeavoring to use cullets to produce new glass which also affects the pricing policy and results in toleration of higher prices. Their sustainability strategies pursue the goal of using a high relative amount of waste glass in their new production. The purer the cullets, the more of them can be added to the melt. The higher the amount of cullets, the lower the impact on the environment.

In Austria, prices for waste glass are subject to supply and demand as in the general market mechanism. Higher quality does not need to be more expensive. Occasionally, special glassworks producing cosmetics glass or high-quality bottles will pay higher prices for clear glass. In general, however, prices for cullets have reached a point at which the use of primary raw materials may become economically feasible again.

3.5 Quality benchmark in glass recycling

The conducted market analysis did not provide any information on a benchmark for cullets. Producers of waste glass were therefore asked to identify one. For input materials, the quality standard, which is seen as quality benchmark, used by the industry is the 'Quality requirements for cullets to be used in the container glass industry' guideline T 120. Besides, the Bundesverband Glasindustrie e.V. (BV Glas), the Bundesverband der Deutschen Entsorgungs-, Wasser- und Rohstoffwirtschaft e.V. (BDE) and the Bundesverband Sekundärrohstoffe und Entsorgung e.V. (BVSE) have jointly developed general specifications regarding the quality of waste glass and oven-ready cullets. The 'Minimum quality specification of oven-ready cullets' guideline TR 310, which is seen as the quality benchmark for over-ready cullets, is observed by all European glassworks.

4. CONCLUSIONS

Whether there is a correlation between price and qual-

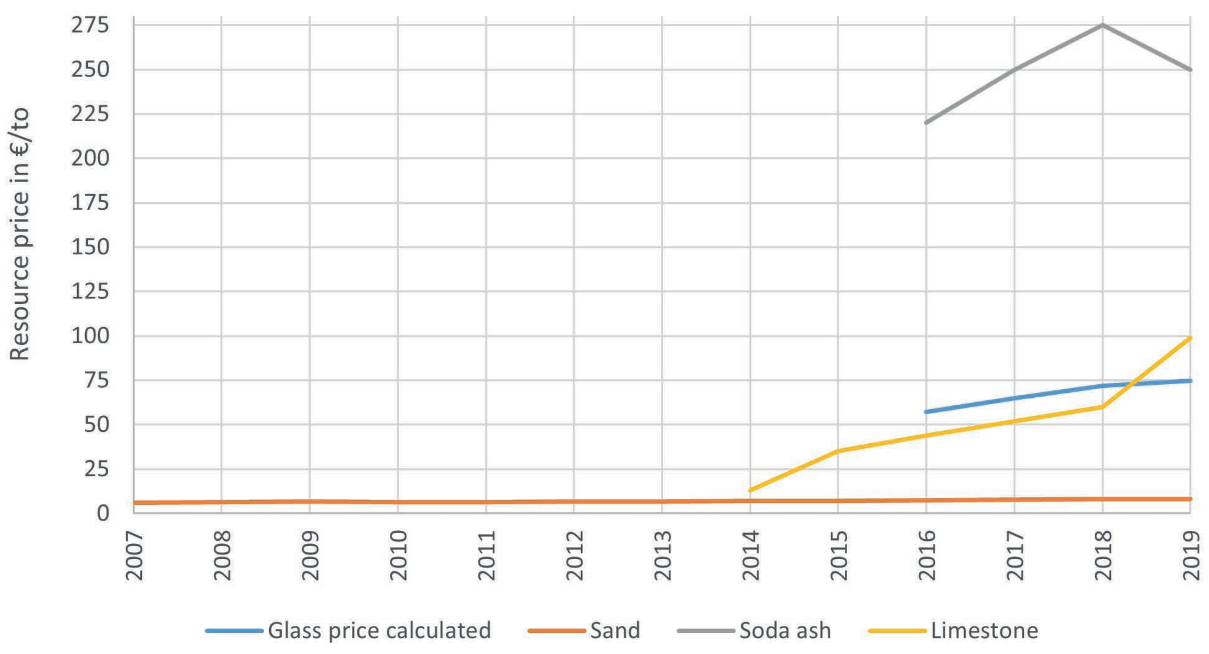

FIGURE 3: Price development of the primary resources for glass production.

ity of waste glass could not be conclusively determined. The interviewed experts from the container glass industry do not perceive an interaction of these two parameters. It could merely be perceived that a higher price for better quality will be paid if a company intends to use a high content of cullets in production to achieve sustainability or higher capacity. Therefore, the price of waste glass emerges primarily from the equilibrium of supply and demand on the general market. Furthermore, the costs for collecting waste glass, processing cullets and transporting to the glassworks enter calculations. Waste glass is naturally priced based on the price of primary raw materials and energy consumption during processing. The market price for waste glass also varies by location of the processing plant and distance to the glassworks.

Quality inspection of waste glass is usually executed by material acceptance staff. Based on empirical values, reliable assessment of the quality of delivered glass waste can be expected. This assessment is compelling for further processing. The parameters 'CSP' (ceramics, stones, porcelain), ferrous and non-ferrous metals as well as heat-resistant glass severely impact the quality of cullets. In addition to these parameters, staff consider the content of lead-containing glass, organics content and fine-grain content when assessing the quality of waste glass.

Currently, there is no minimum quality stipulated in Austria for waste glass applied by the container glass industry or for oven-ready cullets. The Bundesverband Glasindustrie e.V. (BV Glas), the Bundesverband der Deutschen Entsorgungs-, Wasser- und Rohstoffwirtschaft e.V. (BDE) and the Bundesverband Sekundärrohstoffe und Entsorgung e.V. (BVSE) have jointly developed general specifications regarding the quality of waste glass and oven-ready cullets, which set the quality benchmark in the glass industry. The agreed upon specifications are respected by all European glassworks. Therefore, the great goal of standardisation of quality requirements in glass recycling and glass processing was reached, which enabled the industry to produce while achieving consistent quality.

Finally, the use of waste glass in container glass production can be considered essential and furthermore called the most important raw material for new production. In recent years the use of waste glass was considerably and successfully pushed by waste glass processors, manufacturers of preparation equipment and glassworks. However, an important part of achieving waste glass quality happens at the source, where the willingness of citizens to separate different types of glass decides whether high-quality container glass can be produced from recycled waste glass.

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APPENDIX A: LISTS OF INTERVIEW RESULTS

1. How is the price for glass cullets determined?

Company X

The price is determined by geographical conditions. The two main factors are the origin and the delivery place of the cullets.
A gradient in price from west to east Europe is noticeable. In eastern Europe less glass is produced than in western Europe because in the west there are more glass factories. Therefore in the east the glass is more expensive. The price for primary resources also influences the price of glass cullets.
The following prices for glass cullets in Austria:

White glass 80 +/- 20 €/to
Brown glass 60 +/- 10 €/to
Green glass 40 +/- 10 €/to
Mixed glass 20 +/- 5 €/to
The price never exceeds 100 €/to

Company Y

Price is dependent on the geographical conditions; distance between glass factory and glass processer. In south Germany there is surplus of white glass and not enough glass factories. The glass factories are located in the north. Therefor the price for white glass is higher in the north. The opposite can be observed for green glass. The prices are declining due to cheaper energy. The price never exceeds 100 €/to.

2. What is the coherence between price and quality?

Company X

The goal is to use as many glass cullets as possible for glass production. The melting facility benefits form the usage of glass cullets as lower temperatures are sufficient in comparison to melting primary resources. For high quality cullets the prices is higher.

Company Y

Glass cullets with a high purity are more expensive than cullets with a lower purity. The main amount of processed glass originates from licensed collection systems and the price for that is uniform.

3. How is the quality of glass cullets determined? Which parameters are essential?

Company X

-

Company Y

On site. Parameters are: CSP, colour distribution, impurities

4. Additional information

Company X

Porcelain can influence the form stability of the container glass because it does not fully melt. This leads to bubbles and inclusions on the surface of the container glass.
White glass: the content of iron has to be kept low to achieve the white colour.
Green glass: the content of iron needs to be high for the colour. It consists of 80 % waste glass and 20 % primary resources. The melting temperature in the melting facility is 1.600 °C. Around 50 % of energy can be saved due to the lower melting point of glass cullets in comparison to primary resources.
The retention time is 24 hours. The finished bottles are analysed by a computer after cooling.

Company Y

-

APPENDIX B: Lists of results to the e-mailed questions

1. Which impurities are especially problematic for the molten glass? What are the limit values for these impurities?

Company A

The quality parameters for waste container glass cullets are CSP, purity and color distribution. They are evaluated on site (not in a separate laboratory).

Company B

An important parameter that is being analysed is the content of ceramics.

Company C

Limit values for glass (TA 120 germany); the following values do not apply for special glasses (heat resistant glass etc.)
CSP 20 g/to
Metall 5 g/to
Organics 400 g/to
PBO 200 ppm
Colored class in white glass 0-6

Company D

In Germany general quality agreements for glass between three economical bonds (BDE, BVSE and BV Glas) exist. Additional quality standards exist between waste glass processing companies and glass factories (e.g. the guidline TA 120 "quality requirements for glass cullets for usage in container glass industry").
The following quality parameters are essential: CSP, metals, heat resistant glass, lead glass, organics and fine-grained material.
A high content of the first three parameters causes production problematics and inclusion due to their higher melting point. The content of lead glass causes a high lead content in the produced container glass (limit value for lead in packaging material). Organics and fine-grained material cause problems in the controlling aspect of the process.

Company E

The input material (~ 600 to/d) is visually analysed for impurities.

Company F

	Parameter														
Quality parameter	Ceramics/Porcelain in g/to			FE in g/to			NE in g/to			Stones in g/to			Organics in g/to		
Glass type	min	max	tol.	min	max	tol.	min	max	tol.	min	max	tol.	min	max	tol.
green	0	10	+5	0	5	+3	0	5	+3	0	30	+10	0	1000	+500
white	0	10	+5	0	5	+3	0	5	+3	0	30	+10	0	1000	+500

2. How is the quality of the glass cullets evaluated? Which parameters are analysed?

Company A

They are evaluated on site (not in a separate laboratory).

Company B

The cullets are either being analysed when delivered or directly at the supplier to save time during the delivery. The delivered cullets are not accepted in the case of insufficient quality.

Company C

Limit values for glass (TA 120 germany); the following values do not apply for special glasses (heat resistant glass etc.)
CSP 20 g/to
Metall 5 g/to
Organics 400 g/to
PBO 200 ppm
Colored class in white glass 0-6

Company D

In Germany general quality agreements for glass between three economical bonds (BDE, BVSE and BV Glas) exist. Additional quality standards exist between waste glass processing companies and glass factories (e.g. the guidline TA 120 "quality requirements for glass cullets for usage in container glass industry").
The following quality parameters are essential: CSP, metals, heat resistant glass, lead glass, organics and fine-grained material.

Company E

The input material (~ 600 to/d) is visually analysed for impurities.

Company F

-

3. How is the price for processed glass cullets determined? Which parameters are essential??

Company A
The price is determined by the purity and the cost for transportation. A high purity causes a high price.
Company B
The produced container glass is continuously analysed by machines and humans. One important parameter that is being analysed is the content of ceramics. The price for cullets is determined by supply and demand; the quality is secondary. White glass: ca. 85 – 90 €/to Brown glass: ca. 85 – 90 €/to Green Glass: ca. 65 – 70 €/to Mixed glass cullets are not being used in Germany; prices never exceed 100 €/ton
Company C
Buying price is determined by the costs for collection, processing and transport; quality parameters are not considered. On delivery samples are taken and analysed (e.g. lead glass is detected by using a UV-lamp) Selling price is not generally determined by CSP content. The selling price for processed cullets is at a point at which the usage of primary resources might be more beneficial. White glass: ca. 85 – 90 €/to Brown glass: ca. 85 – 90 €/to Green Glass: ca. 65 – 70 €/to Processing costs are around 20 – 30 €/to
Company D
Selling price is not determined by quality, though the requirements are set in the TA 120. The material is not accepted by the glass factory if these requirements are not fulfilled. The price is also dependent on the distance between glass factory and glass processer. Processing cost are around 30 €/to.
Company E
The price for cullets are negotiable. High purities lead to higher prices.
Company F
-

4. What is the coherence between price and quality of glass cullets?

Company A
A high purity causes a high price.
Company B
The price for cullets is determined by supply and demand; the quality is secondary.
Company C
Selling price is not generally determined by CSP content.
Company D
The price is also dependent on the distance between glass factory and glass processer.
Company E
The price for cullets are negotiable. High purities lead to higher prices.
Company F
-

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ANAEROBIC DIGESTION OF FOOD WASTE AT VARYING OPERATING CONDITIONS

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ABSTRACT

Food waste is one of the major problems contributing to the degradation of the environment, and thus needs serious attention. Among different options, anaerobic digestion is possibly the most effective technique for managing degradable waste, and produce renewable energy and fertilizer. Despite multiple-use and benefits of the technology, its application is limited due to a few technical challenges. This study focuses on the anaerobic digestion of food waste with the addition of different percentages of digested cow manure as inoculum to it, at different total solid content in ambient temperature. Anaerobic digestion of food waste in batch and semi-continuous processes were carried out in three different trials at an average temperature range of 20-26°C: Food waste with 20% inoculum, food waste with 50%, 100%, and 200% inoculum and 10% total solid content in batch process and food waste with 20% inoculum with 6% and 10% total solids content in the semi-continuous process. During each trial, some amount of gas production was observed, however, the gas composition showed a negligible amount of methane production (maximum 13% of CH_4). There were two common problems detected in each trial: the inability to complete the methanogenesis process, and instability of the overall process due to the high degradability and acidic nature of food waste. Therefore, this study suggests that the mono digestion of food waste is not a suitable option. However, anaerobic co-digestion of food waste with different organic substrate might provide a favorable condition for stable anaerobic digestion as seen from preliminary results.

1. INTRODUCTION

Anaerobic Digestion (AD) is the process of decomposition of organic matter by a microbial consortium in an absence of oxygen (Sawatdeenarunat et. al., 2015; Lohani et.al., 2018).The degradation occurs in four stages, namely, Hydrolysis, Acidogenesis, Acetogenesis, and Methanogenesis. The anaerobes breakdown organic compounds to biogas that mainly consists of methane and carbon dioxide. Biogas is generally composed of 48-65% methane, 36-41% carbon dioxide, up to 17% nitrogen, <1% oxygen, 32-169 ppm hydrogen sulphide, and traces of other gases (Gould et.al., 2015). For the successful operation of a biogas plant, it is important to understand the anaerobic digestion (AD) process, the favorable conditions in which the anaerobes thrive, and the conditions that should be maintained. A key point for the successful description of a bioprocess is the appropriate influent characterization (Huete et. al., 2006; Lohani et. al., 2016). The control of the AD process is directed by the microbial activity itself (Lohani et. al., 2018).

AD of Food waste (FW) is a complex process that should simultaneously digest all the organic substrates (e.g. carbohydrate and protein) in a single-stage system (Zhang et. al., 2014). The key parameters that affect AD process are temperature, volatile fatty acid (VFA), pH, ammonia, nutrients, and trace elements (Xia et.al., 2016). Proper microbial growth requires a good nutrient and trace element balance, along with a stable environment. It is therefore extremely important to maintain the appropriate range of the key parameters for the long-term operation of AD (Xu & Li, 2017; Zhang et.al., 2014). FW contains 70-80% of the water which makes it highly biodegradable (Zamanzadeh et.al., 2017). Given its high biodegradability, the management of FW through landfilling, incineration, or composting impacts the environment negatively. Alternatively, AD of FW can be the most promising cost-effective technology for renewable energy production and waste management of this energy-rich material (Morales-polo et.al., 2018; Posmanik et al., 2017).

Previous studies in the AD of FW have already established process principles, bioenergy, and waste management potential through this process (Lin et al., 2013; Zhang et al., 2014; Zhang et.al., 2013). However, adopting the AD for FW are found to pose several technical challenges such

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as accumulation of volatile fatty acids, process instability, low buffer capacity, foaming, among others. Hence, it is important to identify the current challenges in the AD of food waste (Banks et.al., 2012; Zhang et al., 2013).

FW composition and its physicochemical properties vary depending on the country, food consumption pattern, and the cultural and economic aspects (Lin et al., 2013). AD of FW conducted at an organic loading rate of 4.0 g VS/(L.d) and 2.0gVS/(L.d) resulted in high VFAs concentration and low pH in the digestion chamber leading to low biogas production rate (El-Mashad et.al., 2008). In the same experiment, food waste was co-digested with dairy manure which indicated a significant increase in biogas production rate, compared to mono-digestion of food waste. In a study by Zhang et. al. (2014), the AD of FW was done for stabilizing the process in a controlled mesophilic condition by adding different quantities of lime mud, and cultured sewage sludge as inoculum. The experiment showed less fatty acid accumulation, increased production, and also a more stable process compared to the mono-digestion of FW. A review of all the scientific articles on behavior of food waste during the anaerobic digestion process from the year 2013 to 2015 was conducted, and. the study revealed that pretreated (thermally and chemically treated) food waste resulted in higher methane yield than the untreated food waste, and more than 18% of the research was performed by pretreating food waste (Komilis et al., 2017). In a study performed by Capson-Tojo et al. (2017), dry mono digestion of FW with the varying substrate to inoculum ratio (0.25, 0.5 and 1) and total solid content (20% and 30%), lower methane yield and longer lag period was observed at the higher substrate to inoculum ratio. Simultaneously, in the same experiment, co-digestion of the FW and cardboard paper was performed with the same parameters, which showed less acid accumulation and the small difference in yield obtained. Upon thorough research, the same authors concluded that the substrate and inoculum ratio was a major parameter in AD of food waste, and a lower substrate to the inoculum ratio resulted in inefficient methane production. At higher substrate to inoculum ratio, lower degradation of the substrate was observed resulting in hydrogen production (Capson-Tojo et al., 2017). Liu et. al. (2009) studied the effect of feed to inoculum ratio by performing AD of FW in a 1 L batch reactor under controlled thermophilic and mesophilic conditions. The result suggested an inverse relation of feed to inoculum ratio and biogas yield. The greater yield was observed in the thermophilic condition under less feed to the inoculum ratio; the yield under mesophilic condition was quite lower than that obtained at thermophilic temperature. Previously carried out studies have shown that the alkalinity addition by varying inoculum and substrate ratio and lowering OLR could be the solution for successful anaerobic digestion of food waste in a controlled environment (Li et al., 2018; Liu et al., 2009). In contradiction, the trials of AD of food waste in an ambient and uncontrolled condition were merely seen in the literature. This study aims to fill the gap between controlled laboratory and real field situations so that even the laboratory results could predict the real situation in the field.

In real situation, for the household biogas plant users in Nepal and in other developing countries, it is not viable to maintain controlled temperature as well as to undergo pretreatment of the feeding material and use cultured inoculum for the plant. In Nepal, the widely used household biogas plant, such as modified GGC 2047 and few urban household plants are not subjected to cultured inoculum start-up and are operated at ambient condition. Therefore, the implications derived from the experiments conducted under the temperature-controlled condition and with cultured inoculum for biogas production are not applicable for the majority of the Asian and African households. Such results rather mislead the estimation of biogas production potential in those settings. Hence, this study has been devised to assess the performance of anaerobic digestion of food waste in both batch and semi-continuous process with the addition of a variable percentage of uncultured inoculum (digestate obtained from operating household biogas plants) in an environment that exactly replicates the actual environment where the urban household digesters are being operated. This study also aims to reveal the challenges in the AD process of food waste when subjected to an uncontrolled environment and proposes ways to overcome those challenges.

2. MATERIALS AND METHODS

2.1 Feed Materials

Food waste was obtained from the Kathmandu University (KU) canteen located in Dhulikhel, Nepal. Wastes such as meat bones, lemon, and pickles were hand picked and thrown. Food waste was first ground with a food processor before using it for feed. The collected sample of food waste primarily consisted of cooked and uncooked vegetables along with cooked rice as depicted in Figure 1.

Digestate obtained from a functional household biogas plant treating cattle manure was used as inoculum for all the reactors. For each experiment, the inoculum was used in different proportions, measured by percent-weight of the substrate used such as 15%, 20%, 50%, 100%, and 200%.

2.2 Design and Operation of Digesters

The batch digestion tests were carried out in 500 ml borosilicate bottles and were made airtight using rubber

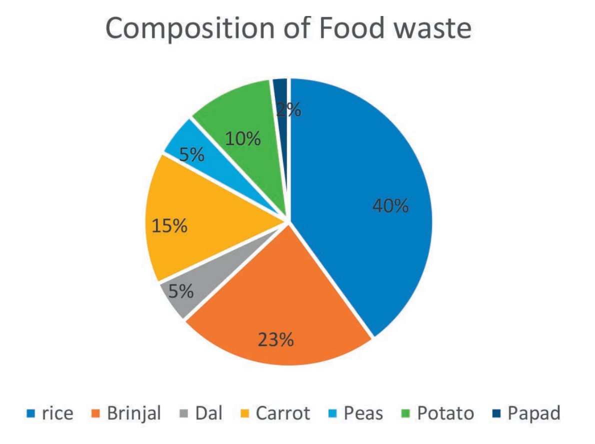


FIGURE 1: Composition of Food waste obtained from KU canteen.

FIGURE 2: Picture of a batch reactors.

FIGURE 3: Picture of a semi-continuous reactors.

corks fitted with infusion sets (IV sets) as shown in Figure 2. The bottle was then connected to NaOH scrubber followed by water displacement setup. All the reactor bottles were kept at ambient conditions for the trials.

The semi-continuous digesters used were 5 L bottles which were used for the semi-continuous feeding and withdrawal of the digestate as shown in Figure 3. A rubber cork fitted with ½ inch PVC pipe was attached to a plastic funnel for the inlet of the substrate. The infusion set was attached to the cork to work as gas outlet for the digester. Similar to the batch setup, all the components were sealed, and digester was made airtight.

The anaerobic digestion of the food waste in batch reactors was carried out in two different trials: (1) FW was independently digested as mono feed and with an inoculum of 20% weight of the substrate (2) FW with an inoculum of 50%, 100% and 200% weight of the substrate in different reactors. 50 g of the substrate with 10% TS was added into each digester. Each batch test was run in duplicate and the test was carried out for 30 days in ambient temperature. The volume of the gas produced by each reactor was measured daily and the measurements of gas composition were analyzed 5 to 8 times during each experiment.

Two sets of semi-continuous digesters were fed with food waste to test for biogas production. Set 1 reactors were fed with food waste of 10% TS and set 2 reactors were fed with food waste of 6% TS. In both sets, 15% of inoculum was added to the weight of the substrate being used. For these reactors, two different OLR 1gVS/L.d and 1.8 gVS/Ld and hydraulic retention time of 50 days were used.

2.3 Analytical methods

Before experimentation, the substrates, and inoculum were characterized. The TS and VS content were determined using standard methods of the American Public Health Association(APHA, 2005). pH was measured using Exotech SOL 100 pH meter. The biogas composition was measured with Sewerin Multitec-545 gas analyzer. The ambient temperature was recorded every half an hour to attribute to temperature fluctuation using temperature logger. The total organic carbon was measured using standard provided by the American Society of Agronomy and Soil Science and the total organic nitrogen was calculated using APHA 4500-Norg Macro- Kjeldahl method. Based on the readings of these instruments the C/N ratio of the substrate was calculated.

3. RESULT AND DISCUSSION

3.1 Physical and Chemical Properties of the substrate

Table 1 shows the total solid, volatile solids, pH, and total organic carbon to nitrogen (C: N) ratio of food waste, inoculum and feeding sample of different Substrate (S) to Inoculum (I) (S: I) ratio used for the different trials of this study. pH of all samples seems very acidic except S: I ratio 1:1 and 1:2, which indicates that a high inoculum ratio is required to increase the pH of the feed.

3.2 Experimental trials in batch reactors

Figure 4 shows the daily gas production and average temperature during the anaerobic digestion of food waste with 20% of inoculum added to it. The experiment was conducted from the second week of May to April 2019 and the

TABLE 1: Properties and characteristics of food waste and inoculum.

Characteristics	Food waste	Inoculum	Substrate(S) to inoculum (I) (S: I ratio) of the Sample					
			15% (20:3)		20%(5:1)	50%(2:1)	100%(1:1)	200%(1:2)
TS%	20.8	10.1	6%	10%	10%	10%	10%	10%
VS% (of TS%)	89.2	60	84.9%	86.4%	83.5%	74.8%	81.9%	80.5%
pH	4.4	7.1	5.2	5.2	5.8	6.13	6.30	6.57
C: N Ratio	22.4	19.6	22.2	22.0	21.9	21.5	20.2	19.8

figure presents the average ambient temperature range during the operation of batch reactors to be between 20-26°C. A small amount of gas production was noted during the initial week that could be due to the inoculum. However, after the first week of operation, the production of gas stopped completely despite that the reactor was subjected to a very small fluctuation in temperature. Figure 5 shows the cumulative gas production during the process along with the average temperature trend.

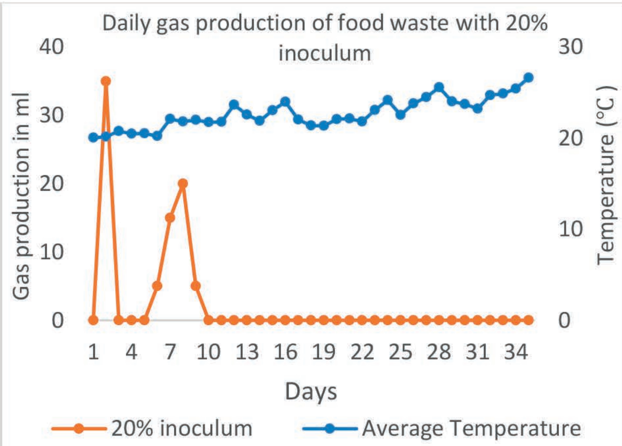
The pH of the feed reduced from 5.78 to 3.32 after two weeks of the digestion process. The abrupt halt of the gas production might be due to acidification of the substrate and absence of the methanogenesis process during the digestion process.

Figure 6 shows the average temperature and gas production during anaerobic digestion of food waste added with 50%, 100% and 200% of inoculum to the weight of the substrate. This shows the production of gas in the initial few days; however, the production is drastically reduced thereafter. The highest value of daily gas production was obtained at 4-6 days of operation in all the reactors. The average temperature during the operation of the AD process was between 23-25°C. The experiment was conducted from the last week of July to the first week of August 2019.

Figure 7 shows the cumulative gas production in this trail. It depicts that after the first week of operation, the gas production in all the reactors differed. A total of four gas composition measurements were taken during the experimental process which showed a negligible amount of methane production (CH_4 range of 0.1-0.7%). During the first week of operation, the methane composition range of 0.1-0.3% was observed. After observing the process for 19 days, the reactors were discarded taking into consideration the result obtained from the gas analyzer (0.7% CH_4 on the 19th day). The pH of the mixture before and after AD in Table 2 shows souring of the feedstock inside the digester. This might be due to excessive accumulation of volatile fatty acid which resulted in a drastic reduction of pH of the substrate which might have led to halting the methanogenesis process during the AD process.

3.3 Experimental trial in semi-continuous reactors

Figure 8 illustrates the daily average temperature and gas production plot during anaerobic digestion of food waste at different total solid content in a semi-continuous reactor, observed at HRT of 50 days. The average temper-



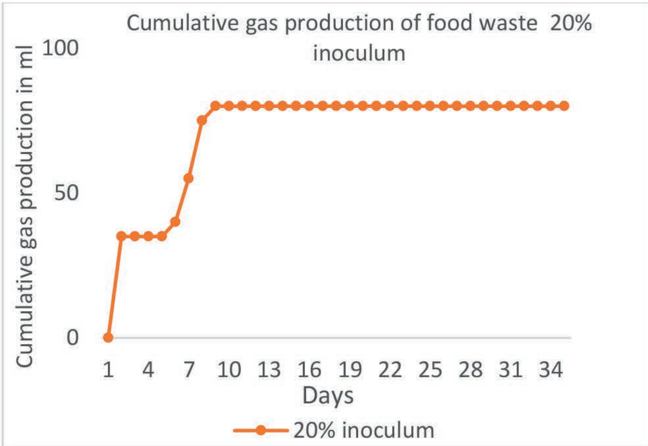
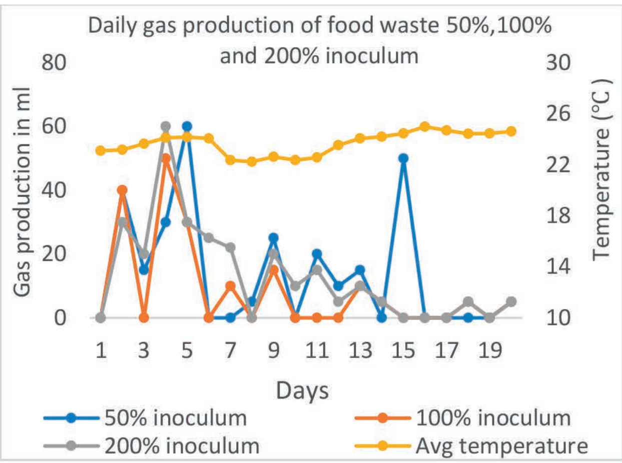


FIGURE 4 AND 5: Picture of a semi-continuous reactors.



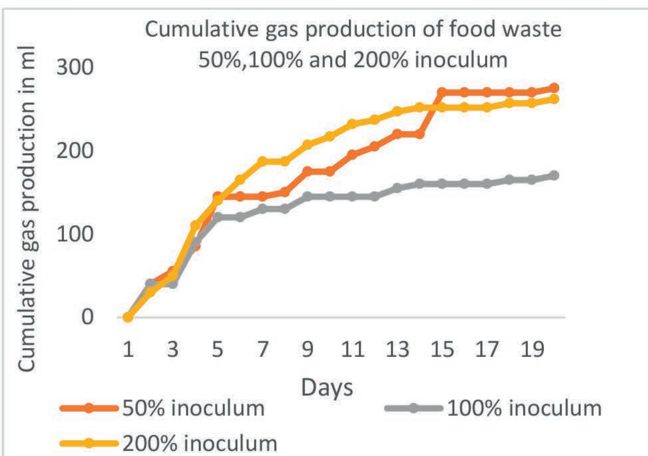


FIGURE 6 AND 7: Daily and cumulative gas production in trial 2 batch reactors.

TABLE 2: pH of the mixture before and after AD.

S. N	Reactor Bottle	pH before AD	pH after AD
1	200%	6.57	4.41
2	100%	6.30	3.65
3	50%	6.13	2.89

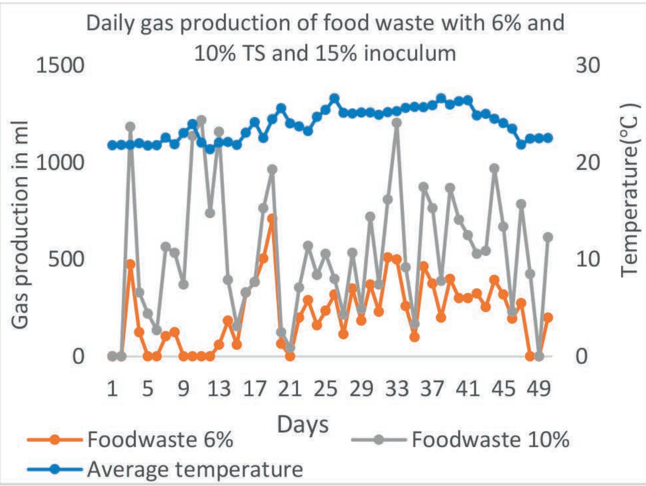
ature during the whole process was between 20-26°C. The AD of food waste was continued from the second week of May to July. Figure 9 shows the gas production in the reactor having food waste with 10%. TS content seems to be higher than the reactor having food waste with 6% TS content. Though the feeding of the reactors was run at relatively low pH, a considerable amount of gas production was observed in both the reactors. The maximum gas yield of 135.24 L/kgVS was obtained from the AD of food waste with 10%TS.

The result from the gas analyzer, however, proved the presence of a very small amount of methane gas. The results from the gas analyzer (8 times during the whole experimental session) showed CH_4 range from 0.1 to 13%. This suggests the inability of the growth of methanogens during the anaerobic digestion process. The inhibition might be created due to the souring of the reactor resulting from the rapid conversion of easily digestible food waste to VFA at the early stage of the digestion process along with low initial pH of the feed. pH observed after the anaerobic digestion process was 3.4. The experiment trials conducted in both batch and semi-continuous reactors suggest that there was rapid acidification in the process, limiting the methanogenesis step due to sudden pH drop. The low initial pH of the substrate, uncontrolled temperature, and unmixed condition could be vital factor for the growth of the methanogens during the process.

A similar study was conducted in Beijing, China using canteen food waste for investigating characteristics of food waste containing different substrates to inoculum (S: I) ratio (0.5, 0.6, 0.7, 0.8, 1.0, and 1.2). In the experiment, it was observed that there was rapid acidification of the reactor as well as reduced biogas production with a higher S: I ratio. Lower the substrate to the inoculum ratio, the high biogas conversion ratio was observed with a shorter lag phase (Li et al., 2018). In another similar experiment, S: I ratio of 1:2,1:1 and 2:1 were used which showed good stability and higher methane productivity in ratio 1:2; however, higher ratio such as 2:1 indicated higher conversion rates of VFA resulting in acidic slurry inside the reactor (Zhang et al., 2019). These findings justify the result obtained in trial 1 of this experiment (Figures 4 and 5) where a high S: I ratio of 5:1 was used, which caused the stoppage of gas production after few days of operation.

In all the experiments conducted by other researchers as indicated in the discussion and Table 3, the trials were performed either incubated in shakers, maintaining mesophilic conditions or were performed in automatic methane potential system (Li et al., 2018; Zhang et al., 2019). Moreover, the inoculum used in most of the similar experiments was either control culture, seeded/inoculated, or pre-treated inoculum, which was observed to help in boosting the methanogenic activities inside reactors (Hobbs et al., 2018; Li et al., 2018; Rafieenia et al., 2018).

The trials in this study were conducted in an uncontrolled environment with uncultured inoculum and at ambient temperature. Therefore, even when varying the same substrate to the inoculum ratio as conducted in other experiments, a different result might have been obtained. The findings of this study is believed to fill the gaps in results achieved between controlled laboratory conditions and the real field condition so that the laboratory results could predict the real situation in the field to help design the program and policies. An example from Nepal Sahari Gharelu Biogas plant introduced in the year 2012/13 by the government of Nepal as a pilot urban domestic biogas project in



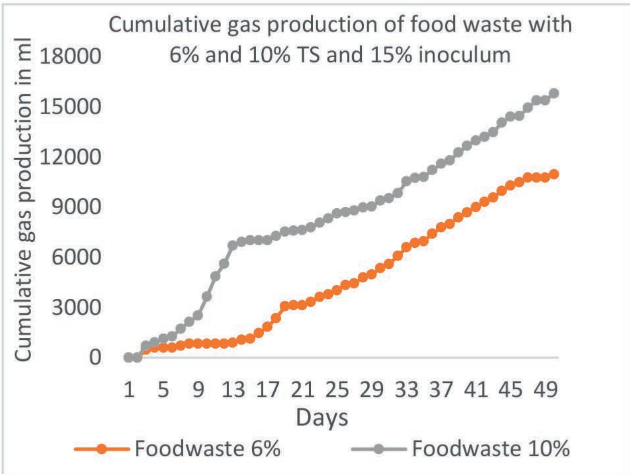


FIGURE 8 AND 9: Daily and Cumulative Gas production in trial 3 Semi-continuous reactors.

TABLE 3: Comparison of the present study with other studies.

Substrates(S)	Inoculum(I)	S: I ratio	Conditions	Remarks	References
Food waste	Poultry litter Goat manure Cow dung Piggery dung Rhinoceros dung	1.5:1 2:1 2:1 1.5:1 1.5:1	Temperature maintained at 30°C, a stirring provision added	Shorter start-up time and higher methane production from food waste inoculated with cow dung at ratio 2:1	Dhamodharan et. al., 2015
Municipal solid waste	Sludge from mesophilic continuously stirred reactor tank treating wastewater	200:3,100:3,50:3,25:3,4:1,1:1, 1:2,1:4	Reactors incubated at temperature 35±20°C	Accumulation of maximum dissolved organic carbon (DOC) at ratio 25:3, maximum methane production and lower DOC at ratio 1:2 and above	Boulanger et al., 2012
Food waste	Inoculum from a mesophilic anaerobic digester treating sewage sludge at the wastewater treatment plant	1:2, 1:3, 1:4	food waste of different particle size (1mm, 2mm, 5mm), maintained mesophilic temperature of 37°C using the water bath	S: I ratio of 1:3 and 1:4 stabilized reactors for smaller particle size (1mm and 2mm) Reduced volatile fatty acid accumulation and 91% reduction of lag time at ratio 1:3 Highest methane yield observed at ratio 1:2	Okoro-Shekwaga et. al., 2020
Food waste	Mesophilic anaerobic sewage sludge from sludge treatment plant	33:100 ,1:2,1:1,2:1,4:1	TS% varied- 4.4% and 10.5%, Temperature maintained at 37±1°C, agitated using shaker at 60 rpm	High organic fraction increased risk of acidification but also contributed to high methane, S: I ratio< 0.33 resulted in higher methane yield	Kawai et al., 2014
Food waste	Digestate from a functional household biogas plant treating cattle manure	20:3, 5:1, 2:1, 1:1,1:2	Ambient room temperature of 20-26°C TS% of 6%,10%	Mono digestion of food waste resulted in unstable process, greater gas production in 10% TS, however, negligible methane composition, varying S: I ratio was not enough for the stable AD process	This study

Kathmandu valley is worth evaluating. A total of 23 plants were disseminated as a pilot project to manage organic kitchen waste in urban and peri-urban regions. The size of the biogas plant was 1m^3 and was a floating drum type which is essentially an Appropriate Rural Technology Institute (ARTI) model biogas plant developed in India (AEPC, 2013). Unfortunately, this model could not gain popularity and was not able to gain acceptability in urban settings of Nepal. The reason for this plant failure was due to its performance; the plant was reported to be unable to function well and all users discarded it in Kathmandu valley (Lohani & Fulford, 2017).

3.4 Preliminary results of co-digestion of food waste

Anaerobic co-digestion of food waste with sewage sludge and chicken litter is in progress in the author's laboratory at Dhulikhel, Nepal. The experiment is being performed in a 5L semi-continuous reactor at ambient temperature. A semi-continuous digester is fed with sewage sludge, poultry litter, and food waste to test for the biogas production. The reactor is fed with sewage sludge, poultry litter, and food waste in a ratio of 2:1:1 at 8% TS. For the reactor, OLR of 0.70gVS/L.d and hydraulic retention time of 50 days are used. The trial is duplicated for the accuracy of the result.

Figure 10 renders high gas production during the co-digestion of food waste when compared to the mono digestion result in figure 8. Although high fluctuation of temperature is seen, the daily average production throughout the process is 800ml of gas. That is to infer, more stability is seen during the co-digestion of food waste with other substrates. Moreover, the gas composition observed up to the date shows the highest methane content of nearly 66%.

4. CONCLUSIONS

In this study, biogas generation from food waste under the ambient condition was performed with different percentages of inoculum, and total solid content in both batches and semi-continuous reactors. In all the experiments, common type of problem was observed, that is, the inability of growth of methanogens to initiate the methanogenic process during the AD process. Hence, no methane was produced during the AD process, though, CO_2 and other gases were observed. This suggests that food waste mixed with up to 200% by weight with inoculum at the batch process and food waste mixed with 15% by weight with inoculum at semi-continuous process does not help significantly in anaerobic digestion process at the ambient condition of temperature range from 20 to 26°C. The instability was due to a large pH drop and the souring of the reactor. It can be concluded that mono-digestion of food

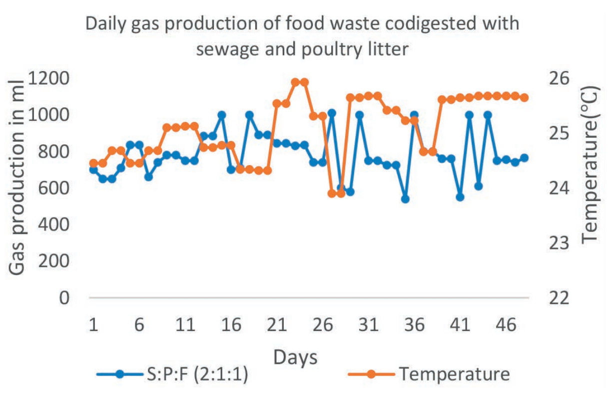


FIGURE 10: Daily gas production with co-digestion of food waste, sewage sludge, and poultry.

waste and food waste with the addition of a few percentages of cow manure as inoculum was not enough for the stable biogas production. Co-digestion of food waste with a different substrate such as cow manure and urine, sewage sludge, poultry droppings could help balance pH and C: N ratio and nutrients in a favorable condition for the stable AD process and biogas production.

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TWO-STAGE ALKALINE AND ACID PRETREATMENT APPLIED TO SUGARCANE BAGASSE TO ENRICH THE CELLULOSIC FRACTION AND IMPROVE ENZYMATIC DIGESTIBILITY

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ABSTRACT
Sugarcane bagasse (SB) is made up of cellulose (32-43%), hemicellulose (19-34%) and lignin (14-30%). Due to high recalcitrant nature of SB, pretreatment is required to deconstruct its structure and enrich the cellulosic fraction. A two-stage NaOH and maleic acid pretreatment was applied to SB to enrich its cellulosic fraction. SB used in the present study is composed of cellulose (40.4 wt%), hemicellulose (20.9 wt%), lignin (22.5 wt%) and ash (4.0 wt%). After one-stage NaOH pretreatment, its cellulosic fraction increased to 61.8 wt% and later increased to 80.1 wt% after the second-stage acid pretreatment. Lignin fraction decreased to 3.0 wt% after one-stage NaOH pretreatment and remained unaffected after the acid pretreatment step. Hemicellulose fraction decreased substantially after the second-stage pretreatment with maleic acid. Pretreated SB displayed high crystallinity index and improved enzymatic digestibility. Hydrolysates of pretreated SB contained very low amount of xylose and subsequent fermentation by *Saccharomyces cerevisiae* -IQAr/45-1 resulted to ethanol level of 8.94 g/L. Maximal ethanol yield of 0.49 g/g (95.8% of theoretical yield) and productivity of 0.28 g/L/h was attained. At the same time, biomass yield and productivity of 0.47 g/g and 0.27 g/L/h respectively were obtained. Two-stage NaOH and maleic acid pretreatment led to ~ two-fold increase in cellulosic fraction and enhanced the enzymatic digestibility of SB up to 70.4%. The resulted enzymatic hydrolysate was efficiently utilized by *S. cerevisiae* -IQAr/45-1 to produce high yield of ethanol. Thus, optimization of enzymatic hydrolysis at low enzyme loading is expected to further improve the process and reduce cost.

1. INTRODUCTION

Sugarcane bagasse (SB) is an important feedstock for second generation bioethanol production due to its large abundance, non-competitiveness with food or feed requirement, easy transportation and rich in accessible carbohydrates (Singh et al. 2015; Chandel et al., 2012). The composition and productivity of bagasse is dependent on sugarcane variety, climate, location, plant age, and soil types (Zhao and Li, 2015). SB is made up of cellulose (32-43%), hemicellulose (19-34%) and lignin (14-30%) (Brienzo et al., 2016; Timung et al., 2016).

Cellulose exists as D-glucose subunits, linked by β-1,4 glycosidic bonds (Jönsson et al., 2016) and its microfibrils are chemically bound to lignin and hemicellulose (Zhang et al., 2016). The cellulose in a plant consists of parts with a crystalline (organized) structure, and parts with not well-organized, amorphous structure (Rongpipi et al., 2019).

Hemicelluloses, unlike cellulose, are polymers constituted of heterogeneous, branched polysaccharide composed of C_5 sugars (xylose, arabinose) and C_6 sugars (mannose, glucose and galactose), which serve as a connection between lignin and the cellulose fibers (Lee et al., 2014). Xylan is a dominant component of hemicellulose from hardwood and agricultural plants, such as grasses and straw while for softwood it is glucomannan (Álvarez et al., 2016). Lignin is a complex, cross-linked, three-dimensional polymers of phenolic monomers having both aliphatic and aromatic constituents (Karunarathna and Smith, 2020). Lignin is a constituent that is known to inhibit enzymatic saccharification and fermentative microorganisms (Kucharska et al., 2020). Nevertheless, the matrix structure of lignocellulosic biomass prevents the enzymatic saccharification and subsequently sugars fermentation to bioeth-

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anol (Sun et al., 2015). Thus, pretreatment is required to deconstruct the intact structure by removal of hemicellulose/lignin and improve the enzyme accessibility to cellulose (Isaac et al., 2018; Acharjee et al., 2017). On the other hand, pretreatment can only be considered effective if it minimizes carbohydrate degradation and the production of enzyme inhibitors as well as toxic products for fermenting microorganisms (Thite et al., 2019; Laluce et al., 2019a).

Research activities in the last few years have been directed towards improving sugar yield from lignocellulosic biomass through the application of physical, chemical, physicochemical and biological pretreatments (Mohapatra et al., 2017). Chemical pretreatments with dilute inorganic acids and alkali have been applied to hydrolyze lignocellulosic biomass (Laluce et al., 2019b; Matsakas et al., 2018; Silveira et al., 2015). On the other hand, dicarboxylic organic acids such as maleic acid are considered alternative to inorganic acids (e.g. sulfuric acid) because they have lower hazardous properties and lower inhibitory by-products generation as well as the ability to hydrolyze β-(1,4) glycosidic bonds (Girolamo et al., 2016; Jung et al., 2014). Maleic acid pretreatment of oil palm trunk was reported to effectively remove hemicellulose and part of acid soluble lignin (Qiao et. at. 2019).

The combinations of delignification process with dilute acid pretreatment have been applied to remove lignin and solubilize most of the hemicellulose sugars from SB (Zhang et al., 2018; Chandel et al., 2014). However, pretreatment combining NaOH pretreatment (first-stage) and maleic acid pretreatment (second-stage) is scarcely reported in the literature.

Thus, a two-stage pretreatment of SB was proposed in this study to degrade lignin using 3.0% NaOH at 121°C for 60 min in the first-stage and then to solubilize hemicellulose using 0.3% maleic pretreatment at 121°C for 60 min in the second-stage. Subsequently, the one-stage and two-stage pretreated SB was characterized by chemical constituent analysis and X-ray diffraction (XRD) was used to investigate the structural changes that occurred during pretreatments. Furthermore, enzymatic saccharification of pretreated SB and ethanol fermentation performance by *Saccharomyces cerevisiae* (IQAr/45-1) were evaluated.

2. EXPERIMENTAL

2.1 Sample preparation and processing

Sugarcane bagasse was collected from a local sugar mill plant, Santa Cruz (member of São Martinho group) located in Américo Brasiliense, São Paulo, Brazil and transported at low temperature. Bagasse was stored in a laboratory freezer at -20°C. Frozen samples were defrosted and dried at 60°C in a laboratory incubator until a constant weight was obtained (<10 % wt, moisture). Dry bagasse samples were stored in transparent plastic bags at room temperature before use.

Milling the dry sugarcane bagasse (SB) by a physical method initiated the degradation of lignocellulose and conversion of the biomass fibers into particles, thereby increasing biomass surface area (Palmowski and Müller, 2000). For this, sugarcane bagasse was ground for 15 min/cycle in Marconi Ball Mill with Closed Chamber (model MA350) to obtain particles of ≤ 0.5 mm after passing through a set of superposed sieves of different meshes (32, 35, and 150 mesh).

Subsequently, SB was refluxed in a Soxhlet extraction apparatus containing a mixture of toluene:ethanol (2:1, v/v) (Sun et al., 2004). The sample was refluxed in water for 30 min to remove the remaining solvents before being dried in a Biochemical Oxygen Demand (BOD) incubator, as recommended in the literature (Binod et al., 2012) at 60°C to constant weight before being stored in a desiccator at room temperature until use.

2.2 Pretreatment with alkaline solution (one-stage)

Dry sugarcane bagasse (particle size ≤ 0.5 mm) was treated with different concentrations of NaOH solution (0.5%-3%, w/v) in 125 ml Erlenmeyer flasks at solid/liquid ratio of 1:20. Flasks were gently swirled to enable the solid to become completely soaked in the solution before been transferred into the autoclave and heated for 60 minutes at 121°C. After heating, samples were removed from the autoclave and allowed to cool before filtration under vacuum using Whatman No. 1 filter papers.

Filtrate was collected and used to assay for glucose and xylose while the insoluble material (residues) on the filter paper was washed several times with deionized water to neutral pH and dried at 60°C to a constant weight. Dry residues were used for compositional analysis and dilute acid pretreatment step.

2.3 Selection of optimum condition for acid pretreatment step

Dilute acid solutions of maleic acid (MA) in the range of 0.1%-0.5% (w/v) were used to hydrolyze sugarcane bagasse at 121°C (autoclave) for 60 minutes with solid/liquid ratio of 1:20. Sample was removed after autoclaving and allowed to cool on ice bath and filtered under vacuum using Whatman No. 1 filter papers. Filtrate was collected and used to measure the soluble reducing sugars (SRS) using 3,5-Dinitrosalicylic acid (DNS) reagent. The residue on the filter paper was washed with deionized water to neutral pH and dried in a hot air oven at 60°C till the weight remained constant and measured. The lowest concentration of acid that yielded the highest amount of SRS and highest biomass loss was selected for the second step pretreatment.

The amount of released SRS in pretreatment filtrate from SCB was measured using D-glucose as standard and expressed as % released SRS as mentioned below.

$$\%\,released\ SRS\ = \qquad (1)$$
$$= \frac{reducing\ sugar\ released\ in\ filtrate\ during\ pretreatment\ (mg)}{biomass\ used\ for\ pretreatment\ (mg)} \times 100$$

Decrease in weight of dry biomass was calculated as mentioned below and expressed as % weight loss in biomass.

$$\%\,weight\ loss\ in\ biomass = \frac{biomass\ obtained\ after\ drying\ (mg)}{biomass\ used\ for\ pretreatment\ (mg)} \times 100 \qquad (2)$$

2.4 Two-stage alkaline and acid pretreatments

Dry residues which originated from the one-stage pretreatment with different concentrations of NaOH were suspended in dilute solution of maleic acid (0.3%, w/v) in 125 ml Erlenmeyer flask at solid/liquid ratio of 1:20. Flasks were placed in autoclave and heated for 60 minutes at 121°C. Thereafter, flasks were removed and allowed to cool before filtration. Filtrate was collected and used to assay for glucose and xylose, while the residues on the filter paper was washed several times with deionized water to neutral pH before been dried in BOD at 60°C to a constant weight and used for chemical composition analysis.

2.5 Determination of the chemical composition of untreated SB and pretreated fractions

The chemical components of untreated and pretreated SB were analyzed using the method described by National Renewable Energy Laboratory (Sluiter et al., 2008). 300 mg dry residues resulting from pretreatment or untreated SB was weighed into a pressure glass tube and 3.0 ml of 72% sulfuric acid was added, the mixture was stirred with glass rod and tube was place in water bath at 30°C for 1 h. The mixture was stirred while in water bath at every 10 min. After 1 h of hydrolysis, tube was removed from water bath and 84 ml of deionized water was added to bring the acid concentration to 4%. The tube was covered and inverted several times to allow sample to mix and autoclaved at 121°C for 1 h.

After autoclaving, the mixture was cooled down to room temperature and separated into solid and liquid fractions by vacuum filtration using Whatman No.1 filter papers. Solid fraction was then dried at 105°C in order to measure acid insoluble residue (AIR). Acid insoluble residue was burnt in muffle furnace at 600°C for about 6 h to measure acid insoluble lignin (AIL) content as the difference between AIR and ash. Before neutralization, the absorbance of liquid fraction at 240 nm was measured by UV-visible spectrophotometer (Cirrus 80 ST, Femto, São Paulo, Brazil) to determine acid soluble lignin (ASL). Liquid fraction was neutralized with calcium carbonate and the resulting filtrate was used to assay for glucose and xylose.

Glucose was assayed using a commercial enzymatic kit (GOD-PAP, Laborlab) and xylose by the phloroglucinol method using xylose as standard (Ebert et al., 1979). Total furans were estimated by a spectrophotometric method based on the difference in absorbance at 284 and 320 nm (Martinez et al., 2000) using a UV/Vis/NIR-spectrometer with a 3D WB Detector Module (Perkin Elmer, Inc., Shelton, CT USA). A standard curve was obtained for each assay by linear regression using the software OriginPro 8 from OriginLab Corporation.

2.6 X-Ray Diffraction (XRD)

XRD was used as a tool to investigate the structural changes that occurred during pretreatments of sugarcane bagasse. The crystalline structure of biomass is mainly due to the strong hydrogen bonding of cellulose chains and Vander Waals forces of glucose molecules in the cellulose (Naik et. al., 2010; Sasmal et. al., 2012).

The crystallinity of the cellulose fiber was evaluated by X-Ray Diffraction (Siemens D5000 diffractometer, Munich, Germany). Copper Kαradiation, 30.0 kV of voltage and 15 mA of electric current, and a rate of 2.0 degrees per minute for a 2θ continuous scan from 5.0 to 50.0 degrees were applied. This analysis allowed the detection of the amorphous part of the lignocellulosic biomass, as well as the modification of the crystalline structure of the cellulose.

The crystallinity index (CI) was obtained from the ratio of the maximum peak intensity 002 (I_{002}, 2θ = 22.5) and minimal depression (I_{am} 2θ = 18.5) between peaks 001 and 002 (Segal et al., 1959; Rodrigues et al., 2007) as mentioned below.

$$CI(\%) = \frac{I_{002} - I_{am}}{I_{002}} \text{x}100 \tag{3}$$

where I_{002} is the maximum intensity of the 002 peak and I_{am} the minimal depression of the amorphous structure.

2.7 Enzymatic hydrolysis

Dry pretreated SB and untreated SB were each soaked in 50 mM sodium citrate buffer (pH 4.8) at solid loading of 5% (w/v) in 125 ml Erlenmeyer flask. Flask was placed in incubator (Tecnal TE-391, Piracicaba, SP, Brazil) and incubated for 2 h at 50°C with shaking speed of 150 rpm. Thereafter, the soaked sample was supplemented with 8.7 FPU/g dried sample of Cellulase from *Trichoderma reesei*, (Sigma Aldrich) and 5.6 IU/g dry sample of β-glucosidase (Sigma Aldrich). A dose of 0.005% sodium azide was introduced to avoid any microbial contamination and 1.0% (v/v) Tween 80 was added to facilitate the enzymatic action. Enzymatic hydrolysis was performed at 50°C for 72 h with shaking at 150 rpm. Samples were withdrawn at 6 h, 12 h, 24 h, 48 h and 72 h intervals and enzymes were inactivated by boiling at 100°C for 10 min after which samples were cooled on ice before subsequently analyzed for glucose released using a commercial enzymatic kit (GOD-PAP, Laborlab).

Saccharification (%) was calculated as mentioned below:

$$\% \text{ saccharification} = \tag{4}$$

$$= \frac{\text{reducing sugar released by enzymatic hydrolysis (mg)}}{\text{initial solid biomass used for hydrolysis (mg)}} \times 100$$

2.8 Fermentation

The *Saccharomyces cerevisiae* -IQAr/45-1 is a thermotolerant ethanologenic yeast strain obtained from the hybridization between parental strains of *S. cerevisiae* and three Brazilian industrial strains (PE-2, CAT-1, SA-1) during fermentation of non-sterilized molasses (Laluce et. al. 2013). This strain can only ferment hexose sugars and it was maintained in medium containing (g/L): glucose, 30.0; yeast extract, 3.0; peptone, 5.0; agar, 20.0 at pH 6.0 ± 0.2 and temperature 30°C. Starter culture was developed by growing the cells at 30°C for 24 h in a culture medium containing (g/L): glucose, 30.0; yeasts extract, 3.0; peptone, 5.0; pH 6.0 ± 0.2.

The fermentation of enzymatic hydrolysates of two-stage NaOH+MA pretreated SB (17.3 g/L glucose, pH 6.0 ± 0.2) was carried out in 125 ml Erlenmeyer flask with a working volume of 50 ml supplemented with 3 g/L of yeast extract and 5 g/L of peptone. It was inoculated with *S. cerevisiae* (10.0% v/v) at optical density (OD_{600}) of 0.6. Sample was incubated at 30°C for 30 h with shaking at 150 rpm.

Samples were centrifuged at 10,000 g for 15 min at 4°C and the cell free supernatant was used to determine the ethanol and residual sugar concentration.

Cell concentrations were measured at optical density of 600 nm and related to the cell dry weight through a calibration curve. Ethanol was estimated using acidified potassium dichromate solution as described by Caputi et al., (1968).

Fermentation parameters were calculated as mentioned below:

$$Y_{P/S} = \frac{E_f - E_i}{(S_i - S_f)} \quad (5)$$

$Y_{P/S}$ is the ethanol yield, E_i and E_f are the ethanol concentration at the beginning of the fermentation and the end of the fermentation (g/L) respectively; while S_i and S_f are the total sugar concentration at the beginning of the fermentation and the end of the fermentation (g/L), respectively.

$$Q_{P_P} = \frac{E_f - E_i}{t_f - t_i} \quad (6)$$

Q_{P_p} is the volumetric ethanol productivity (g/L/h), E_i and E_f are the ethanol concentration at the beginning and end of fermentation (g/L), respectively; while t_i and t_f are the fermentation time at the beginning of the fermentation and the end of the fermentation (g/L), respectively.

$$Y_{X/S} = \frac{X_f - X_i}{S_i - S_f} \quad (7)$$

$Y_{X/S}$ is the biomass yield, X_i and X_f are the biomass concentration at the beginning of the fermentation and the end of the fermentation (g/L), respectively; while S_i and S_f are the total sugar concentration at the beginning of the fermentation and the end of the fermentation (g/L), respectively.

$$Q_{P_X} = \frac{X_f - X_i}{t_f - t_I} \quad (8)$$

Q_{P_x} is the biomass productivity (g/L/h), X_i and X_f are the biomass concentration at the beginning and end of fermentation (g/L), respectively; while t_0 and t_r are the fermentation time at the beginning of the fermentation and the end of the fermentation (g/L), respectively.

$$Fermentation\ efficiency\ (\%) = \frac{Actual\ Yield}{Theoretical\ Yield} \times 100 \quad (9)$$

where theoretical yield is equivalent to 0.511 g/g.

2.9 Data analysis

The graphs were created using the software OriginPro 8 from OriginLab Corporation. Each data was expressed as a mean of standard deviation (SD) of triplicate measurements.

3. RESULTS AND DISCUSSION

3.1 Pretreatments

A preliminary study to select the optimal condition for acid pretreated step is presented in Table 1. One-stage maleic acid pretreatment of SB resulted to loss of dry weight ranging from 10.6% to 23.4%, while the yield of SRS in the acid hydrolysate ranges from 8.5% to 10.6%. The loss of dry weight was mainly attributed to hemicellulose solubilization during acid pretreatment (Baruah et al., 2018, Li et al., 2016). Thus, the yield of SRS and loss of dry weight were directly correlated with acid concentration. The mass differences between loss of dry weight and yield of SRS could be connected to the presence of other extractive components in the SB, which were solubilized during acid pretreatment.

Table 2 shows the results of one-stage NaOH and two-stage NaOH and maleic acid pretreatments applied to SB. It was found that increases in NaOH concentration from 0.5% to 3.0% led to corresponding increases in cellulosic fractions of SB from 45.5 wt% to 61.8 wt%. The proportionality observed between NaOH concentration and cellulosic fraction was due to lignin removal at increasing NaOH concentration. According to the literature, alkaline pretreatment can facilitate dissociation of entire cell wall polymers by breaking hydrogen and covalent bonds thereby enabling effective lignin removal (Thite et al., 2019; Rezende et al., 2011).

The second-stage pretreatment with 0.3% maleic acid led to increase in cellulosic fractions from 46.7 to 80.1 wt%. This increase could be attributed to hemicellulose removal during the acid pretreatment step. Dilute acid pretreatments have been reported to cause hemicellulose solubilization and cellulose enrichment (Zhang et al., 2018; Rezende et al., 2011).

Interestingly, cellulosic fractions of SB resulting from two-stage NaOH and maleic acid pretreatments are connected to the cellulosic fractions from the first-stage NaOH pretreatment. This implies that the higher the cellulosic fraction after first-stage NaOH pretreatment, the higher the cellulosic fraction after the second-stage acid pretreatment. The correlations between first- and second-stage pretreatments were mainly due to hemicellulose solubilization during the second-stage acid pretreatment.

3.2 Chemical composition of untreated (raw) and pretreated sugarcane bagasse.

SB used in the present study consists of ~ 10% moisture and particle size of about 0.50 mm. Its chemical composition is made up of cellulose (40.4 wt%), hemicellulose (20.9 wt%), lignin (22.5 wt%) and ash (4.0 wt%) as shown in Table 3. These values are similar to those reported by other authors (Rabelo et al., 2009; Sporck et al., 2017). Furthermore, chemical composition and productivity of SB are mostly dependent on sugarcane variety, climate, location, plant age and soil types (Zhao and Li, 2015).

TABLE 1: Effect of maleic acid concentration on the release of soluble reducing sugars (%) in pretreatment filtrate and loss of dry weight (%) during the pretreatment of sugarcane bagasse in autoclave (121°C) for 60 min.

Maleic acid (%, w/v)	Temperature (°C)	Time (min)	Soluble reducing sugars (%)	Loss of dry weight (%)
0.1	121	60	8.5 ± 0.4	10.6 ± 0.4
0.2			10.0 ± 0.6	13.5 ± 0.6
0.3			15.6 ± 0.3	19.1 ± 0.5
0.4			16.9 ± 1.2	20.0 ± 0.3
0.5			19.2 ± 1.1	21.9 ± 0.4
0.6			20.8 ± 0.7	23.4 ± 0.5

TABLE 2: Cellulosic fractions resulting from first-stage NaOH pretreatment and second-stage pretreatment with 0.3% maleic acid in autoclave.

Pretreatment							
First-stage				Second-stage			
NaOH (%, w/v)	Temp (°C)	Time (min)	Cellulose (wt%)	Acid (w/v)	Temp (°C)	Time (min)	Cellulose (wt%)
0.0	121	60	42.7 ± 0.6	0.3% MA	121	60	46.7 ± 0.7
0.5			45.5 ± 0.6				56.5 ± 1.5
1.0			48.9 ± 1.0				63.0 ± 1.4
1.5			55.5 ± 1.0				68.7 ± 1.2
2.0			58.4 ± 0.3				73.8 ± 1.2
2.5			59.8 ± 0.8				78.7 ± 0.9
3.0			61.8 ± 1.3				80.1 ± 1.8

TABLE 3: Chemical composition of solid fractions of untreated and pretreated sugarcane bagasse and composition of pretreatment hydrolysates.

Pretreatment (60 min, 121°C)	Chemical composition of dry solid fraction				Constituents of pretreatment filtrate	
	Cellulose (wt%)	Hemicellulose (wt%)	Lignin (wt%)	Ash (wt%)	Xylose# (%, g/g)	Total furans (mg/L)
Untreated SB	40.4 ± 0.9	20.9 ± 0.7	22.5 ± 0.4	4.0 ± 0.0	na*	na*
3% NaOH	61.8 ± 1.3	17.6 ± 0.5	3.0 ± 0.1	2.7 ± 0.1	20.7 ± 1.8	na*
3% NaOH +0.3% MA	80.1 ± 1.8	4.0 ± 0.5	3.7 ± 0.1	1.9 ± 0.2	40.0 ± 1.1	20.6±0.7

*#Xylose, (g/g$_{hemicellulose}$); *na, not analyzed*

Chemical composition of SB varied significantly after pretreatments were applied. Based on the cellulosic fractions obtained after pretreatment, one-stage NaOH (3% NaOH) pretreated SB showed appreciable increase in cellulosic fraction (61.8 wt%) compared to untreated SB (40.4 wt%), while its hemicellulose and lignin fractions decreased to 17.6 wt% and 3.0 wt% respectively. Similarly, the second-stage pretreatment with 0.3% maleic acid showed much significant increase in cellulosic fraction (80.1 wt%), while hemicellulose and lignin fractions decreased to 4.0 wt% and 3.7 wt% respectively. Thus, lignin removal from SB was mainly attributed to NaOH pretreatment step. Delignification of SB by alkaline pretreatments has been reported in the literature (Zhang et al., 2018). Concerning the ash content, one-stage NaOH pretreatment led to significant decrease in ash content (2.7 wt%), while no significant effect on the ash content was obtained after second-stage pretreatment with maleic acid.

Alkaline pretreatment of SB using NaOH led to the removal of major part of lignin fraction with the retention of cellulose and hemicellulose fractions. Conversely, maleic acid pretreatment removed major part of hemicellulose and acid soluble fraction of lignin, while the cellulosic components and insoluble lignin remained largely unaffected. However, pretreatment filtrates from first-stage pretreatment with 3%NaOH and second-stage pretreatment with 0.3% maleic acid contain significant amount of xylose. This suggests that part of hemicellulose was solubilized during both the first-stage NaOH and second-stage maleic acid pretreatments. Furthermore, filtrates from the second-stage maleic acid pretreatment were found to contain very low amount of total furans (20.6 mg/L). The low amount of total furans obtained in this study emphasizes the main benefit associated with the use of organic acid such as maleic acid for pretreatment of SB.

3.3 XRD analysis of untreated (raw) and pretreated SB biomass

Lignocellulose crystallinity could be transformed via pretreatment by opening crystal hydrogen bonding, degrading amorphous constituents and increasing crystal regions, thereby affecting subsequent enzymatic saccharification (Zhang et al., 2018). Thus, the XRD patterns and CI of untreated and pretreated SB were investigated and the results are as shown in Figure 1. The CI of untreated SB was 55.2% and after pretreatment with 3.0% NaOH, CI increased to 69.0%, while a higher CI of 79.2% resulted from the combination of NaOH and maleic acid pretreatment steps.

However, all the pretreated SB presented higher CIs than the control (raw material). This phenomenon was mainly due to the removal of amorphous hemicellulose and lignin. The highest CI value indicates greater removal of hemicelluloses, leaving the crystalline cellulose fraction intact in the pretreated solid residues (Timung et al., 2016). Hence, crystallinity of cellulose was found to increase after pretreatments were applied to SB, mainly due to the increase in cellulose content (Rezende et al., 2011).The CI was found to correlate with the result of chemical composition analysis.

3.4 Studies on amenability of the pretreated biomass to enzymatic hydrolysis

In the literature, plenty of reports are available showing cellulose conversion rates of variedly pretreated lignocellulosic biomass after subjecting them to enzymatic hydrolysis.

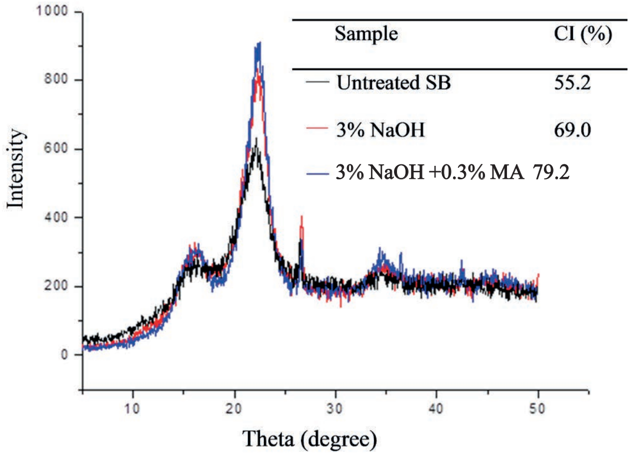

FIGURE 1: Diffractograms and crystallinity index (CI) of untreated SB and solid fractions resulting from pretreatment with 3.0% NaOH (one-stage pretreatment) and 3.0%NaOH+0.3%MA (one-stage NaOH pretreatment followed by the second-stage maleic acid pretreatment) respectively.

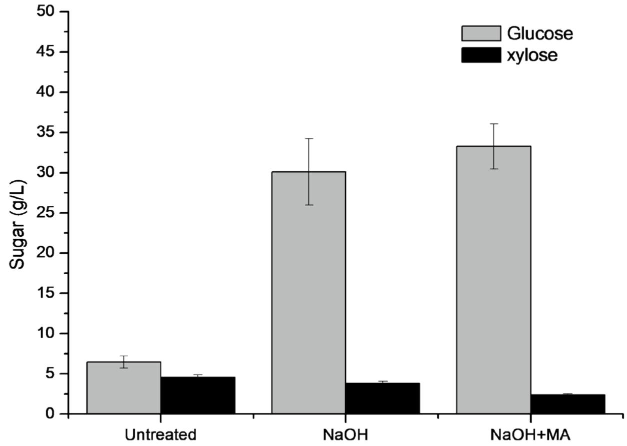

FIGURE 2: Xylose and glucose concentration after 72 h of enzymatic hydrolysis of untreated SB and solid fractions resulting from pretreatment of SB with 3.0% NaOH (one-stage pretreatment) and 3.0%NaOH+0.3% maleic acid (one-stage NaOH pretreatment followed by the second-stage maleic acid pretreatment respectively.

Few of them are listed and compared with the results in the present study (Table 4). The lowest conversion rate of 12.9% was obtained from untreated SB (raw material), while the highest conversion rate of 70.4% was obtained from SB emanating from two-stage NaOH and maleic acid pretreatments. The conversion rate of SB (60.2%) emanating from first-stage NaOH pretreatment was significantly higher than the untreated SB. This suggests that the first-stage pretreatment with 3% NaOH effectively remove lignin from SB, thereby improving its enzymatic digestibility. On the other hand, the second-stage pretreatment with 0.3% maleic acid was able to remove greater part of hemicellulose, which further enhanced enzymatic digestibility of SB.

The presence of lignin and hemicellulose negatively affects enzymatic hydrolysis by binding to cellulose, thereby impeding its access to cellulose (Zheng et al., 2018; Sabanci et al., 2018). Therefore, the lowest conversion rate obtained from untreated SB suggests inaccessibility of the enzymes to cellulose due to the presence of lignin and hemicellulose. The enzymes inaccessibility to cellulose appeared to be drastically reduced via pretreatments, hence the improved cellulose conversion rate obtained from the pretreated SB. On the other hand, enzymatic hydrolysates contained low amount of xylose (Figure 2), which can be attributed to the effectiveness of the acid pretreatment step in hemicellulose removal. Also, the presence of xylose in the hydrolysates may be connected to the action of cellulases and β-glucosidase enzymes which caused the disruption of lignocellulosic matrix thereby releasing xylose (Guilherme et al., 2015).

3.5 Fermentation

Table 5 shows the batch fermentation profile of the *S. cerevisiae* -IQAr/45-1 on enzymatic hydrolysate of two-stage

TABLE 4: Comparison of one-stage NaOH pretreatment, two-stage NaOH + maleic acid pretreatment and other pretreatment methods on the yield of fermentable sugars from sugarcane bagasse after enzymatic saccharification.

Biomass source	Pretreatment conditions	Type of Enzyme cocktail	Saccharification/ Conversion rate	Reference
Sugarcane bagasse	Raw (untreated)	Cellulase + β- glucosidase (Novozyme 188)	12.9%	This study
	3%NaOH (121°C, 60 min)		60.2%	This study
	Two-stage, 3%NaOH (121°C, 60 min) + 0.3% Maleic acid (121°C, 60 min)		70.4%	This study
	1% NaOH at 600 W for 4 min, microwave	Commercial cellulase (Zytex)	66.5%	(Binod et al., 2012)
	1%NaOH (115°C, 20 min)	Primafast 200	38.8%	(Thite et al., 2019)
	Two-stage, 1%H_2SO_4 (120°C, 30 min) + 0.5% NaOH (120°C, 60 min)	Cellic CTec2	71.4%	(Zhang et al., 2018)
	Two-stage, 1%H_2SO_4 (120°C, 30 min) + 60% ethanol (120°C, 60 min)		45.9%	(Zhang et al., 2018)
	1% H_2SO_4 (115°C, 20 min)	Primafast 200	25.6%	(Thite et al., 2019)

TABLE 5: Fermentation profile of enzymatic hydrolysate of two-stage (3.0%NaOH+0.3%maleic acid) pretreated SB by *Saccharomyces cerevisiae* (IQAr/45-1 strain) after 30 h.

Time (h)	Glucose (g/L)	Ethanol (g/L)	$Y_{E/S}$ (g/g)	Q_{Pp} (g/L/h)	Biomass (g/L)	$Y_{X/S}$ (g/g)	Q_{Px} (g/L/h)	E_F (%)
0	17.3 ± 0.20	0.62 ± 0.03	0.04	0.00	0.17 ± 0.00	0.01	0.00	0.0
30	0.27 ± 0.01	8.94 ± 0.12	0.49	0.28	8.17 ± 0.18	0.47	0.27	95.8

NaOH and maleic acid pretreated SB after 30 h. Glucose was depleted within 30 h of fermentation, by this time the ethanol concentration and ethanol yield have reached 8.94 g/L and 0.49 g/g respectively. Ethanol productivity of 0.28 g/L/h was obtained after 30 h with the corresponding fermentation efficiency of 95.8% based on the theoretical ethanol yield. On the other hand, the yeast cell biomass increased to a maximum level of 8.19 g/L and biomass yield of 0.47 g/g was obtained after 30 h with productivity of 0.27 g/L/h.

High ethanol yield is an indication of efficient conversion of glucose to ethanol by *S. cerevisiae* -IQAr/45-1. The maximum ethanol yield in this study was higher than those found in many literature reports. For example, de Albuquerque Wanderley and co-workers (2013) reported a maximum ethanol yield of 0.40 g/g (equivalent to fermentation efficiency of 78.47%) from batch fermentation of enzymatic hydrolysate of delignified SB by *S. cerevisiae*. Also, Wi et al. (2015) reported ethanol yield of 0.435 g/g (equivalent to 85.0 % of the maximum theoretical ethanol yield) from enzymatic hydrolysates of rice straw pretreated with hydrogen peroxide-acetic acid (HPAC) after fermentation by *S. cerevisiae*.

The high biomass yield obtained in this study could be attributed to the presence of yeast extract and peptone in the fermentation media. According to Chang and Webb, (2017), essential elements (nitrogen, phosphorous and vitamins) need to be supplied to the fermentation media as prerequisite to optimize ethanol yield, since hydrolysate from lignocellulosic biomass is generally regarded to be nutrient-deficient (Lau and Dale, 2009).

4. CONCLUSIONS

The effect of one-stage NaOH and two-stage NaOH and maleic acid pretreatments on cellulose enrichment and enzymatic digestibility of SB were studied. Two-stage NaOH and maleic acid pretreatment led to ~ two-fold increase in cellulosic fractions from SB. First-stage NaOH pretreatment efficiently removed lignin fraction, while the second-stage pretreatment with maleic acid efficiently removed hemicellulose fraction. The result of XRD analysis correlated with the changes in the chemical composition of SB as a result of pretreatments.

The presence of low amount of inhibitors in the pretreatment filtrate after the second-stage maleic acid pretreatment highlights the advantage of maleic acid over inorganic acid. Furthermore, two-stage NaOH and maleic acid pretreatment enhanced the enzymatic digestibility of SB up to 70.4%. The enzymatic hydrolysate was efficiently utilized by *S. cerevisiae* -IQAr/45-1 to produce high yield of ethanol. Thus, optimization of enzymatic hydrolysis at low enzyme loading is expected to further improve the process and reduce cost.

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COMPARATIVE STUDY OF CONSTRUCTION AND DEMOLITION WASTE MANAGEMENT IN CHINA AND THE EUROPEAN UNION

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ABSTRACT

Construction and demolition waste (CDW) accounts for 40% of urban municipal waste in China and around 25% in the European Union (EU). Since the EU is more developed and urbanized than China, its experience with managing CDW may be helpful to China. This study therefore compared China and the EU with respect to the flow of CDW materials and the policies, laws and regulations for CDW management. The results reveal that the CDW management practices and facilities in China are relatively underdeveloped with a large amount of low-value inert material going to landfill compared with the EU. The study also reveals the important role of government involvement in CDW management, including the use of punitive measures and preferential policies; most EU members states achieved their waste recovery rates by 2016 due to mature CDW legalization. To improve the management of CDW in China, a series of suggestions are proposed including waste prevention strategies, establishment of supervision mechanisms, and financial support.

1. INTRODUCTION

Construction and demolition waste (CDW), generated from construction, renovation, and demolition activities, account for 40% of urban municipal waste in China and 25%-30% of all waste generated in the European Union (European Commission, 2020; Jin et al., 2017). The main disposal methods for CDW are incineration, recycling, and landfill, with the latter being the most widely used method in China. Ortiz et al. (2010) assessed the three disposal methods in terms of carbon emissions and found that landfill caused the greatest carbon emission compared to the other two CDW disposal methods. As two of the largest economies in the world, China and the European Union (EU) generate massive amounts of CDW annually at 2.36 billion tonnes and 307 million tonnes respectively (Figure 1), with landfill and dumping having caused serious environmental problems in both jurisdictions (Deloitte, 2017; Zheng et al., 2017).

Although some regulations and laws have mitigated the impacts caused by CDW, a large number of environmental issues are still caused by illegal dumping and landfills (Gao et al., 2015; Liu et al., 2019; Nie et al., 2015). For instance, large areas of land have been contaminated by landfill that contain hazardous waste from CDW. Accordingly, CDW management has attracted worldwide attention as a way to deal with the serious environmental problems caused by increasing amounts of CDW. One of the most important guiding principles of CDW management is the 3R principle, which stands for reduce, reuse, and recycle (Huang et al., 2018). Preventing waste generation at source can be achieved through modular building, advanced design standards, and high labor quality (Esin and Cosgun, 2007). Recycling and reuse of CDW depends on such things as regulations, the market for recycled materials, awareness, and recycling technologies and systems (Begum et al., 2009; Jin et al., 2017).

CDW typically comprises metal, glass, plastics, timber, concrete, mortar, and bricks, all of which have great potential for recovery and reuse. The background of CDW management in China and the EU is very different. Due to insufficient attention to the management of construction waste, the overall recycling rate of CDW in China is less than 10% in China; the country still has a long way to go to achieve a 13% CDW recovery rate before the end of 2020 as proposed by the 'Waste Recycling Development Guidance' (The Ministry of Commerce of the People's Republic of China, 2017). By comparison, the average CDW recovery rate in the EU was about 89% for the year of 2016, even though the recycling rates ranged from about 100% to less than 5%

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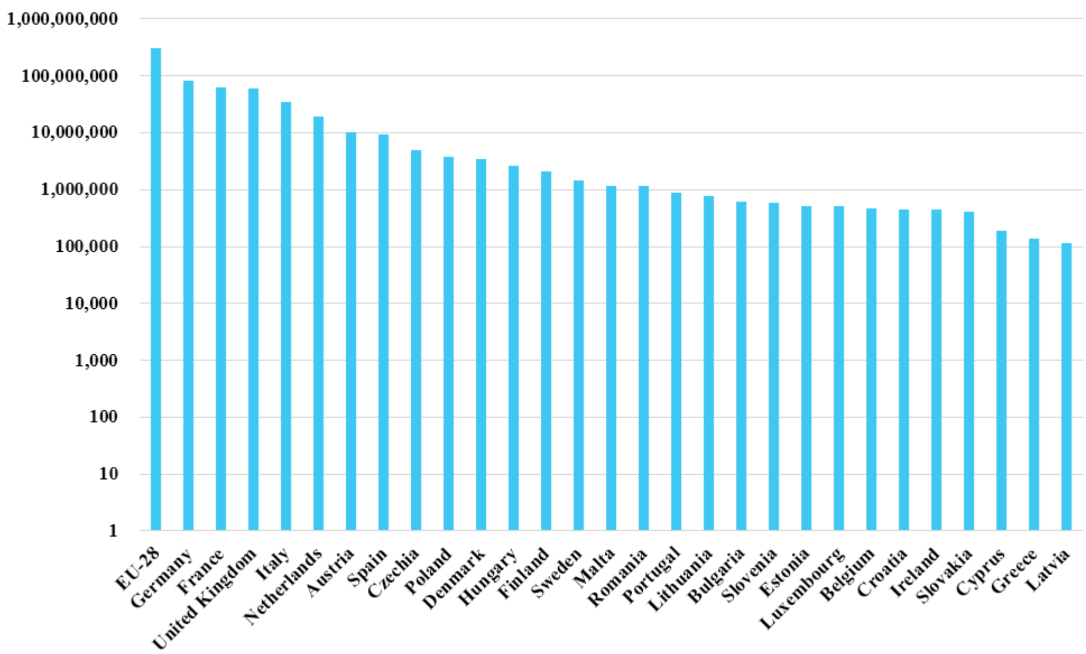

FIGURE 1: CDW generated by the 28 member states of the EU in 2016.

among the 28 different member states due to variations in the level and performance of CDW management practices in those states (Eurostat, 2020); the European Parliament (2008) approved a Directive to require all member states to increase their recovery rate to at least 70% by 2020. Clearly, more effort is required by China and the EU to achieve their respective sustainable goals for CDW. Although the current circumstances and waste management strategies differ in China and the EU, efficient CDW management practices can be further developed to help in both jurisdictions. This paper compares the current situation and CDW management regulations and policies in China and the EU, and discusses the future development direction for CDW management.

2. CDW MANAGEMENT IN CHINA AND THE EU

This section compares CDW management in China and the EU in relation to the flow of CDW materials, policies and strategies.

2.1 Flow of CDW materials

2.1.1 China

The generation and disposal of CDW in China has been extensively studied. Zheng et al. (2017) presented the ideal life cycle of CDW after it has been generated at construction and demolition sites (Figure 2): hazardous wastes are collected separately and sent to special disposal place, while other CDW is then preliminarily sorted to remove materials with a high recycle value, such as metal, plastic, and timber, which are then sold to companies that make recycled products. However, this represents less than 10% of the total CDW generated on site with the vast majority going to landfill or recycled disposal. A typical on-site waste storage facility in China is shown in Figure 3, which shows the storage of (a) hazardous waste (asbestos), (b) plastics, and (c) metal. Lack of on-site and off-site facilities and equipment lead to low recycling rates for the majority of CDW, including concrete, mortar, and bricks. These inert wastes can be crushed by machinery at construction or recycling centers and the resulting aggregate can be used as the raw materials for bricks, concrete, or sub-crust. Some projects have been implemented in China to enhance the development of on-site sorting and off-site CDW treatment (Bao et al., 2019), although most of the generated waste still goes to landfill or is dumped.

2.1.2 European Union

The level of CDW management in the EU varies considerably. For example, the rate of landfill for CDW range from 1% for the Netherlands to 99% for Greece (Deloitte, 2017). Though the recycling strategies are different, the general CDW generation and disposal flows are similar throughout Europe. According to a case study in Spain (Mercante et al., 2012), the generated waste is collected and stored on site

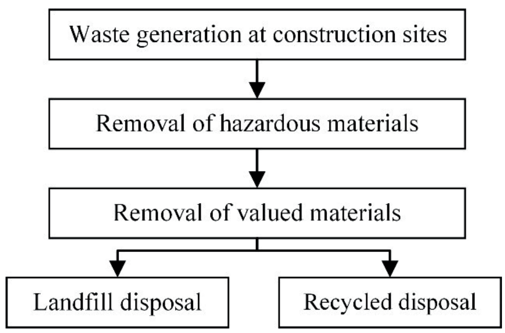

FIGURE 2: Material flow of CDW in China.

FIGURE 3: On-site separate collection: (a) asbestos; (b) plastic; and (c) rebar.

without sorting. The stored CDW is the periodically transported to off-site sorting plants to classify valuable waste (metal, paper, plastics, and timber) and inert waste (concrete and brick). These two types of waste are then sent to companies that produce corresponding products and recycled aggregate. The residues that do not have a recycled value are landfilled or incinerated. As shown in Figure 4, 24 of the 28 countries that made up the EU in 2016 reached their targets in 2020, which is a 70% recovery rate (European Parliament, 2008). Especially for the Netherlands, Luxembourg, and Belgium, the CDW recovery rates are more than 99%. Compared with China, these EU countries have more advanced CDW management facilities and technologies, and use inert waste to produce secondary aggregate for road construction (Deloitte, 2017).

2.2 Laws and regulations for CDW management

Laws and regulations can have a positive impact on the management of CDW through fair reward and punishment mechanisms. This section presents the current policies and regulations for stimulating the development of CDW recycling in China and the EU.

2.2.1 China

A plethora of laws and regulations have been published by China's Central Government in the past thirty years to promote CDW management (Table 1). These policies provide guidance to regional governments and offer tax breaks for companies involved in the CDW recycling industry. Although China's Central People's Government publish detailed regulations at provincial and city level to encourage recycling and reuse of CDW, there are still some limitations to the government's policies and strategies.

- Lack of regulations or practices on the reduction and prevention of CDW generation. Even though the many laws and regulations emphasize the importance of recycling and reuse of CDW, there are no government measures in place to prevent and reduce the generation of CDW at source.
- Lack of preferential policy for recycled products. There are very few policies that encourage and support the development of CDW recycling industries through tax exemptions and the like.
- Lack of supervisory mechanisms. Even though some regulation state that construction companies should

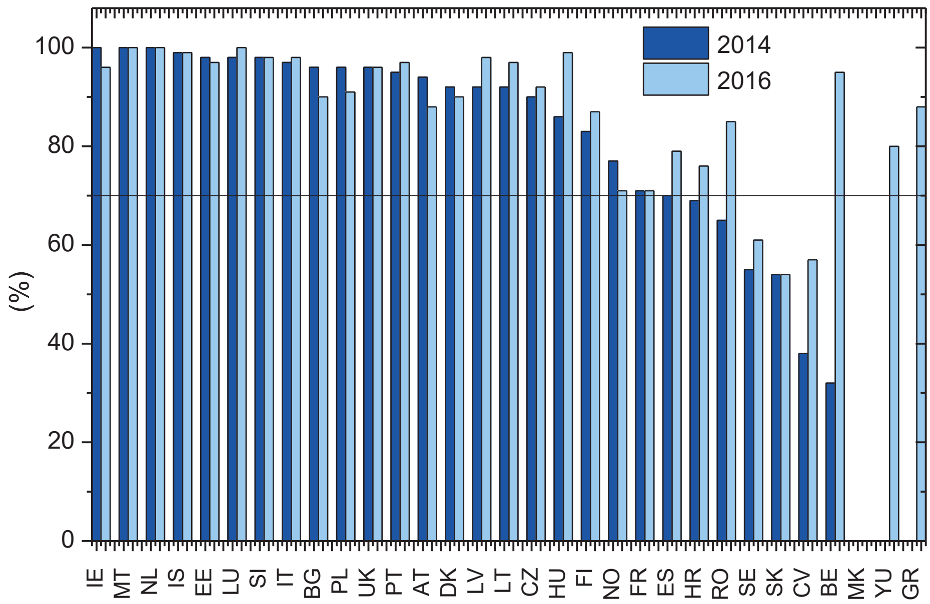

FIGURE 4: Recovery rate of construction and demolition waste in EU-27 (Eurostat, 2020).

TABLE 1: Laws and regulations for CDW management in China.

Type	Name	Year Issued	Content
Laws	Law of the People's Republic of China on the Prevention and Control of Atmospheric Pollution*	1987	Construction earthwork, muck and waste should be timely cleaned. Construction muck and waste should be utilized as a resource.
	Environmental Protection Act	1989	Authorities should take proper actions to organize the classification, recycling, and disposal.
	Law on the Prevention and Control of Environmental Pollution by Solid Waste**	1995	Construction companies should clean, transport, and dispose of CDW generated during the construction process and avoid contamination.
	Law on Promoting Clean Production	2003	Companies that use waste as raw materials to manufacture products shall have a tax preference in accordance with state regulations.
	Circular Economy Promotion Law	2008	Construction companies should be responsible for the recycling and reuse of CDW. For waste that cannot be recycled, qualified organizations shall be entrusted with the disposal of non-hazardous waste.
Regulations	Regulations on the management of urban construction waste	2005	This regulation formalized the construction waste disposal activities in urban areas, including dumping, transferring, landfilling and reusing.
	Implementation Plan for Comprehensive Utilization of Bulk Solid Waste	2011	Achieving 30% CDW recycling rate in cities.
	12th Five-Year Plan Guidance on Comprehensive Utilization of Resources	2011	Promoting the use of recycled materials on road construction and backfill and building complete recycling system for CDW.
	Catalogue of Value-Added Tax Preferences for Products and Labor Involving Utilization of Resources	2015	Production and sales of recycled aggregates should be exempted from value-added tax when the waste account for at least 90% of the raw materials for production.
Standards	Recycled fine aggregate for concrete and mortar	2010	Technical standard for recycled fine aggregate for concrete and mortar.
	Recycled coarse aggregate for concrete	2010	Technical standard for coarse aggregate for concrete.

*Note: *-amended in 2015; **-amended in 2016*

be responsible for the recycling, reuse, and disposal of CDW, the laxed supervisory mechanism makes it very difficult to implement.

- Lack of a standardization system. Apart from some technical standards for recycled products, no norms and standards for the demolition, sorting, transportation, and disposal have been developed to build a standardized system of CDW management.

2.2.2 European Union

The EU's Waste Framework Directive (2008/98/EC - European Parliament, 2008) imposes relevant recycling goals for waste to be achieved by 2020. The directive states that at least of 50% of waste such as plastics, metal, paper, and glass have to be prepared for reuse and recycled. Furthermore, not less than 70% of non-hazardous construction and demolition waste has to be prepared for reuse, recycled or recovered, including for back filling operations. The European Commission (2015) launched an EU action plan for the implementation of a circular economy by "closing the loop". The plan highlights CDW as one of the five priority areas, together with plastics, a bio-based economy, critical raw materials, and secondary raw materials, for implementation of a circular economy in the EU.

Most member states of the EU have introduced the target directive of a 70% CDW recovery rate by 2020 into their national legislation for CDW management. Germany, Estonia, Flanders of Belgium, and the Netherlands have higher targets, which are 75%, 80%, 85%, and 90% CDW recovery rates respectively. In addition, some countries of the EU have built their own legal framework to promote CDW management and the recycling of waste. Deloitte (2017) assessed the level of CDW management legalization in member states of the EU and concluded that Austria, Belgium, Denmark, Finland, France, Germany, Luxembourg, the Netherlands, and Sweden have relatively mature legal frameworks for sustainable and resource efficient CDW management. Table 2 summarizes the current laws and regulations that comprise the framework and the obligations for specific activities related to CDW in these countries. The whole disposal process of CDW can be defined from waste generation to the end of the life cycle, including the obligation of selective demolition, on-site or off-site sorting, separate collection, green public procurement, and landfill taxes or bans. Among these policies, landfill taxes and bans could be two of the most efficient practices to enhance CDW recovery rates (Deloitte, 2017). However, although 28 member states in the EU in 2019 had implemented landfill taxes, Sáez and Osmani (2019) demon-

TABLE 2: Legal framework for EU countries with mature CDW management legislation.

Centuries	Laws and regulations	Obligations in legalisation
Austria	Waste Management Act 2002 Remediation of Contaminated Sites Austrian Ordinance for Tracking Waste List of waste ordinance Separation of Construction Waste Ordinance Landfill Ordinance Hazardous Waste Ordinance End-of-Waste Act Recycled Construction Materials Regulation	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Belgium	Three different legislation in Flemish, Brussels Capital, and Walloon regions respectively	☑Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Denmark	Environmental Protection Act No. 879 26/06/2010 Statutory Order No. 1309/2012 Statutory Order No. 1662/2010 Circular of 15 July 1985	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Finland	Waste Act 646/2011 Land use and building Act 132/1999 Environmental protection Act 527/2014	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
France	Law 2009-967 Decree n°2011-610 Law 2010-788 Decree n°2014-1501	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Germany	Waste Disposal Act The Circular Economy Act The Circular Economy Act Waste Classification Ordinance Federal Soil Protection Ordinance Federal Soil Protection Act Ordinance on the Management of Municipal Wastes Landfill Directive	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Luxembourg	Law of 21 March 2012 on management of waste Grand-Ducal Regulation of 24 February 2003 on landfilling of waste	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Netherlands	Environmental Protection Act The National Waste Plan The Decree on landfills and waste bans The Decree on notification of industrial and hazardous waste From Waste to Resource	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax
Sweden	Ordinance on Environmental assessment SFS 2013:251 Waste Ordinance SFS 2011:927 Building Code (SFS 2010:900) Swedish Ordinance on PCB (SFS 2007:19) The Regulation NFS 2004:4	☑ Selective demolition ☑ CDW sorting ☑ Separate collection ☑ Hazardous waste management ☑ Green public procurement ☑ Landfill tax

strated that the CDW recovery rate was not correlated with the level of landfill tax due to the complicated situation in each country.

3. RESULTS AND DISCUSSION

The literature review summarized the materials flow of CDW and related policies for CDW management in China and the EU. This section compares CDW management in these two respects and identifies the differences as shown in Table 3.

3.1 Comparison of CDW management in China and the EU

3.1.1 Materials flow of CDW

In China, CDW management practices and facilities are still developing. Only waste that has a high recycled value

TABLE 3: Comparison of CDW management in China and the EU.

Terms	China	European Union
CDW recovery rate	<10%	89%
Distribution of CDW recycling facilities	Some facilities have been in operation in some developed cities.	Relevant waste recycling facilities were distributed in most member states.
Preferential policies	Tax exemption for recycled products.	Tax exemption and loans for use of recycled materials in some nations.
Landfill taxes/bans	No.	24 of 28 member states have implemented landfill taxes or bans.
Regulatory mechanisms	Lack of punitive regulations and corresponding organizations.	17 of 28 member states have implemented pre-demolition audits.

or hazardous waste can be sorted and collated separately on site or from plants off-site. Other low-value inert materials, such as concrete, brick, mortar, and masonry, accounting for 90% of total generated CDW, are typically transferred to landfills as mixture (Villoria Sáez et al., 2019). This type of waste can be of disposed by specific machinery or plants to produce recycled aggregates, which can then be used for backfilling and raw material for concrete and road construction. Although this technology is not yet widely applied in China, it is being used in some regions of the country. For example, in the city of Suzhou a public-private partnership (PPP) project initiated by the government established a CDW recycling company that is capable of disposing of more than one million tons of CDW annually over a six-year operational period (Bao et al., 2019); the recycled products include recycled bricks and aggregates used for buildings and road construction. However, since low profitability and massive initial investments can be barriers to the wider adoption of such CDW recycling facilities in China, the success of this type of project depends heavily on strong government support.

By comparison to the situation in China, most member states in the EU have relatively developed CDW recycling technologies and facilities. In addition, the recovery amount and rate of CDW is highly related to the density of the recycling network in the member countries (Sáez and Osmani, 2019). Deloitte (2017) determined that insufficient CDW recycling facilities and systems are the main reason for Cyprus' and Slovakia's relatively low recovery rate at 58% and 55% respectively in 2016 (see Figure 5). Early planning before construction and demolition activities begin could also be a practice for enhancing CDW management in the EU. Selective demolition is an obligation under national or regional legalization in the EU countries with mature CDW management legislation as shown in Table 2, and 17 of the 28 EU member states have implemented pre-demolition audits to evaluate the types and amount of CDW generated in deconstruction activities (Deloitte, 2017). The advantage of waste recycling of selective demolition is that most of the materials with high recycled value are manually demol-

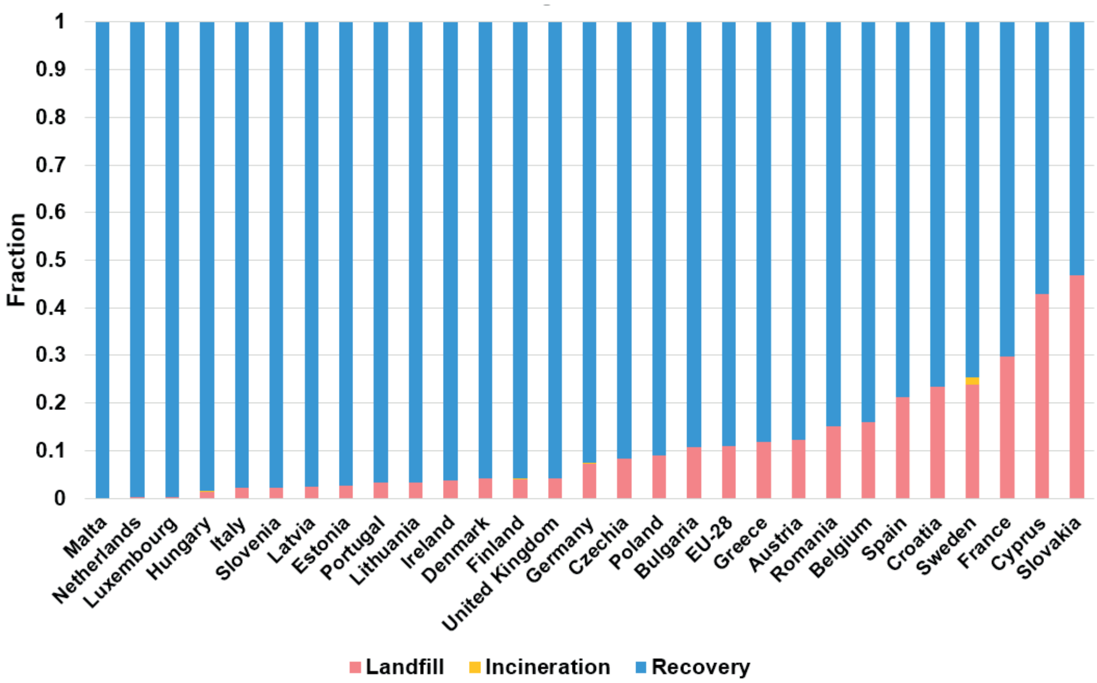

FIGURE 5: Management of CDW in the 28 member states of the EU in 2016.

ished by tools or light machinery; thus, a higher recovery rate can be achieved than by unsorted waste generated by conventional demolition (Kourmpanis et al., 2008).

3.1.2 Laws and regulations

Government involvement plays an important role in CDW management. The maturity level of the legal framework and preferential policies can significantly affect the development of CDW management. Although many laws and regulations govern the responsibility of CDW recycling, reuse, and disposal at the national and provincial level of China's government structure, implementation depends heavily on the punitive measures in place at the municipal authority level to encourage compliance. In terms of preferential policies, China's Ministry of Finance introduced a value-added tax preference and exemptions on the production and sale of recycled materials (State Administration of Taxation, 2011). In addition, relevant departments in some areas invested in and assisted the establishment of companies that focus on CDW recycling and reuse. Related CDW recycling facilities and plants are operating in some big cities in China, such as Suzhou, Chongqing, Dongguan, and Beijing (Bao et al., 2019).

An early start in many countries of the EU has led to relatively developed and mature legalization for CDW management. In 2008, the Waste Framework Directive was published by the European Parliament (2008) setting the goal of a 70% CDW recovery rate in all member states by 2020. By 2016, almost all 28 member states, with the exception of Slovakia and Cyprus, had achieved that goal (Eurostat, 2020). Different countries have developed various legal frameworks for CDW management as shown in Table 2, but CDW recovery can be affected by many other factors apart from policies and regulations. For instance, although no laws or regulations were specifically made for the promotion of CDW management in Poland, the CDW recovery rate was 92% in that EU member state (Sáez and Osmani, 2019). Deloitte (2017) claimed that the introduction of landfill taxes in most EU member states has been an effective practice for enhancing CDW recovery rates in the EU, even though previous studies had shown that CDW recovery rates did not have a correlation with the rate of landfill taxes (Sáez and Osmani, 2019).

3.2 Suggestions for future improvement of CDW management

To further enhance the recovery and management of CDW, the following suggestions are made:

- *Waste prevention strategy*
  Reducing waste at source is the most effective method for mitigating the environmental impacts caused by CDW generation. However, currently only a very few strategies have been implemented to prevent the generation of CDW, even though goals might be set in some countries or regions (Deloitte, 2017). Using waste prevention strategies in the design phase has been proven to be a possible solution. For instance, from a case study in South Korea it was found that using a building information model or flexible design can avoid 4.3% to 15.2% waste produced from design changes due to clashes of building elements (Won et al., 2016). Other practices, such as prefabrication and accurate quantity take-off, can also be effective techniques for waste prevention (Gálvez-Martos et al., 2018). Proper guidance by relevant governments or departments are required to promote and implement waste prevention.

- *Establishment of mature supervision mechanism*
  The aim of punitive regulations, such as landfill taxes, is to promote development of the recycling industry by imposing additional cost on unwanted activities. However, without mature supervision mechanisms, implementation of punitive regulations can give rise to an increasing number of illegal activities. For instance, owners might tend to send the waste to illegal landfills instead of approved recycling facilities if government imposed landfill taxes and are high and there is a lack of supervision (Deloitte, 2017). Huang et al. (2018) proposed a CDW supervision system for China, which includes: (1) a department for practice standardization; (2) a monitoring system for the whole CDW life cycle; (3) strict punitive regulations; (4) guidelines and norms for CDW prevention and recycled products; and (5) encouragement to use CDW recycled products.

- *Financial support*
  High investment risk is regarded as the main barrier to operating a CDW recycling facility (Zhao et al., 2010). Although CDW recycling centers have been operating in some areas of China and the EU, which show that it is economically feasible, several case studies have found that profitability is highly correlated with support policies and market maturity (Bao et al., 2019; Nunes et al., 2007). The financial aid (loans) and related policy support (public procurements) offered by governments can assist companies with their initial investment in CDW recycling plants. Attracting social capital is therefore very much involved in the development of CDW management.

4. CONCLUSIONS

If not dealt with appropriately, the massive amounts of CDW generated in China and the EU could lead to very serious environmental problems. Governments and related industries have made a great effort to minimize the potential risks through CDW management. However, differences in urban development, technologies, policies have led to variations in the level of CDW management and formulation of relevant regulations between China and the EU. This study compared CDW management in China and the EU in respect of waste flows and legal frameworks and arrived at the following conclusions:

- The EU has much more advanced CDW management systems compared to China. The CDW recovery rate in 2016 for the EU was 89%, whereas it was less than 10% in China.
- Landfill is the main CDW disposal method in China but new CDW recycling facilities are being developed.

Dense CDW recycling networks and tools such as selective demolition could be the reasons for the high CDW recovery rate in the EU. However, factors such as lack of supervision mechanisms, inadequate practices, and limited standardization systems, give rise to the low efficiency of some CDW management systems.
- The EU member states have relatively mature legal frameworks and management systems for CDW. Although the specific regulations differ from state to state, most member states have laws or regulations specifically governing CDW management, such as landfill taxes or bans, and green public procurement.
- Suggestions for future development include improved waste prevention strategies, establishment of supervision mechanisms for illegal CDW disposal, and economic support for recycling facilities.

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HIERARCHICAL MODELLING FOR RECYCLING-ORIENTED CLASSIFICATION OF SHREDDED SPENT FLAT MONITOR PRODUCTS BASED ON HYPERSPECTRAL IMAGING

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ABSTRACT

The number of flat monitors from televisions, notebooks and tablets has increased dramatically in recent years, thus resulting in a corresponding rise in Waste from Electrical and Electronic Equipment (WEEE). This fact is linked to the production of new high-performance electronic devices. Taking into account a future volume growth trend of WEEE, the implementation of adequate recycling architectures embedding recognition/classification logics to handle the collected WEEE physical-chemical attributes, is thus necessary. These integrated hardware and software architectures should be efficient, reliable, low cost, and capable of performing detection/control actions to assess: i) WEEE composition and ii) physical-chemical attributes of the resulting recovered flow streams. This information is fundamental in setting up and implementing appropriate recycling actions. In this study, a hierarchical classification modelling approach, based on Near InfraRed (NIR) - Hyperspectral Imaging (HSI), was carried out. More in detail, a 3-step hierarchical modelling procedure was designed, implemented and set up in order to recognize different materials present in a specific WEEE stream: End-of-Life (EoL) shredded monitors and flat screens. By adopting the proposed approach, different categories are correctly recognized. The results obtained showed how the proposed approach not only allows the set up of a "one shot" quality control system, but also contributes towards improving the sorting process.

1. INTRODUCTION

In recent decades, the volume of flat monitors deriving from televisions, notebooks and tablets present in Waste from Electrical and Electronic Equipment (WEEE) has increased dramatically (Zeng et al., 2018; Salhofer et al., 2011). Indeed, the continuous and rapid change in technologies rapidly renders devices obsolete, therefore proving easy to discard and replace with a newer version (Chancerel and Rotter, 2009; Oliveira et al., 2012; Palmieri et al., 2014). However, a significant amount of valuable materials is contained in WEEE, and metals and/or alloys, precious metals, and high-quality plastics can be profitably recovered. Copper, aluminum, lead and zinc are the main valuable non-ferrous metals contained in WEEE, although precious metals such as gold, platinum, palladium and silver may also be detected.

The implementation of a metal valorization action (i.e. recovery and recycling) at the end of an industrial process is a technological challenge, starting from a thorough characterization of this specific material stream (Bonifazi et al., 2018; Robinson 2009). One of the most interesting components of WEEE is represented by Printed Circuit Boards (i.e. PCBs), rich in copper and potentially containing precious metals such as gold, silver and palladium (UNEP 2009). Plastics are utilized in electronic equipment manufacturing due to their excellent properties and low-cost, with polymers being lightweight, highly flexible and readily workable at low temperatures. As reported in Table 1, significant amounts of total generated WEEE are represented by polymers (Makenji et al., 2012; Ongondo et al., 2011). Flat monitors are viewed as a relatively "young" WEEE product due to their recent introduction onto the market. Different categories of materials may be recovered from this waste due to its heterogeneity. More in detail, liquid crystal displays (i.e. LCDs), focus of the present study, consist of several parts: top cover, lightbox unit (i.e. consisting of a metal frame, LCD glass panel, plastic frame, a number of plastic diffuser sheets, Perspex sheets, cold cathode fluorescent, reflective foil and lightbox support frame), PCB mounting

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frame, LCD control layer (i.e. PCBs, speakers, cables) and back cover (Ryan et al., 2011). Table 2 shows a typical LCD composition in terms of materials and weight (%).

Polycarbonate (PC) or polymethyl methacrylate (PMMA) are found in End-of-Life (EoL) monitors and LCD screens (Tarantili et al., 2010; Suresh et al., 2017; Suresh et al., 2018). PMMA is used in monitors and screens for its optical clarity, high light transmission, toughness and high impact resistance, and both PCs or PMMAs are used in Light Guide Panel (i.e. LGP) manufacturing (Hwang and Ko, 2019). LGP is often present in EoL monitors and LCD screens (Lee et al., 2006; Suresh et al., 2018), as a part of backlight units (BLU). The identification of these plastic-based fragments is therefore crucial for recycling purposes, in order to obtain "waste-recovered" polymers for re-use (Suresh et al., 2017). PMMA and optical enhancement films are of pure optical quality and suitable for re-use and alternative applications (Veit and Bernardes, 2015). The recovery and reuse of flat monitor optical components for high-end applications may also contribute towards achieving economical sustainability for small and medium recycling plants (Ljungkvist et al., 2016).

The treatment of EoL monitors and LCD screens involves a series of complex activities: manual disassembly, removal of hazardous components (i.e. mercury-containing fluorescent lamps, CCFL from LCD panels) and mechanical recycling (i.e. size reduction and sorting stages). Typically, materials having a commercial value that can be recovered from flat monitors include metals (i.e. zinc-coated steel, aluminum and cable copper content), PCBs, PMMA light diffusers, optical enhancement films and recyclable plastics ABS, HIPS and polycarbonate (Veit and Bernardes, 2015). Indeed, heterogeneous materials such as metals, PCBs, electrical components and plastics were detected in the analyzed sample. These product ranges are of considerable interest for their residual economic value in a full circular economy logic. More in detail, a Near Infrared (NIR) - Hyperspectral Imaging (HSI) based approach was applied to characterize materials in the analysed sample, in order to define quality control/sorting logics in a recycling scenario.

Due to the potential risks associated with the incorrect treatment of WEEE and the considerable amount of "resources" contained in these wastes, it is essential to ensure the implementation of appropriate actions in a WEEE recycling process, both from an economic and environmental point of view. Generally, the chemical composition and physical properties of a material will define the recycling options available. Therefore, the importance of characterization is linked to the possibility of developing processing steps aimed at the recovery of secondary raw materials for use in a range of industrial applications. The present study was carried out in line with this perspective.

2. MATERIALS AND METHODS

2.1 Samples

The samples analyzed (Figure 1) consisted of EoL flat monitors and screens originating from a shredding line (i.e. hammer mill shredder) of a WEEE recycling plant, following semi-automatic disassembling aimed at removing the main electronic components. Sample collection was performed by a coning and quartering procedure, followed by manual sorting; each collected and analyzed sample weighed 215 g. The main components detected in the samples studies included Light Guide Panel (LGP) fragments, black plastics, metals, PCBs and electrical components and other materials (i.e. different polymers/plastics and cellulose-based particles). Weight percentages of the hand-sorted material categories are shown in Figure 1.

In order to build the hierarchical classification model, the sample was split into two sub-sets: i) a training set to calibrate the classification model and ii) a validation set to validate the model. The training set was composed of 228 particles (77% in weight of the sample), subdivided according to material categories: " LGP fragments" (33 Wt%), "Black plastics" (25 Wt%), "Metals, PCBs and electrical components" (32 Wt%) and "Other" (10 Wt%), whilst the validation set was made up of 68 particles (approx. 23% in weight of the total sample). Nine hyperspectral images were acquired: 7 images were used for training purposes, while 2 images were used to validate the hierarchical model (Figure 2).

2.2 Hyperspectral imaging

HSI utilizes an integrated hardware and software architecture to digitally capture and handle spectra (Hyvarinen

TABLE 1: Typical materials contained in a WEEE product (Makenji et al., 2012).

Material type	Weight (%)
Metals	60
Plastics	15
Cathode ray tube (CRT) and liquid crystal display (LCD) screens	12
Metals/plastic mixture	5
Pollutants	3
Cables	2
Printed circuit boards	2
Others	1
Total	100

TABLE 2: Typical materials contained in an LCD (Ryan et al., 2011).

Material type	Weight (%)
Ferrous	45
Plastic	21
PCBs	10
Glass	9
Non-Ferrous	3
Speaker	3
Hips	2
Sheets	2
Others	6
Total	100

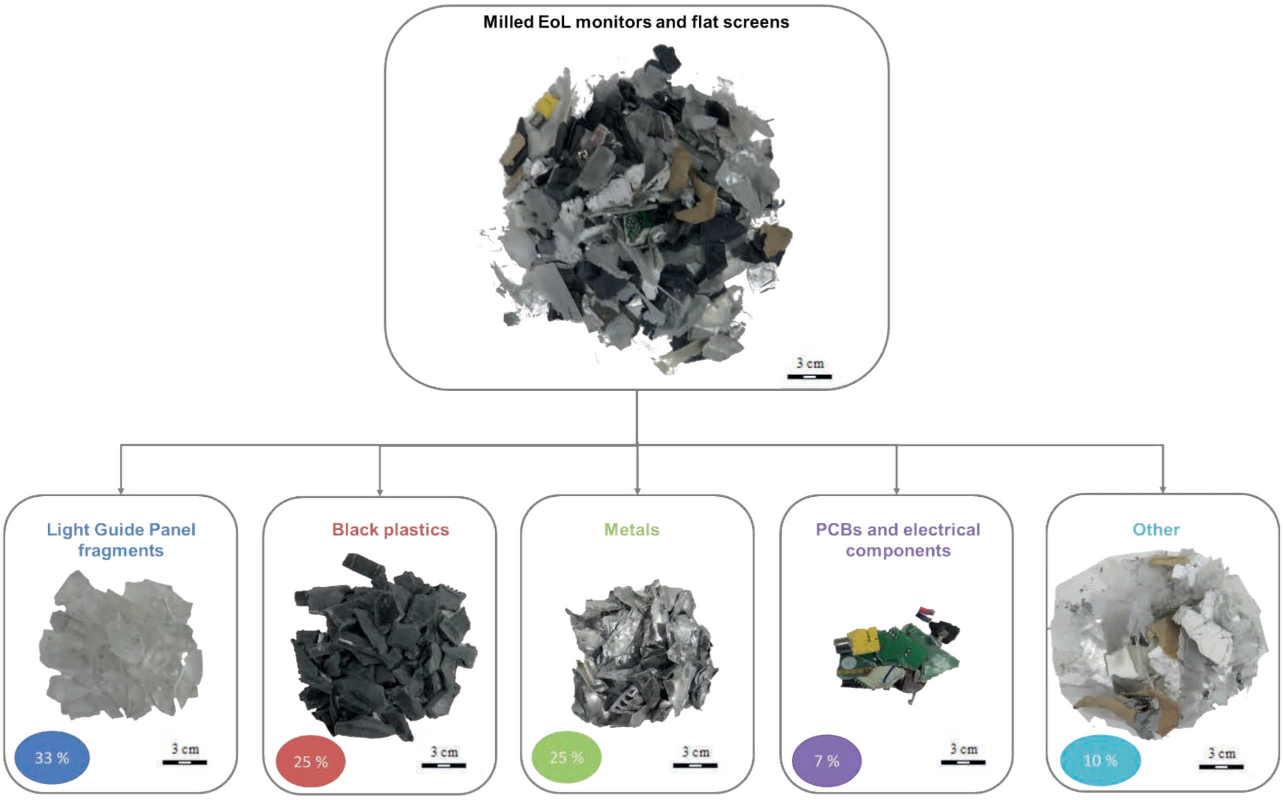

FIGURE 1: Diagram of the analyzed sample: product resulting from a milling line of a WEEE recycling plant and main material categories resulting from manual sorting. The composition (weight %) of the different main material categories is reported.

et al., 1998; Geladi et al., 2007), allowing both spatial and spectral information to be collected concomitantly from the investigated material. This information is enclosed in a 3D dataset (i.e. the "hypercube"), in which two dimensions are spatial and the other provides spectral information. The HSI technology facilitates the study of several physical and chemical characteristics of a sample: different features can thus be analyzed according to the investigated wavelengths.

NIRS (Near InfraRed Spectroscopy) techniques are utilized to perform both qualitative and quantitative analysis in different fields: i.e. in the primary/secondary raw materials sector (Masoumi et al., 2012; Bonifazi et al. 2015), in cultural heritage (Agresti et al., 2013; Capobianco et al., 2015), in the agricultural and food industry (Teixeira dos Santos et al., 2013; Kumuravelu et al., 2015; Tsuchikawa et al. 2015; Serranti et al. 2018a; Serranti et al. 2018b), in the pharmaceutical and chemical industry

FIGURE 2: Digital images representing the samples used as validation set.

(Larrechi et al. 2003; Roggo et al. 2007), in clinical application (Gasbarrone et al. 2018; Bonifazi et al. 2018a; Currà et al., 2019) and, more generally, in analytical science (Pasquini, 2003).

In recent years, the use of NIR-HSI has grown rapidly in many sectors, including the solid waste sector (Bonifazi et al., 2019). In literature, different HSI-based approaches are proposed for plastic recycling (Ulrici et al., 2013; Bonifazi et al., 2015), construction and demolition waste recycling (Palmieri et al., 2014; Serranti et al., 2015; Bonifazi et al., 2015; Bonifazi et al., 2017; Bonifazi et al., 2018c) and WEEE recycling (Palmieri et al., 2014; Bonifazi et al. 2018b).

The novelty of the present study relies on the possibility of recognizing different material categories in a specific WEEE stream (i.e. shredded EoL flat monitors and screens) by utilizing a hierarchical modelling-based approach.

2.2.1 Hyperspectral images acquisitions and data handling

Hyperspectral imaging acquisitions were carried out at the Raw Materials Laboratory (Latina, Italy) of the Department of Chemical Engineering, Materials and Environment (Sapienza - University of Rome, Italy). A NIR Spectral Camera™ equipped with an ImSpector™ N17E (SPECIM Ltd, Finland), a spectrograph working in the Near Infrared wavelength range (i.e.1000 - 1700 nm), was used to perform hyperspectral image acquisitions. Spectral Scanner (ver. 1.2) software was used to acquire and collect hyperspectral data. Spectral data were then analyzed using PLS_Toolbox (Version 8.7, Eigenvector Research, Inc.) under MATLAB (Version R2019a, The Mathworks, Inc.) environment.

2.2.2 Hierarchical classification procedure

The hierarchical classification procedure was set up based on 3 rules. In other words, three classification steps were performed to recognize different materials constituting the representative sample of the investigated WEEE (Figure 3). In the 1st classification step, LGP fragments (i.e. "Light Guide Panel fragments" class) were distinguished from other materials (i.e. "Other (1)"); in the 2nd step, "Black Plastic" was distinguished from "Other (2)" starting from the "Other (1)" class; in the 3rd classification step, the particles belonging to the "Other (2)" class were identified either as "Metals, PCBs and electrical components" or as "Other (3)". Following this approach, each final recovered product could thus be forwarded to the relevant recycling lines in order to recover metals/alloys and/or plastics.

In order to recognize the different analyzed categories, a Partial Least Squares - Discriminant Analysis (PLS-DA) was applied for each step of the classification, according to a cascade detection procedure (Barker and Rayens, 2003; Ballabio and Consonni, 2013). An ad hoc combination of pre-treatment algorithms was applied to the data for each rule (Rinnan et al., 2009). In the 1st rule, preprocessing algorithms applied were Standard Normal Variate (SNV) and Mean Center (MC); in the 2nd rule, the algorithm applied was MC; in the 3rd rule, preprocessing algorithms applied were SNV, Smoothing and MC. Each model was cross-validated using the Venetian-blinds algorithm.

Since PC and PMMA are the most common polymers in LGP (Chen and Yu, 2007; Hwang and Ko, 2018), an additional classification procedure was set up and implemented to identify polymers. Virgin PC and PMMA were used as training samples in order to recognize the polymer constituting LGP fragments. In this case, PLS-DA was chosen as classification method (Barker and Rayens, 2003; Ballabio and Consonni, 2013) and Mean Center was used as pre-processing algorithm (Rinnan et al., 2009).

3. EXPERIMENTAL RESULTS

3.1 Hierarchical classification

Raw reflectance spectra used to calibrate the hierarchical classification model is shown in Figure 4. Validation set with real classes and the prediction map, as resulting from the hierarchical classification modelling, are shown in Figure 5. Figure 6 illustrates the reference map of particles. The number of pixels correctly classified (Table 3) was computed for each particle, according to its label, as shown in Figure 6.

The hierarchical model reached a Recognition (number of particles correctly assigned divided by the total number of particles in the set) equal to 0.926, 5 particles (4 "Oth-

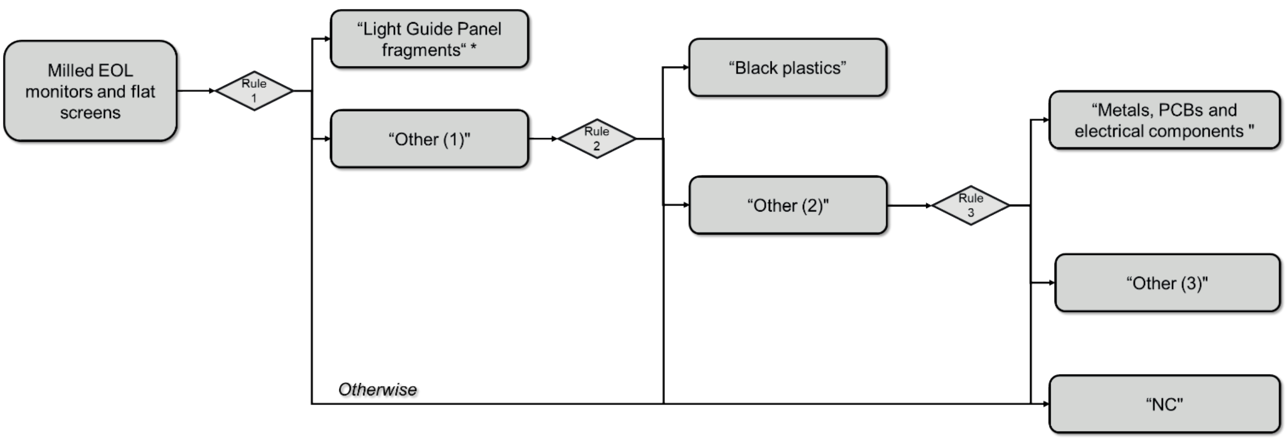

FIGURE 3: Flow-chart of the implemented hierarchical classification modelling. *Starting from Light Guide Panel (LGP) fragments, a classification procedure was applied to perform polymers (PC and PMMA) identification.

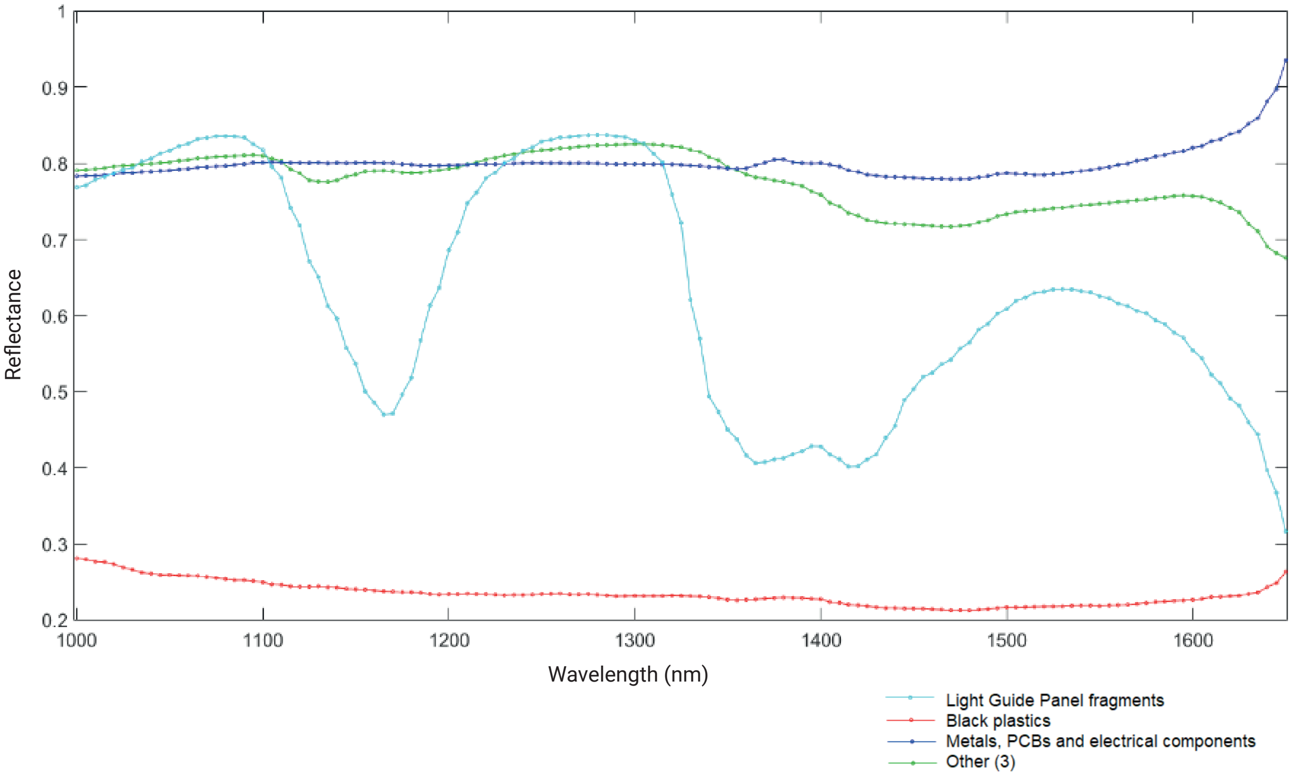

FIGURE 4: Raw spectra of modelled classes from calibration set.

er (3)" particles and 1 "Metals, PCBs and electrical components" particles) out of 68 were not correctly classified (7.35 %).

3.2 Polymer identification

The image obtained, including only the particles classified as "LGP fragments", was used as validation set to perform a new classification addressed at identifying polymer type. Literature reports (Hwang and Ko, 2018) reveal how LGP is mostly constituted by PC or PMMA. This fact can be "simply" verified comparing the reflectance spectra of two polymers: indeed, LGP collected spectral signatures are, on initial visual inspection, very similar to PMMA (Figure 7). To correctly assess LGP polymeric composition, reflectance

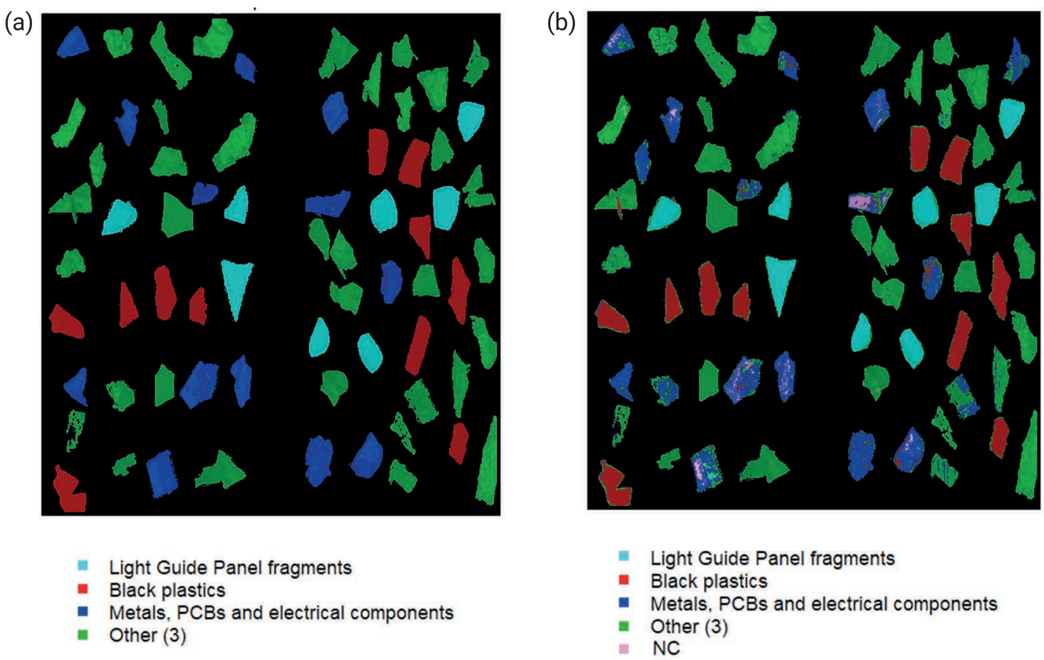

FIGURE 5: Validation set with real classes (a) and the prediction map resulting from the hierarchical classification modelling (b).

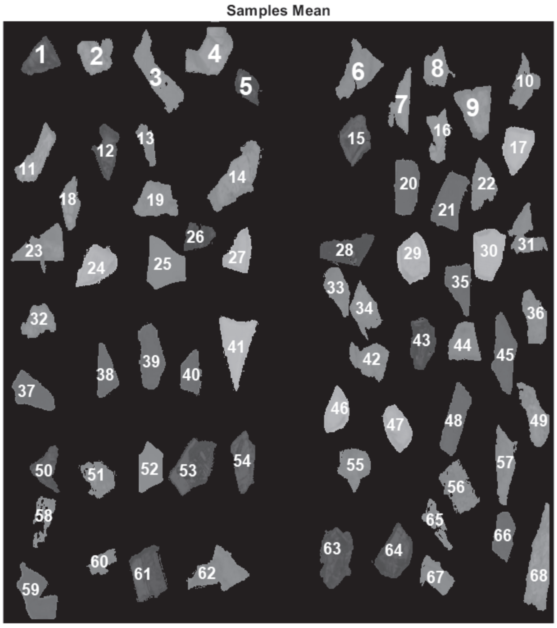

FIGURE 6: Reference map with particle labels used to compute the number of pixels correctly classified for each particle.

spectra of virgin PC and PMMA pellets were acquired and utilized to build the PLS-DA classification model.

Raw reflectance spectra of PC and PMMA classes were pre-processed using the mean center algorithm prior to the classification procedure. Classification results, in terms of prediction map, are shown in Figure 8. Sensitivity and Specificity in calibration, cross-validation and validation are

TABLE 3: Number of pixels correctly classified computed for each particle, reported in Figure 6.

Particle label	Real Class	Pixels correctly classified (%)
1	Metals, PCBs and Electrical Components	76.15%
2	Other (3)	95.11%
3	Other (3)	100.00%
4	Other (3)	98.85%
5	Metals, PCBs and Electrical Components	71.50%
6	Other (3)	99.13%
7	Other (3)	100.00%
8	Other (3)	99.51%
9	Other (3)	99.34%
10	**Other (3) ***	**28.60%**
11	Other (3)	97.14%
12	Metals, PCBs and Electrical Components	82.33%
13	Other (3)	98.74%
14	Other (3)	97.34%
15	Metals, PCBs and Electrical Components	88.52%
16	Other (3)	95.65%
17	Light Guide Panel fragments	90.33%
18	**Other (3) ***	**19.68%**
19	Other (3)	100.00%
20	Black plastics	88.48%
21	Black plastics	90.12%
22	Other (3)	100.00%
23	Other (3)	95.80%
24	Light Guide Panel fragments	80.87%
25	Other (3)	100.00%
26	Metals, PCBs and Electrical Components	56.97%
27	Light Guide Panel fragments	83.39%
28	**Metals, PCBs and Electrical Components *****	**29.05%**
29	Light Guide Panel fragments	84.20%
30	Light Guide Panel fragments	90.14%
31	Other (3)	95.10%
32	Other (3)	92.08%
33	Other (3)	98.89%
34	Other (3)	100.00%
35	Black plastics	84.95%
36	Other (3)	99.15%
37	Black plastics	83.25%
38	Black plastics	81.98%
39	Black plastics	89.53%
40	Black plastics	82.85%
41	Light Guide Panel fragments	86.39%
42	Other (3)	99.64%
43	Metals, PCBs and Electrical Components	69.26%
44	Other (3)	100.00%
45	Black plastics	90.92%
46	Light Guide Panel fragments	83.35%
47	Light Guide Panel fragments	87.50%
48	Black plastics	90.75%
49	**Other (3) ****	**8.83%**
50	Metals, PCBs and Electrical Components	91.93%
51	**Other (3) ***	**43.77%**
52	Other (3)	100.00%
53	Metals, PCBs and Electrical Components	71.63%
54	Metals, PCBs and Electrical Components	88.24%
55	Other (3)	99.29%
56	Other (3)	62.42%
57	Other (3)	92.60%
58	Other (3)	92.64%
59	Black plastics	80.74%
60	Other (3)	98.81%
61	Metals, PCBs and Electrical Components	54.37%
62	Other (3)	96.97%
63	Metals, PCBs and Electrical Components	90.82%
64	Metals, PCBs and Electrical Components	81.83%
65	Other (3)	99.72%
66	Black plastics	88.54%
67	Black plastics	85.50%
68	Other (3)	99.96%

** Misclassified as "Metal, PCBs and Electrical components", ** Misclassified as "Black plastics"; *** Not Classified - "NC"*

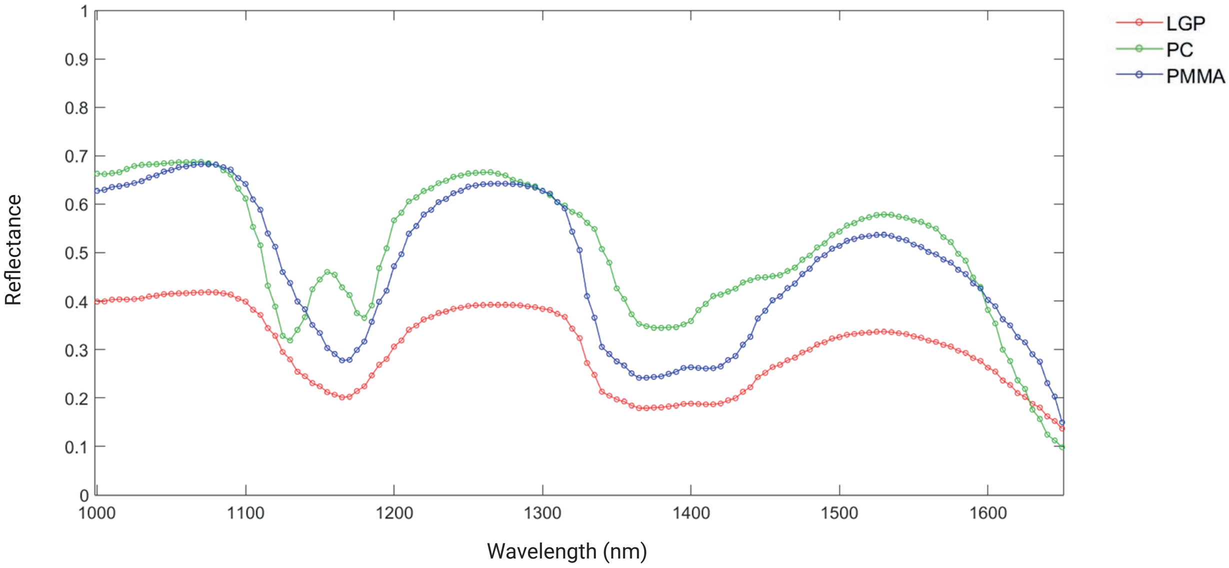

FIGURE 7: Raw spectra of virgin "PC" and "PMMA" polymers compared with LGP spectral signatures.

shown in Table 4: the built model identifies only PMMA as LGP constituent.

4. CONCLUSIONS

The present study was carried out with the aim of recognizing different categories of materials present in EoL milled monitors and flat screens using NIR-HSI techniques. Three classifiers were built to identify "Light Guide Panel fragments" (LGP), "Black Plastics" and "Metals, PCBs and electronical components". A hierarchical classification model was set-up and implemented, facilitating identification of the target categories in each step.

Promising results in classification were achieved. The presence of a few misclassified pixels is likely due to light scattering phenomena, sample surface heterogeneity and a possible presence of impurities. In order to improve the classification, a "machine vision" logic capable of assigning only one of the available classes according to a set threshold should be implemented. The logic to be used for thresholding should be based on the correctly assigned pixel percentage in each particle domain (e.g. correctly assigned pixels > 50 % of the total pixels included in the particle domain). Therefore, only one class could be attributed to each object in the image to be predicted.

TABLE 4: Performance indicators for PLS-DA model classifying "PC" and "PMMA" polymers.

	Class	Sensitivity	Specificity
Calibration	PC	1.000	1.000
	PMMA	1.000	1.000
Cross-validation	PC	1.000	1.000
	PMMA	1.000	1.000
Validation	PC	-	0.133
	PMMA	0.866	-

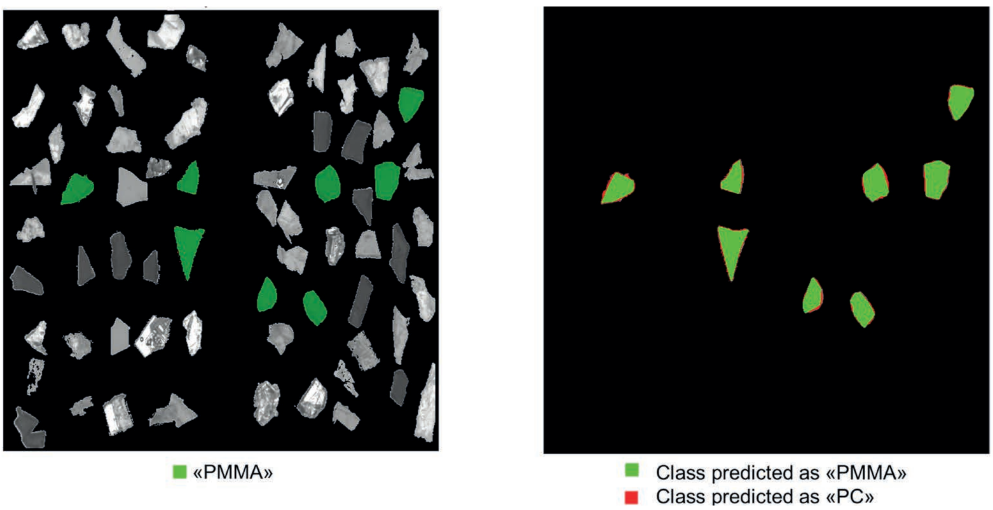

FIGURE 8: Reference map with particle labels used to compute the number of pixels correctly classified for each particle.

Additionally, HyperSpectral imaging in the NIR range was used to identify Light Guide Panel (LGP) fragments and subsequently recognize the constituting polymer type. An automatic NIR-HSI based recognition system of polycarbonate (PC) and polymethyl methacrylate (PMMA) applied to LGP fragments was developed. More in detail, Partial Least Square Discriminant Analysis (PLS-DA) method was used to set up the classification, starting from virgin pellets used as training set. All LGP fragments were correctly distinguished from the other particles. All LGP fragments were classified as PMMA, which, assuming that only one polymer constituting the analyzed LGP, is a reasonable finding. The results obtained represent a meaningful starting point for the implementation of a fast, non-invasive and reliable procedure for use as a driving force in the separation and quality control of materials originating from spent flat monitor waste stream.

The possibility of conducting a full quality check of materials throughout the entire chain of the recycling plant would likely contribute to the development of "on-line" quality control strategies and facilitate the issuing of material certification directly on site. The proposed approach features the advantages of being rapid, non-destructive and low cost. The reduction of processing costs is an important goal to pursue in the secondary raw materials sector, in which the use of expensive devices should be avoided in order to yield an efficient and economically feasible recycling process. The implemented procedure may be profitably employed to set up on-line strategies aimed at boosting the efficiency of recycling processes, reducing costs and improving the "final quality" of recovered products. Finally, if fully implemented, the proposed NIR-HSI approach would afford the possibility of developing a system capable of recognizing a series of different materials in a WEEE stream, suited to use not only as an analytical core with which to perform quality control, but also as a sorting engine.

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CHEMICAL EXPLORATORY ANALYSIS OF PRINTED CIRCUIT BOARD (PCB) USING INDUCTIVELY COUPLED PLASMA OPTICAL EMISSION SPECTROMETRY (ICP OES): DATA TREATMENT AND ELEMENTS CORRELATION

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ABSTRACT

Electronic waste is the fastest growing class of residue in the world. This material presents several electric and electronic equipment (EEE) with a huge amount of base, valuable and toxic elements, thus increasing its recycling interest. This study is aimed to perform an exploratory analysis of printed circuit board (PCB) using Inductively coupled plasma optical emission spectrometry (ICP OES). A PCB from hard disk (HD) was split in 77 sub-samples using a lathe following by mineralization process. This step was conducted without milling process. So, the sub-samples were weighted and mixed with concentrated aqua regia solution, followed by mineralization using microwave oven radiation. Twenty elements were determined by ICP OES (Al, Au, Ba, Ca, Co, Cr, Cu, Fe, Mg, Mn, Nd, Ni, Pb, Pd, Pt, Sb, Sn, Ti and Zn), and Flame atomic absorption spectrometry - FAAS (Ag). With the concentration results, several graphical analyzes were performed: (1) scores map and loading plot; (2) correlation plot and; (3) PCB element distribution maps. With this exploratory analysis, it was possible visualize and understand the data, observing correlations among the elements, how close these correlations are and how is this correlation around the PCB components. This strategy was a good way to observe the PCB complexity and the importance of recycling these materials.

1. INTRODUCTION

Nowadays, around 60 different chemicals elements are found in electrical and electronic equipment (EEE), making this type of material a powerful economic and technological source. While the lifespan of these equipment is decreasing over the years due to the fast-technological development, the amount of electronic waste (e-waste) increases (Baldé et al., 2017; Andrade et al., 2019a; Costa et al., 2018).

The composition of e-waste contains precious metals, valuable bulky materials, rare earth elements (REE), hazardous and scarce metals, being extremely important to recycle these obsolete materials. Initiatives related to recycling avoid health and environmental risks, minimizing impacts caused by primary metal extraction, being also a strategic commercial sector (Zhang et al., 2017; Bookhagen et al., 2018). Therefore, the e-waste should be considered a source of valuable resources, and not a common waste (Cucchiella et al., 2015).

Printed circuit board (PCB) is one type of e-waste, which is found in almost all EEE. It consists of a board composed of layers of polymers and fibrous materials, such as, glass fiber. Conductive lines are printed and, electronic components (transistors, capacitors, integrated circuits) are mounted on the board (Yamane et al., 2011; Cayumil et al., 2018). There are many elements on the PCB, increasing its complexity and heterogeneity, even so, the interest to recycle this type of material increase over the years (Yamane et al., 2011; Cayumil et al., 2018).

It is necessary several steps to recycle this material: pre-treatment (manual disassembling); processing (comminution, granulometric classification, magnetic and electrostatic separation); concentration (pyro and hydrometallurgical processing); recovery and purification (solvent extraction, precipitation/cementation) (Silvas et al., 2015). In the literature, there are some studies that show several recycling routes (Cui & Zhang, 2008; Huang et al., 2009; Park & Fray, 2009; Rao et al., 2020).

In the study proposed by Dervisevic et at., 2013, the quantitative analysis of the PCBs (individual components and complete PCB) from computer and mobile phones

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were performed using inductively coupled plasma optical emission spectrometry (ICP OES) and x-ray fluorescence (XRF). Besides that, structure and chemical composition, phase transformations and microstructural analysis were performed. In addition, a recycling procedure was proposed observing the effect of the extraction in different parts of PCB before the melting process. The authors suggested also new materials that can replace the harmful ones, as an example, Ga-Sb-Zn as a good lead-free solder material.

Tanvar et al., 2020, evaluate hard disk drive (HDD) as a source of Cu and REE and, performed physical separation in a customized water fluidization classifier for Cu recovery from PCB. The extraction of REE elements from the magnet were performed using microwave exposure-leaching and precipitation route. Another study was performed by Moosakazemi et al., 2020, where the authors proposed a cementation process using waste Al-based heat sinks as cementing agent to precipitate Sn and Pb from the dissolution of PCBs in HCl. El-Nasr et al., 2020, used a leach solution (ammonia salt leaching process) for PCB from old computers to prepare Cu nanoparticle using an ecofriendly and low-cost method with L-ascorbic acid as reductant and stabilizer. Mesquita et al., 2018, described a chemical characterization of connector pins from PCB of computers using Scanning Electron Microscopy – Energy Dispersive Spectroscopy (SEM-EDS) and ICP OES. The authors concluded that, due to leaching procedure of the mixed sample without grinding, the operating costs were reduced and metals recovery were maximized.

Andrade et al., 2019b, prepared a reference material for inorganic constituents on PCB samples. They evaluated different acid mixtures for leaching procedure. Diluted aqua regia presented the best performance using microwave radiation and ICP OES for determination of several elements. Instrumental neutron activation analysis (INAA) was used to compare the results obtained. After that, several characterizations were performed according to ISO Guides 30-35, such as homogeneity, stability (short and long) and chemical characterization of the reference material (Andrade et al., 2019c).

An important aspect in all studies is the characterization of the material and, one of the most used instrumental analytical techniques is ICP OES (Castro & Pereira-Filho, 2018). This technique is already well established, with many advantages such as multielement determination, accurate and precise determinations and, low limits of detection (LODs), being necessary dissolution of the solid samples. In this study, ICP OES was used to analyze a PCB from HDD. The goal was dedicated to exploratory analysis using correlation plot, PCB element distribution maps and, scores maps and loading plot using the concentrations acquired from ICP OES. The images help to interpret and correlate the different elements on the PCB, characterizing this material and collaborating with urban mining.

2. MATERIAL AND METHODS

2.1 Sample preparation, digestion procedure and ICP OES analysis

A PCB from HDD was chosen to perform this study. This PCB was mechanically fragmented in 77 square-shaped sub-samples (11 rows and 7 columns) with the help of a lathe. The size of each fragment was around 1 by 1 cm. To avoid intense sample preparation procedures, the fragments were not crushed and milled, but entirely used. Therefore, it was not possible to obtain replicates due to heterogeneity of the sample. Even for a single manufacturer, PCB's are not exactly the same in composition.

The fragments' weight ranged from 0.41 to 1.63 g, and were separated according to their weights to further mineralization, which was performed using a microwave system (Speedwave four, Berghof, Eningen, BW, Germany). Seven mL of concentrated aqua regia (HNO_3 and HCl in a 1:3 ratio) was used as acid mixture. The acids used were previously purified in a sub-boiling distillation system. Table 1 shows the heating program used in the microwave for DAK 100 tube, that allows up to 100 mL of volume.

After the mineralization, the final solutions volume was completed until 20 mL with deionized water (Milli-Q®, Millipore, Bedford, MA, USA). Consecutive dilutions were performed to ensure that the solutions would be fit for maximum allowed for dissolved solids and acidity. ICP OES (Thermo Scientific, iCAP 7000) was used to determine Al, Au, Ba, Ca, Co, Cr, Cu, Fe, Mg, Mn, Nd, Ni, Pb, Pd, Pt, Sb, Sn, Ti and Zn. The determinations were performed in axial mode, except for Ba and Ca, that were determined in radial mode, due to high concentration of these elements on the sub-samples. Table 2 shows the ICP OES instrumental parameters.

For Ag, Flame Atomic Absorption Spectrometry (FAAS) was used. The FAAS instrument (Varian AA240FS, Mulgrave, Australia) performed the determinations with a hollow cathode lamp of Ag. The spectrometer parameters used were wavelength of 328.1 nm, lamp current of 3 mA (75% of the manufacturer's recommendation), spectral resolution of 0.5 nm in absorbance measurement mode. The flame type was Air-C_2H_2 with a flow of 1.3 L min^{-1}.

Limits of detection and quantification were calculated

TABLE 1: Microwave heating program used for fragments' mineralization.

Steps	Temperature (°C)	Ramp time (min)	Hold time (min)	Percentage of maximum pressure* used (%)
1	120	5	5	60
2	150	5	2	70
3	200	5	15	80

** The maximum pressure is 30 bar*

TABLE 2: Instrumental parameters for ICP OES determinations.

Instrumental Parameters	Operation conditions
Power applied by RF (W)	1150
Nebulizer gas flow rate (L/min)	0.70
Auxiliary gas flow rate (L/min)	0.5
Argon gas flow rate (L/min)	12
Integration time / s	15 for low and 5 for high emission lines
Analytes	Emission Lines (nm)
Al	I 308.215
Au	I 267.595
Ba	II 493.41
Ca	I 422.673
Co	II 228.616
Cr	II 284.325
Cu	I 327.396
Fe	II 239.562
Mg	II 280.27
Mn	II 260.569
Nd	II 401.225
Ni	II 231.604
Pb	II 220.353
Pd	I 340.458
Pt	II 203.646
Sb	I 217.581
Sn	I 283.999
Ti	II 338.376
Zn	I 213.856

according to Equation 1 and 2:

$$LOD = \frac{3 \times sd_{blank}}{sensitivity} \quad (1)$$

$$LOQ = \frac{10 \times sd_{blank}}{sensitivity} \quad (2)$$

Where sd_{blank} is the standard deviation of blank solutions signals.

2.2 Exploratory analysis

With the concentrations obtained by ICP OES determination, PCB element distribution maps, scores maps and loading plot, besides correlation plots were performed in order to visualize and interpret the data. For the scores and loading, the data were auto-scaled and the images were constructed according to the results from Principal Component Analysis (PCA). For correlation plot, the values were calculated with Pearson correlation. More details about these plots are written in the section "Results and Discussion". All data organization, treatment, calculation, and figures preparation were made in MATLAB 2017b (Matworks, Natick, MA, USA.

3. RESULTS AND DISCUSSION

3.1 Concentration values acquired by ICP OES

A PCB from HDD was studied and, in this case, the goal was to verify the distribution of the elements on the board without milling process. The main advantages are minimum sample preparation and maximum recovery of the elements, since each small fragment is analyzed individually. On the other hand, for some fragments the resulting mineralized solution was not entirely free of solid residues. These samples contain a lot of polymers and silicon.

Table 3 presents the concentration values obtained by ICP OES for Al, Au, Ba, Ca, Co, Cr, Fe, Mg, Mn, Nd, Ni, Pb, Pd, Pt, Sn, Ti and Zn. For Ag, the results were obtained using FAAS. LOD and LOQ are also showed in Table 3. The PCB was split in 77 sub-samples, which were named according to the position of them on the board. For example, row 1 and column 1 is the first sub-sample. There are 11 rows and 7 columns. The concentration values are in a range from the minimum to maximum of each row among the 7 columns. In the case of Ag, concentration obtained in the row 1 (R: 1) and the 7 columns (C: 1 to 7) ranged from 201 to 415 mg kg^{-1} (see Table 3).

The most abundant element on the PCB is Cu with maximum concentration around 30% m m^{-1} in row 11 and column 3 (see Table 3, concentration value of 299482 mg kg^{-1}), followed by Fe (~ 11% m m^{-1}, row 7, column 3), Sn (~ 8% m m^{-1}, row 11, column 3), Ca (~ 7% m m^{-1}, row 8, column 6) and Pb (~ 4% m m^{-1}, row 11, column 2). Among the precious or noble metals, Pd is the most abundant on the PCB with 0.31% m m^{-1} in row 3 and column 2, followed by Au (0.15% m m^{-1}, row 10, column 3), Pt (0.11% m m^{-1}, row 7, column 3) and Ag (0.07% m m^{-1}, row 9, column 6).

Harmful elements, as Cr and Pb, were determined on the PCB with low concentration for Cr (maximum of 43 mg kg^{-1}). On the other hand, high concentrations for Pb were determined (from 0.16 to 4% m m^{-1}), which is present on the solder material. Neodymium, a REE, was also determined with high concentrations for some fragments (0.90% m m^{-1} for row 3 and column 4). Therefore, it can be observed huge variability of different elements that are present on a PCB from HDD.

3.2 Exploratory analysis using different tools

As described in the last section and Table 3, the results obtained for a single PCB are chemically rich, but how these elements are correlated? Is it possible to identify a group of elements that can be recycled together? These questions will be clarified in this section. For better visualization of the data, several images were built. The scores map and loading plot are based on PCA, as mentioned before, that describe the original dataset, simplifying and reducing the number of variables without losing information (Sperança et al., 2017; Carvalho et al., 2015; Santos et al., 2018). The data were organized in a matrix with 77 samples (sub-samples) and 21 variables (elements concentration) and, this dataset was auto-scaled to give the same importance to all variables. The dataset is represented in a new space with principal components (PC's). In this case, 21 variables or dimensions (elements concentration) were reduced to 2 new variables with 20% and 18% of explained variance for PC1 and PC2, respectively. Figure 1 shows the original PCB before the lathe process (split in 77 sub-samples with 11 rows and 7 columns), the scores and loadings for PC1

TABLE 3: Concentration acquired by ICP OES and FAAS (Ag), LOD and LOQ for all elements determined.

Fragments	Range	Ag	Al	Au	Ba	Ca	Co	Cr	Cu	Fe	Mg	Mn	Nd	Ni	Pb	Pd	Pt	Sb	Sn	Ti	Zn
R: 1 C: 1 to 7	Min	201	2875	< LOD	86	6974	<LOD	<LOD	39546	106	126	< LOD	<LOD	227	2448	< LOD	<LOD	<LOD	3358	< LOD	<LOD
	Max	415	29104		8708	65459	29	40	126196	11249	755	5464	4163	5217	8798	1336	143	1047	11910	2816	1753
R: 2 C: 1 to 7	Min	291	8733	< LOD	114	20212	<LOD	17	106846	308	245	< LOD	<LOD	977	4590	17	<LOD	<LOD	6525	102	195
	Max	457	28836	548	18411	64668	27	31	189153	8438	661	485	498	4018	11688	740	73	2750	16588	3088	951
R: 3 C: 1 to 7	Min	236	4883	< LOD	272	11019	<LOD	<LOD	88702	142	145	< LOD	<LOD	853	6796	28	<LOD	<LOD	9378	221	228
	Max	400	29409	111	24962	64790	27	30	160208	13625	693	5249	9046	5324	13158	3146	122	837	15999	2316	1681
R: 4 C: 1 to 7	Min	222	7576	< LOD	103	16517	<LOD	<LOD	108210	208	204	< LOD	<LOD	230	2559	29	<LOD	<LOD	3032	121	228
	Max	437	23482	490	14102	52864	12	22	200032	605	598	18	5547	2274	12568	789		1955	15034	2028	497
R: 5 C: 1 to 7	Min	115	6095	< LOD	<LOD	12931	<LOD	<LOD	88280	192	164	< LOD	<LOD	<LOD	2608	8	<LOD	187	4772	58	121
	Max	357	17339	385	4522	40698	70	32	181054	33442	446	324	12	16368	14619	96	332	3480	18275	1018	368
R: 6 C: 1 to 7	Min	90	2186	< LOD	<LOD	<LOD	<LOD	<LOD	85272	93	< LOD	< LOD	<LOD	<LOD	1571	< LOD	<LOD	<LOD	3193	< LOD	94
	Max	360	17945	308	6906	40461	1801	25	131434	25632	481	5535	5248	13606	13062	284	230	3540	16801	1648	465
R: 7 C: 1 to 7	Min	108	2456	< LOD	<LOD	6901	<LOD	<LOD	82691	326	120	< LOD	<LOD	<LOD	4440	< LOD	<LOD	<LOD	6144	< LOD	138
	Max	577	15188	355	8346	34296	2623	24	211269	107030	424	399	566	17008	14692	273	1058	2077	19523	1487	14174
R: 8 C: 1 to 7	Min	122	7837	< LOD	92	18060	<LOD	10	92856	341	229	< LOD	<LOD	<LOD	5212	< LOD	<LOD	<LOD	6735	102	171
	Max	489	31256	747	13138	67034	36	29	278284	6138	662	3408	15	3956	13492	228	48	2314	14383	172	330
R: 9 C: 1 to 7	Min	111	2017	< LOD	<LOD	4027	<LOD	<LOD	84071	445	< LOD	< LOD	<LOD	<LOD	5331	14	<LOD	<LOD	7845	< LOD	213
	Max	652	21807	255	9409	49073		29	213297	842	699	11129	17	3427	22926	427		974	25428	1992	5464
R: 10 C: 1 to 7	Min	135	1979	< LOD	<LOD	3935	<LOD	<LOD	67185	75	139	< LOD	<LOD	<LOD	4243	< LOD	<LOD	<LOD	6170	< LOD	155
	Max	518	31477	1477	174	61094	145	43	219580	26718	760	202		13258	26107	106	275	3280	31426	50	317
R: 11 C: 1 to 7	Min	429	9981	< LOD	110	20421	<LOD	<LOD	76793	287	275	< LOD	<LOD	269	8692	15	<LOD	<LOD	12342	125	128
	Max	548	23444	334	8729	47686		23	299482	571	492	17	16	2183	42288	318			78318	1701	11976
LOD		0.03	0.4	0.08	0.5	0.3	0.07	0.08	0.3	0.5	0.6	0.09	0.06	0.4	0.03	0.08	0.07	0.5	0.07	0.4	0.4
LOQ		0.09	1	0.3	2	0.9	0.2	0.3	1	2	2	0.3	0.2	1	0.1	0.3	0.2	2	0.2	1	1

** For each row (R), from column (C) 1 to 7, is presented the concentration range with minimum (Min) and maximum (Max) values for each element determined. The concentrations are in mg kg^{-1}*

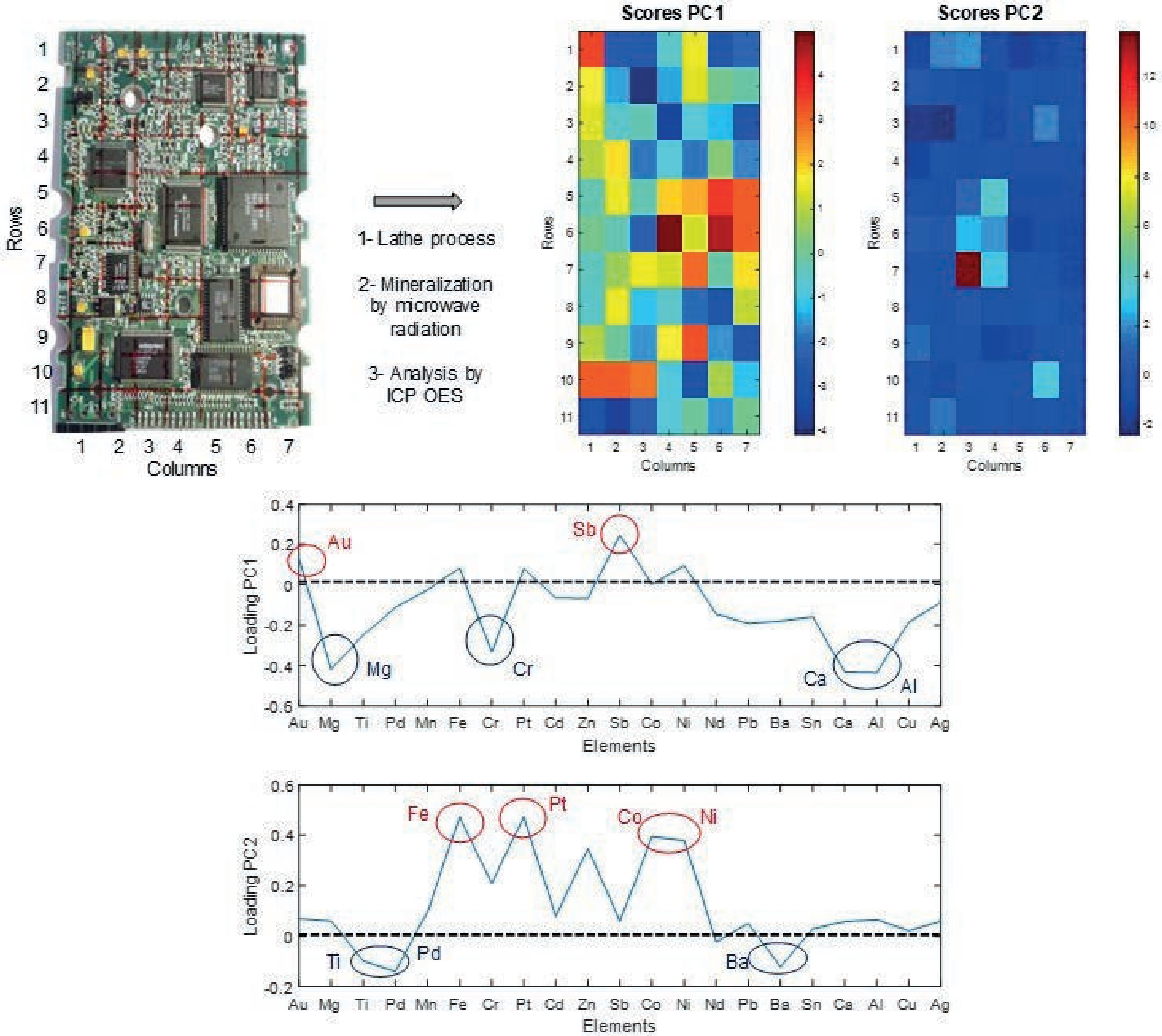

FIGURE 1: Scores map and loading plot of the PCB with decomposition in two principal components (PC1 and PC2).

and PC2. The scores represent the sub-samples and, the loadings the variables.

In this case, the red and blue colors are correlated with positive and negative loadings, respectively. According to PC1, the red spots (scores PC1) are more correlated with Au and Sb (loading PC1), and the blue spots with Mg, Cr, Ca and Al. For PC2, the red spot (scores PC2) is more correlated with Fe, Pt, Co and Ni (loading PC2), and the blue spots with Ti, Pd and Ba.

Another way to visualize this data is with the correlation plot that is shown in Figure 2. This plot is based on Pearson correlation, that shows how close two variables are to obtaining a linear relationship. The correlation value (R) range from -1 to 1 and this number refers: (1) correlations close to 1, that means a positive correlation where the variables change in the same direction; (2) null correlation (close to 0), none relationship between variables; (3) correlations close to -1, that means negative correlation where the variables change in an opposite way. The red spots have a positive correlation (close to 1), while the blue spots negative correlation (see the color bar for better interpretation).

Ca, Al and Mg, for instance, have a correlation close to 1 (maximum), and these three elements are correlated with Cr in a range of 0.8. Figure 3 shows the individual distribution on the PCB of these four elements using the concentrations acquired from ICP OES. The plots were made using the function "imagesc" from MATLAB. With Figure 3, it is possible to observe that Ca, Al and Mg have the same profile, with high concentration on the same sub-samples (rows 1 and 2 with columns 2 and 3, for example). And low concentration (blue spots) for sub-samples on the rows 5 and 6 with columns 5, 6 and 7 (also similar for Cr). The correlation of these three elements with Cr is not so high because just in some fragments the profile of Cr is similar with Ca, Al and Mg (row 1 with columns 2 and 3, for example). Ca, Al and Mg are present in the glass fiber used on the PCB. This material is composed of SiO_2 with the highest concentration, followed by CaO_2 and Al_2O_3. MgO is also

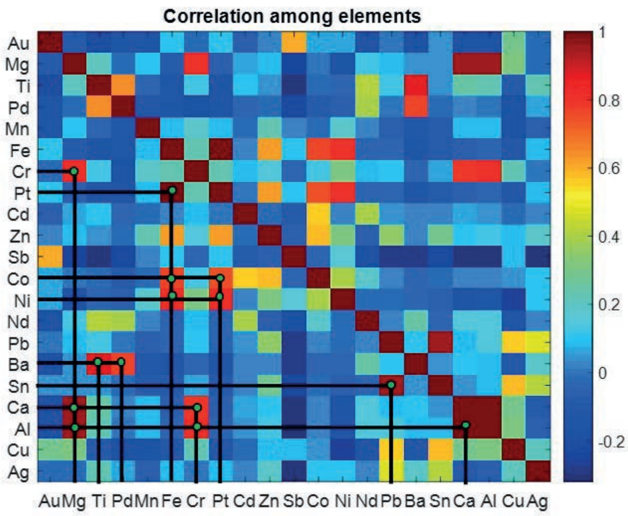

FIGURE 2: Correlation plot for all elements determined by ICP OES.

in the composition but with a lower concentration than Ca and Al (Sanapala, 2008), and this can be also observed in Figure 3 (see the color bar with the concentration values).

Other elements that have correlation close to 1 are Sn and Pb (Figure 2), which are on the solder material used to assemble the components on the PCB. The lead-based solders are banned in European Union, but in some regions are still used due to physic-chemical properties, high degree of "wetting" and the cost of Pb. Sn-Ag-Cu (Sanapala, 2008) and Ga-Sb-Zn (Dervisevic et al., 2013) alloys can be used as free-lead solder material, being the Sn-Ag-Cu alloy more used than Ga-Sb-Zn alloy. Figure 4 shows the individual distribution of Pb and Sn on the PCB and can be observed the same behavior for both elements, mainly concentrated on the row 11. Sn concentration is much higher than Pb concentration, which can be also found in anticorrosive coatings of other metals.

According to PC2 loading plot (Figure 1), Ba, Ti and Pd are correlated and observing Figure 2, this correlation is around 0.8. On the other hand, it is more about Ba/Ti and Ba/Pd. The relationship between Ba and Ti may be due to the barium titanate present on the ceramic material as dielectric in capacitors. Figure 5 shows the individual distribution of Ba, Ti and Pd. It can be observed the similar profile between Ba and Ti. But, for Ba and Pd, it can be considered an indirect correlation, where on row 3 and columns 1, 2 and 4, there are high concentration of Ba and Pd. The platinum group elements are used as coatings in switches and sensors.

Elements as Pt, Fe, Co and Ni appear correlated in PC2 loading plot (Figure 1) with Zn being also in the same plot quadrant. The correlation plot (Figure 2) contain Pt with Fe in a correlation close to 1, Co/Ni with Fe and Co/Ni with Pt in a correlation close to 0.8, but for Zn the correlation decrease around 0.6. Figure 6 shows the individual distribution for these elements. They have a similar behavior, mainly on the row 7 and columns 3 and 4. These elements can be found in some components on PCB, Fe in magnetic components and Ni in conductive films in resistors, for example.

A last correlation in the PC1 loading plot (Figure 1) is for Au and Sb with a value around 0.6 (Figure 2, correlation plot). Figure 7 shows the individual distribution on the PCB. The profile is a little bit similar with high concentration on row 10 and column 3, for example. Gold is used as coating of connection pins of micro-chips and in integrated circuits. Antimony can be used in soldering, Cu plating or connect to it on the connecting pins, as a semiconductor dopant, in addition, as additive of flame retardant polymeric composites.

There are other elements in Figure 2, but with low correlations with each other. For example, Cu is the most abundant element on the PCB and, in Figure 2 appears with low correlation with other elements. Cu is used on printed circuit tracks and in the connecting pins, in addition, it is also used in some electronic components.

4. CONCLUSIONS

The possibility of sample preparation without gridding

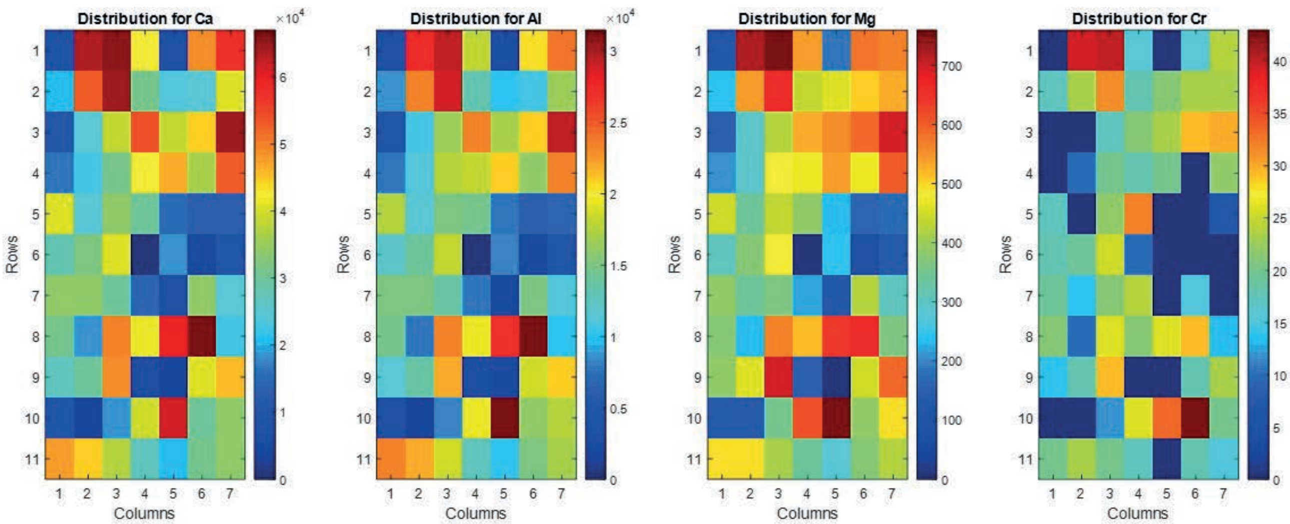

FIGURE 3: Distribution of Ca, Al, Mg and Cr on the PCB with individual concentration acquired by ICP OES.

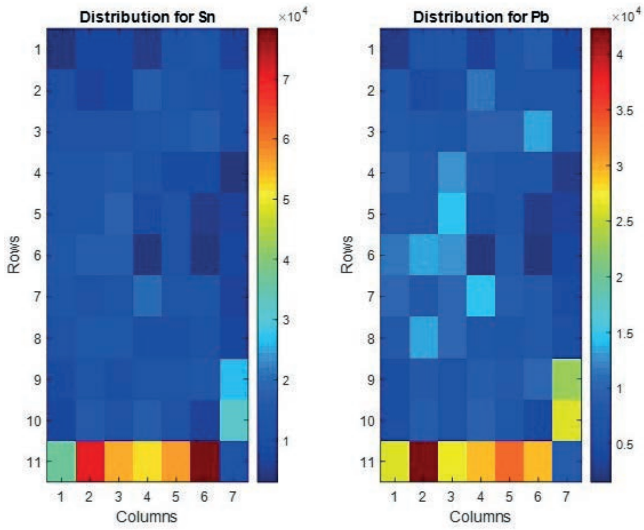

FIGURE 4: Distribution of Sn and Pb on the PCB with individual concentration acquired by ICP OES.

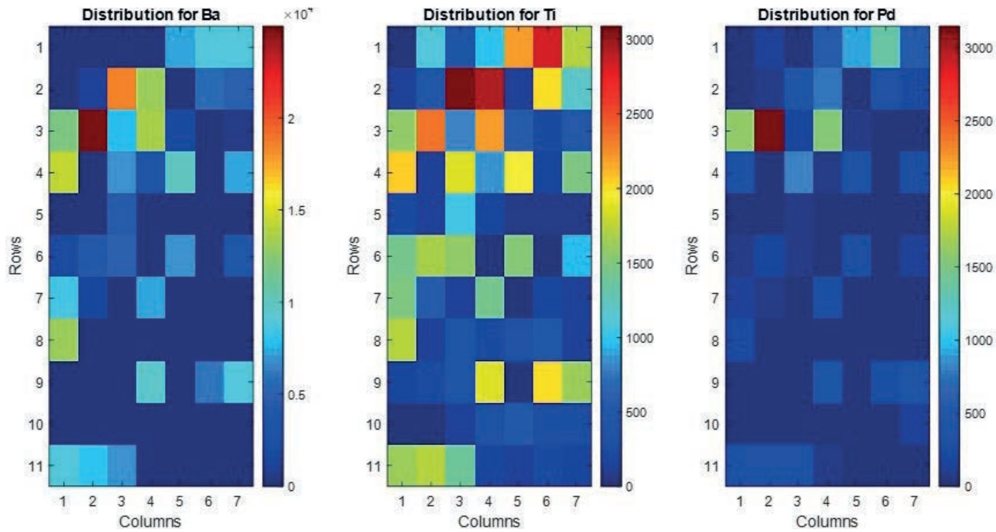

FIGURE 5: Distribution of Ba, Ti and Pd on the PCB with individual concentration acquired by ICP OES.

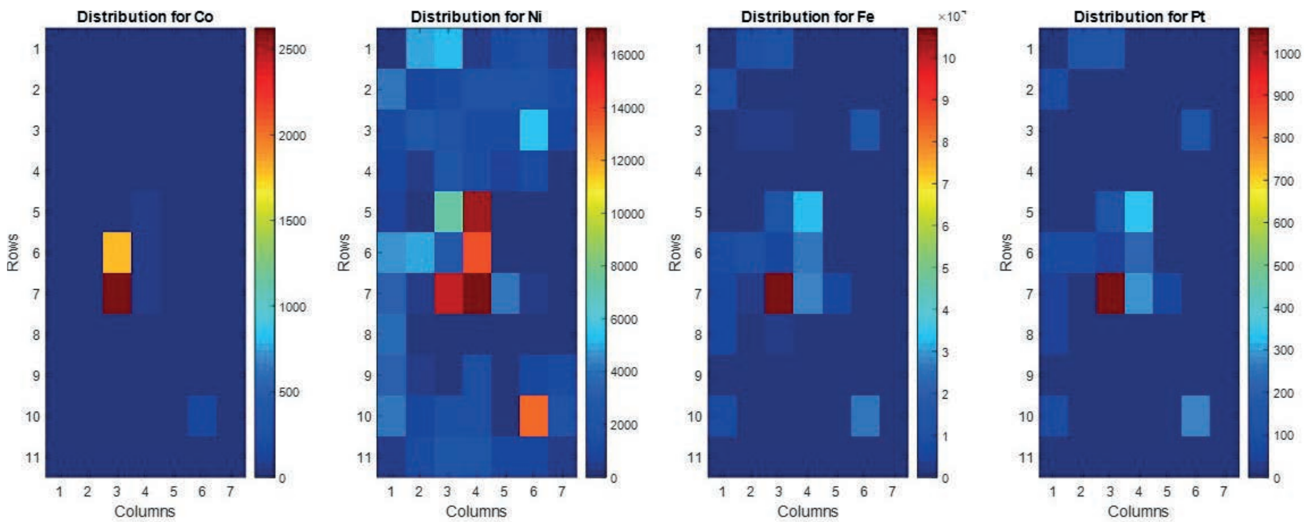

FIGURE 6: Distribution of Co, Ni, Fe and Pt on the PCB with individual concentration acquired by ICP OES.

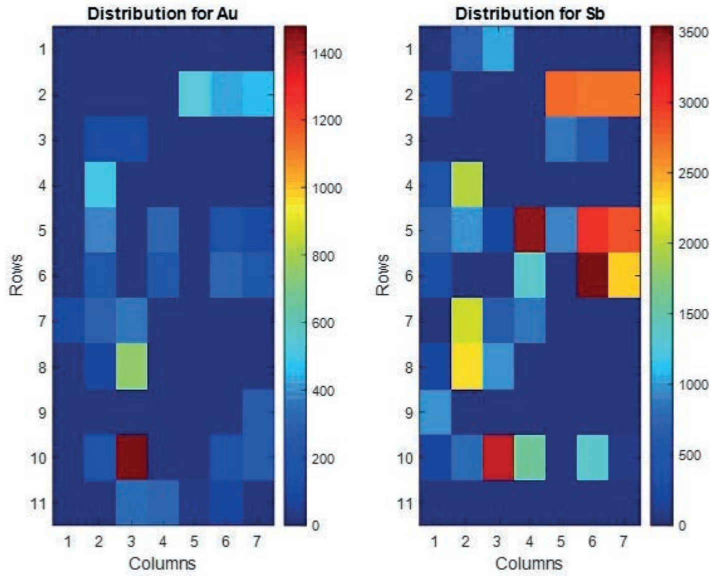

FIGURE 7: Distribution of Au and Sb on the PCB with individual concentration acquired by ICP OE.

process may avoid material loss, improving the detectability, and figures of merit of the proposed method. In addition, the use of several images contributed to better interpretation of ICP OES results. Each figure complements the other, while scores map and loading plot show the main correlation among elements, correlation plot presents how closely correlated these elements are and, individual distribution shows how is this correlation around the PCB components. Therefore, it is a promising way to visualize and interpret the data, contributing to urban mining and recycling.

ACKNOWLEDGMENTS

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DRYING OF REFUSE-DERIVED FUEL (RDF) USING SOLAR TUNNEL DRYER INTEGRATED WITH FLAT-PLATE SOLAR COLLECTOR: AN EXPERIMENTAL APPROACH

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ABSTRACT

Solar energy can be effectively used for drying, especially when the cost of fuel used for conventional dryers is high. The drying of refuse-derived fuel (RDF) is a complicated process involving mass and heat transfer and depends on external variables such as humidity, temperature, and air velocity as well as internal variables, e.g., material surface, physical structure, chemical composition, size, and shape. Herein, a solar tunnel dryer integrated with a flat-plate solar collector was designed and fabricated at the Faculty of Engineering, Suez Canal University, Egypt. The experiments were performed during July 2019. The weather characteristics as well as the variation in moisture content with time were recorded continually. The results indicated that the maximum efficiencies of the solar collector were 46.6%, 53.4%, and 40.0% on 2, 4, and 6 July, 2019, respectively. Moreover, on these days, the moisture content of RDF decreased from 35.6% to 9.6%, from 28.3% to 8.5%, and from 43.5% to 14.0%. The dryer efficiency varied over 14.1-29.5% depending on the drying air stream temperature. The maximum dryer efficiencies recorded were 23.8%, 29.5%, and 25.8% on 2, 4, and 6 July, 2019, respectively.

1. INTRODUCTION

Drying is one of the most important applications of solar energy. One of the oldest practices is the drying of food in the sun. Heat is generated in the interior and surface of the product, which enhances heat transfer. The absorption of solar radiation by the product is the main principle of direct solar drying, Ekechukwu, O. and Norton, B., (1999). In indirect solar drying, the products are spread in thin layers and exposed directly to the sun's radiation. This method is the least expensive and is employed worldwide, but there is significant risk of deterioration owing to infestation and dust accumulation, Fadhel, A., et al. (2005). To avoid these problems, solar dryers should be used on a large scale for drying products, especially in the food industry. There are three main categories of solar dryers: direct, indirect, and infrared radiant heat dryers. Solar dryers have been used in several different environmental engineering applications such as refuse-derived fuel (RDF) drying, municipal solid waste (MSW) drying, and sludge dewatering, He, P., (2009).

Presently, the most common and widely used MSW drying methods include bio-stabilisation, bio-drying, thermal drying, and solar drying, Ferreira, A., et al. (2014). Refuse-derived fuel refers to the materials derived from MSW through many processes such as sorting, screening, separation at source, blending, and pelletising, Asadi, F., (2016). Compared with coal, RDF, as a fuel, has lower calorific value and S content and higher amounts of Cl. Many factors should be considered in the use of RDF, such as moisture content, alkaline and S compounds, calorific value, and ash content, Beyene, H., et al. (2018). Furthermore, RDF can be utilised to generate thermal energy in combustion, gasification, and pyrolysis. The most important problem faced by industries currently is the high water content of all solid fuels, which must be decreased to improve their fuel heating values, as reported in the literature, [Luca, A. and Raffaello, C., (2015), Golisz, E., (2013) and Yuan, J. et al. (2017)].

Organic waste from food factories, lunchrooms, and kitchens has high moisture content and high biodegradability. To reuse such waste in industries, its moisture content should be reduced and controlled as soon as possible to obtain low ash content. At low water content, the risk of spontaneous combustion increases and the energy efficiency of shredding such waste enhances. However, at excessive water content, the relaxation coefficient is high, Koser, H., et al. (1982).

Hence, the control process and presence of moisture content significantly impact the calorific values of fuels,

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e.g., humidity must not exceed 15% in a cement kiln, Yasuhara, A., et al. (2010). However, the low moisture content of RDF renders it difficult to handle during transportation owing to dust and blockages on conveyor belts. Given the low intrinsic market value of RDF, its collection and transport logistics are considerably important to facilitate adequate quantities that allow economical disposal of waste [Caputo, A., and Pelagagge, P., (2002), Caputo, A., et al. (2004) and Peng Lu. et al. (2017)]. Many scientists and organisations have carried out several studies on RDF drying methods. Some of these studies focus on product types and pre-treatment methods, as discussed by Mumba, J. (1996) and Prasad, J., et al. (2006). Jannot, Y. and Coulibaly, Y., (1998) studied the effects of climate factors such as humidity and drying air temperature on the drying process and solar dryer performance.

The greatest disadvantage of the drying process is its high energy consumption; therefore, solar dryers are considered the most economical alternative. Several studies have been carried out on different drying techniques using solar energy. Kabeel, A., (1998) studied the shape optimisation of the absorber plates of solar air collectors and determined that the shape absorber factor played a vital role in the design of solar air heaters and that the optimum tilt angle of the triangular collectors was in the range 50–60°. Kabeel, A., et al. (2017) presented a detailed review of the design, applications, configurations, and improved methods of solar air heaters. They concluded that artificial roughness can be used to enhance the heat transfer rate and thermo-hydraulic performance of solar air heaters.

Additionally, the use of phase-change materials as heat storage materials was found to improve the efficiency of solar air heaters. Abdullah, A., et al. (2018) presented an experimental study to evaluate the performance of a new double-pass solar air heater, with and without turbulators, at different values of air mass flow rate in the range 0.02–0.05 kg/s. They achieved a maximum daily efficiency of 68% using a staggered double-pass solar air heater at an air mass flow rate of 0.05 kg/s.

Kabeel, A., et al. (2018) studied the performance of a baffled glazed-bladed entrance solar air heater and compared the results with those of a conventional equipment under the same operating conditions. Their results showed that the daily efficiency of the solar air heater considered with 800 baffles was 29.91–51.69% higher than that of the conventional one. Zamrudy, W., et al. (2019) reviewed different RDF drying technologies for subsequent use of RDF in the cement industry. It can be concluded from their review that the main drying equipment for RDF include a conveyor belt dryer, Torbed reactor, screw conveyor, indirect rotary dryer, and direct rotary drum dryer.

Herein, a solar tunnel dryer integrated with a flat-plate solar air collector was designed, fabricated, and tested for drying RDF. The experimental data were used to investigate the performance of the solar dryer and evaluate the flat-plate solar collector efficiency and drying efficiency. The results obtained were validated and compared with previously reported findings for solar air collectors. Furthermore, the results were evaluated and compared with those obtained for the direct exposure of RDF to the sun under the same climatic and operating conditions.

2. MATERIALS AND METHODS

2.1 Experimental Set-up and Procedure

The experimental set-up was designed, installed, and tested at the Faculty of Engineering, Suez Canal University, Ismailia (30°36' N, 32°16' E), Egypt. The set-up consisted of a solar tunnel dryer integrated with a piping system, a flat-plate solar air collector (dimensions of 1.5 × 1 × 0.15 m), solar panels (eight panels of 250 W each), an inverter with a controller, a centrifugal blower, and a fan. The solar tunnel dryer was constructed from materials readily available in the local area and had a metallic frame structure with dimensions of 2 × 1.5 × 1 m. The piping system consisted of two main pipes with a diameter of 2 inches. Small-holed pipes, 0.75 inches in diameter, were welded to the two main pipes to diffuse hot air into the solar tunnel dryer.

The performance of the dryer was evaluated in July 2019. The layout with a schematic diagram of the experimental set-up is shown in Figure 1. A transparent plastic cover was used to cover the tunnel dryer to transmit sunlight into and prevent air leakage from the dryer. An insulating material was used to reduce conductive losses from the bottom. A small fan was installed on the ceiling of the solar tunnel dryer to circulate air when the specific humidity increased. Such an equipment can dry different types of materials and the drying air temperature should be maintained 10-30°C higher than the ambient temperature, as suggested in the literature (Elicin, A., and Sacilik K., (2005), Kooli, S., et al. (2007), Hossain, M., and Bala, B.,(2007) and Usub, T., et al. (2008)).

The air inside the solar tunnel dryer must be maintained at a high temperature to increase the driving force for evaporation. Moreover, the relative humidity must be maintained far from the saturation point to increase the water-vapor-carrying capacity of air and to avoid condensation inside the tunnel dryer. Therefore, ventilation with a certain degree of control was required in the solar tunnel dryer. A personal computer processor fan was placed on the ceiling to change the air feeding rate inside the greenhouse at high values of internal humidity. This fan had eight blades and a diameter of 0.15 m. It was operated for 3 min every 2 h. The solar tunnel dryer was connected to a data acquisition system to automatically record the temperatures using resistance temperature detectors (RTDs). The ambient temperature and the temperatures at the collector outlet and inside the solar tunnel dryer were measured.

The solar radiation intensity and wind velocity were additionally measured on the days of the experiment. Figure 2 shows a photograph of the complete set-up used to obtain the experimental results. The RDF, with a particle size of 60-80 mm, was spread on the drying floor and placed inside the solar tunnel dryer. More than 20 test runs were conducted. The selected days were 2, 4, 6, and 8 July 2019, based on their similar weather characteristics. Each test run was started at 9:00 AM and continued until 5:00 PM and the measured parameters were recorded instantaneously every 30 min. The samples were distributed in layers to receive the same amount of solar radiation. The results of

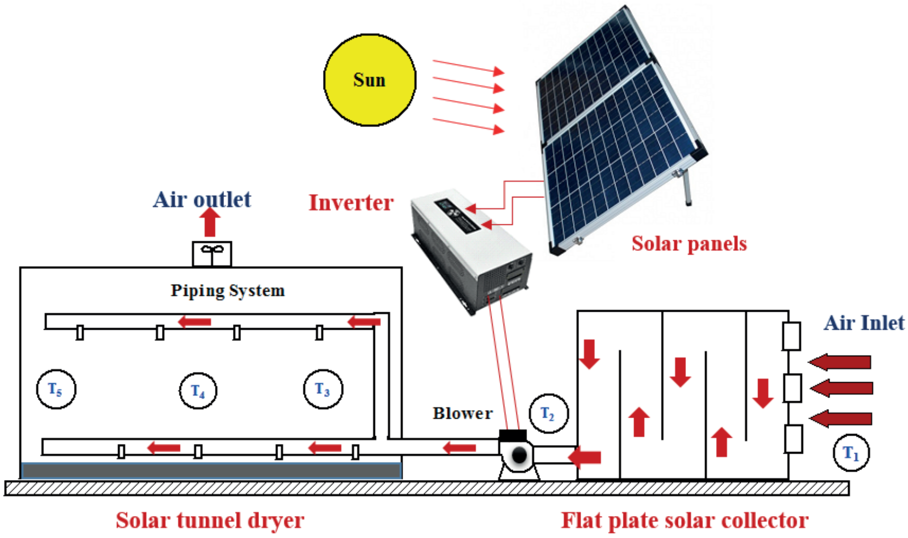

FIGURE 1: Schematic diagram of experimental set-up.

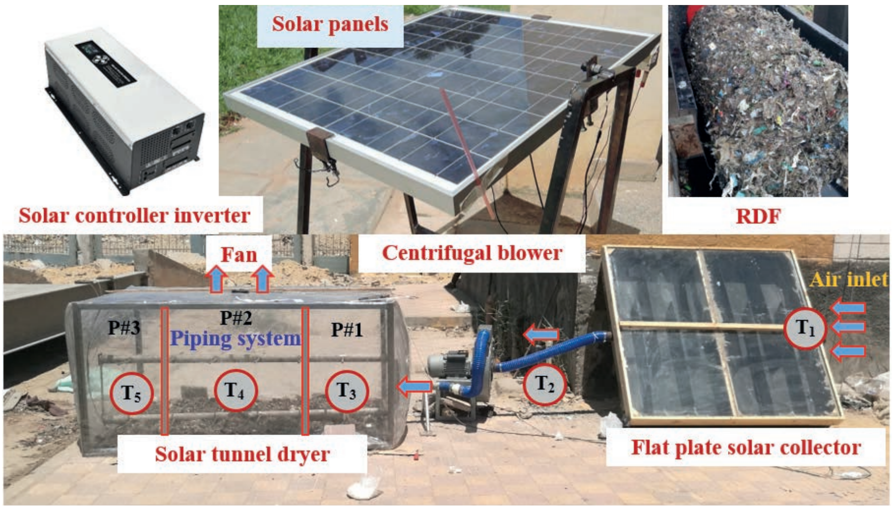

FIGURE 2: Photograph of experimental set-up.

natural drying, wherein RDF was placed in the sun for direct exposure, were additionally obtained and compared. At the beginning of the experiment, the RDF was mixed to ensure homogeneity.

All the experiments were repeated with the same quantity of RDF, which was pre-dried in the instrument and re-wetted to ensure that the apparatus had the ability to dry RDF under various conditions. The initial weight of the sample and ambient conditions were measured. The different climate parameters and RDF moisture content were recorded every hour. The measurements were repeated until the RDF inside the solar tunnel dryer exhibited no change in moisture content.

2.2 Measurements and Instruments

This section presents the instruments used to measure different parameters such as solar radiation intensity, temperature inside the solar tunnel dryer, wind velocity, moisture content, and ambient temperature. Solar radiation intensity is the basic parameter for the drying process. Solar panels (also known as photovoltaic panels) were used to convert sunlight to electricity, which was used to power the centrifugal blower and the fan in the ceiling of the solar tunnel dryer. The inverter was used to convert direct current to alternating current to operate the centrifugal blower. The flat-plate solar collector was used to absorb the incoming solar radiation, convert it to heat, and transfer this heat to the ambient air at the inlet of the collector.

The centrifugal blower was used to draw the ambient air into the flat-plate solar collector. The air started to heat and reached the solar tunnel dryer at a high temperature. In this work, an Eppley PSP pyranometer was used to measure the global solar radiation on the days when the experiment was conducted. This instrument measures the global solar radiation (in W/m^2) incident from all directions. The apparatus was mounted on a table and adjusted horizontally using adjustable levelling screws and a built-in water balance and was connected to a digital output screen to record the values, as shown in Figure 3-a. An advanced platinum RTD temperature sensor was used in the experimental set-up to measure the temperatures at different positions.

Two sensors were used to measure the air temperature at the inlet (T_1) and outlet (T_2) of the solar collector. The RTD temperature sensors were fixed inside the solar dryer at three different positions, i.e., P#1 (T_3), P#2 (T_4), and P#3 (T_5), as shown in Figure 2. These sensors have a data logger for collecting the data measured on the days when the experiment was conducted. The temperature sensors had a measurement range from -50°C to +350°C with a resolution of 0.1°C.

Wind velocity is another parameter affecting the solar dryer performance. On each day of the experiment, the wind velocity was measured every 30 min. A digital advanced anemometer, type Xplorer4, was used to measure the wind velocity. The measurement range of the apparatus was 0-42 m/s. The average values of wind velocity were calculated for each day.

The moisture content was determined using the drying oven and balance apparatus shown in Figure 3-c. The RDF samples were heated and their weight loss owing to moisture evaporation was recorded simultaneously every hour.

2.3 Error Analysis

There are many different types of errors and uncertainties in experiments. These can originate from the selection, calibration, and condition of the instruments used, observation environment, test planning, and reading of measurements. In the RDF drying experiments, the weights and temperatures were measured with the appropriate instruments. The Holman, J., (1994), method was used to estimate the uncertainty in the experimental results. The uncertainty was calculated based on the minimum error, i.e., the ratio of the least count of the apparatus to the minimum output of measured value, Kabeel, A., (2009). Suppose a set of measurements, namely, X_1, X_2, ..., is used to measure 'n' number of experimental variables. These measurements are used to calculate some desired results (R) of the experiment. Thus,

$$R = R(X_1, X_2, X_3, \dots \dots \dots \dots, X_n) \quad (1)$$

Let W_R be the uncertainty in the result, R, and W_1, W_2, W_3, ..., W_n be the uncertainties in the independent variables. The uncertainty is calculated according to the following equation.

$$W_R = \left[\left(\frac{\partial R}{\partial X_1} W_1\right)^2 + \left(\frac{\partial R}{\partial X_2} W_2\right)^2 + \dots + \left(\frac{\partial R}{\partial X_n} W_n\right)^2 \right]^{\frac{1}{2}} \quad (2)$$

If the relationship between the measured parameters and the result, R, is known and the uncertainties in the measurement of each quantity are further known, then the error or uncertainty of the result, W_R, is calculated according to Equation (2). The measured values are very small compared with the data obtained. Table 1 shows the uncertainty values of the measured parameters. It was found that all the uncertainty values were within the allowable range.

2.4 Properties of RDF

The RDF fed to the solar tunnel dryer was obtained from the municipal waste treatment plant of ECARU Company located in the 15th of May City, Egypt. Its properties are listed in Table 2.

2.5 Solar Collector Efficiency

Several types of solar collectors are used in engineering applications. Flat-plate solar collectors are most commonly used to heat the air inside dryers. The efficiency of a solar collector depends on many parameters such as air mass flow rate, solar radiation intensity, and inlet and outlet air temperatures of the collector. The efficiency of the flat-plate solar collector can be calculated according to Sencan, A., and Ozdemir, G., (2007) as follows:

$$\eta = \frac{Q_u}{A_c \times I} \quad (3)$$

where Q_u is the useful energy transferred to the drying air, A_c is the collector area, η is the solar collector thermal efficiency, and I is the intensity of solar radiation.

2.6 Drying Efficiency Calculation

The drying efficiency is defined as the ratio of the energy required to evaporate the moisture in the RDF sample

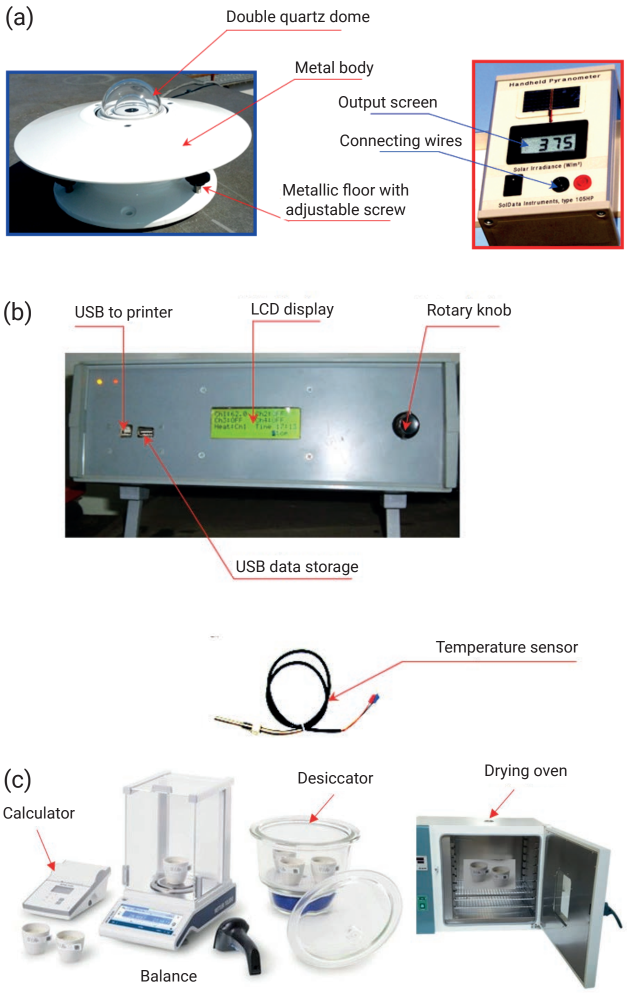

FIGURE 3: Photographs of measuring instruments. (a) Eppley PSP pyranometer with output screen. (b) Data acquisition and RTD temperature sensor. (c) Drying oven and balance apparatus.

TABLE 1: Uncertainties in parameters measured during drying process.

Measuring instrument	Uncertainty
Mercury thermometer	0.050
Eppley pyranometer	0.002
RTD temperature sensor	0.003
Anemometer	0.050
Electronic balance	0.001

to the energy supplied to the solar tunnel dryer. It was assumed that the heat loss from the dryer to the ambient air was negligible and that heat was utilised to increase the product temperature and evaporate the moisture content of the product. The drying efficiency in a certain time period was calculated using the following expression, Kassem, A., et al. (2011):

$$\eta_t = \frac{(W_w \times L + m_p \times C_p \times \Delta T)}{A \times I_t \times t_h} \quad (4)$$

The latent heat of vaporisation is usually expressed as a function of drying air temperature. Hence, in this study, the latent heat of vaporisation (J/kg) was calculated in accordance with ASAE, (1998) as follows:

$$L = 2502535.259 - 2385.764(T_d - 273.16),\ 273.16 \le T_d \le 338.72 \quad (5)$$

where η_t is the drying efficiency (%), A is the surface area of the air heater (m^2), t_h is the desired time period, W_w is the evaporated water (kg), m_p is the mass of the RDF sample at the end of this time period (kg), C_p is the specific heat capacity of RDF, which depends on the RDF composition and was assumed to be constant, as reported by Savage, G.,1989, ΔT is the temperature difference between ambient air and air inside the solar tunnel dryer (°C), I_t is the global solar radiation on a horizontal surface (W/m^2), and T_d is the drying air temperature inside the solar tunnel dryer (K).

TABLE 2: Properties of RDF.

No.	Parameter	Value	Unit
1	Total Cl	Max. 0.7	%
2	Cd + Hg + Tl	Max. 7	ppm SS
3	As	Max. 9	ppm SS
4	Cr	Max. 40	ppm SS
5	Cu soluble	Max. 300	ppm SS
6	Mn	Max. 400	ppm SS
7	Ni	Max. 40	ppm ss
8	Pb	Max. 200	ppm SS
9	S	Max. 0.6	T.q.
10	Size	Max. 30 × 30	mm^2
11	Bulk density	60-80	kg/m^3

SS: suspended solids; T.q.: threshold quantity.

3. RESULTS AND DISCUSSION

Figures 4, 5, and 6 present the variations in the ambient air, internal, and RDF temperatures as well as that of solar radiation intensity as functions of time on the days when the experiment was conducted in July 2019. The hourly solar radiation intensity started to increase from 9:00 AM (local time) and peaked at noon, thereafter decreasing until the end of the day, as shown in Figures 4-6. In addition, the maximum recorded values of solar radiation intensity were 930, 990, and 965 W/m^2 on 2, 4, and 6 July 2019, respectively. Furthermore, a strong relationship between the solar radiation intensity and ambient temperature was observed with their variation curves exhibiting the same trend.

The temperature inside the solar tunnel was much higher than the RDF temperature. The temperature difference started to increase in the morning, peaked at noon, and decreased thereafter until the end of the day. Increasing the air stream temperature endowed it with additional

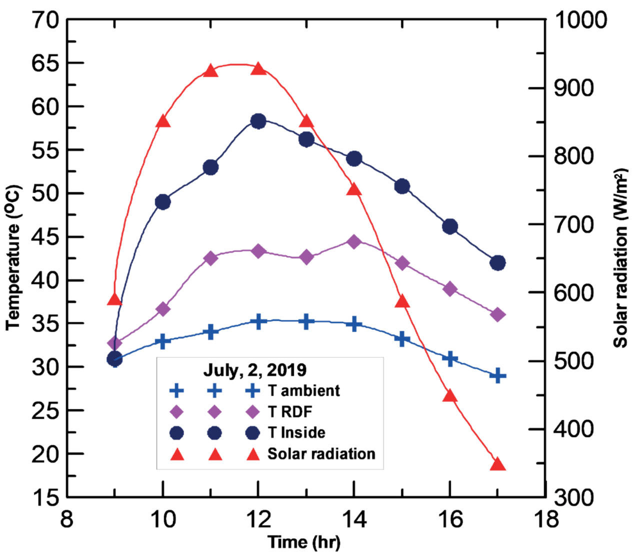

FIGURE 4: Variations in ambient, inside, and RDF temperatures and solar radiation intensity with time on 2 July 2019.

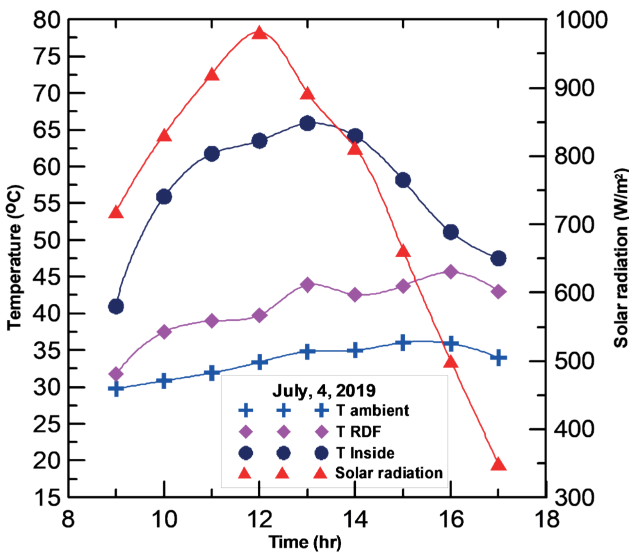

FIGURE 5: Variations in ambient, inside, and RDF temperatures and solar radiation intensity with time on 4 July 2019.

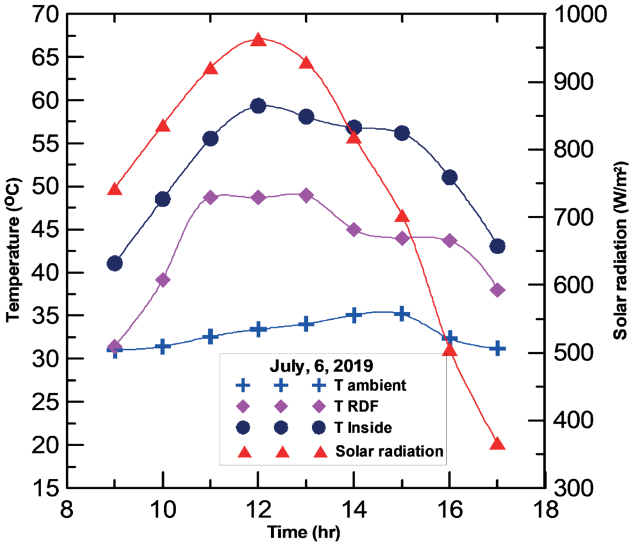

FIGURE 6: Variations in ambient, inside, and RDF temperatures and solar radiation intensity with time on 6 July 2019.

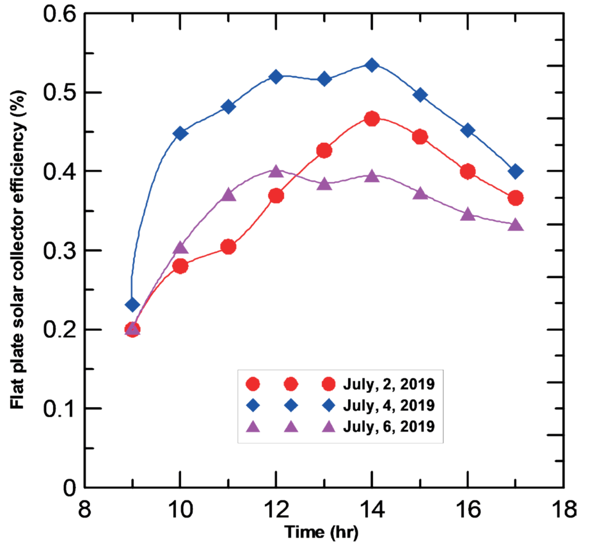

FIGURE 7: Variations in flat-plate solar collector efficiency with time on three days of when the experiment was conducted in July 2019 at air mass flow rate of 0.02 kg/s.

evaporative power, which resulted in more effective drying in a shorter duration. This indicated that the drying process should be conducted at a lower airflow rate and higher drying air stream temperature. The maximum temperatures recorded inside the solar dryer were 58.3, 65.8 and 59.0°C on 2, 4, and 6 July 2019, respectively, and the maximum RDF temperatures on these days were 44.4, 45.6, and 48.9°C, respectively.

It is known that wind velocity has a significant effect on the drying process, dryer efficiency, and flat-plate solar collector efficiency. The wind velocity on the days when the experiment was conducted was measured and recorded every hour using the anemometer. The average recorded values of wind velocity were 2.5, 3.0, and 3.9 m/s on 2, 4, and 6 July 2019, respectively.

Figure 7 shows the variation in the flat-plate solar collector efficiency with local time during the days when the experiment was conducted in July 2019. The variation in the flat-plate solar collector efficiency was similar to that of the solar radiation intensity, which indicated that the proposed experimental set-up as well as its design and manufacture were error-free.

The results obtained on all three days were similar, as depicted in Figure 7. The maximum recorded values of the flat-plate solar collector efficiency were 46.6%, 53.4%, and 40.0% on 2, 4, and 6 July 2019, respectively, at an air mass flow rate of 0.02 kg/s. To verify the accuracy of these results, they should be validated based on previously published findings. At an air mass flow rate of 0.02 kg/s, Abdullah et al. achieved maximum efficiencies of approximately 58.2% and 38.5% using a staggered double-pass and staggered single-pass solar air heaters, respectively. The maximum daily efficiency observed in their work was 68% at an air mass flow rate of 0.05 kg/s using the staggered double-pass solar air heater. Kabeel et al. [22] used a conventional solar air heater at an air mass flow rate of 0.022 kg/s and achieved a maximum daily efficiency of approximately 32.12%. Thus, the results obtained in the present work were similar to previously published findings.

The variations in the RDF moisture content with time at an air mass flow rate of 0.02 kg/s on 2, 4, and 6 July 2019 are shown in Figure 8. The moisture content removal depends on the drying air temperature. In the present work, the increase in the air stream temperature owing to the airflow generated in the solar tunnel dryer was sufficient for the purpose of drying the RDF. Figure 8 shows that the final RDF moisture content was approximately constant at the end of the day, and this was especially the case on 4 July 2019. The moisture content of RDF decreased from 35.6% to 9.6%, from 28.3% to 8.5%, and from 43.5% to 14.0% on 2, 4, and 6 July 2019, respectively. The quantities of water removed on 2, 4, and 6 July 2019 were 26.0%, 19.8%, and 29.5%, respectively.

To study the power of the solar tunnel dryer, an identical quantity of RDF with the same moisture content was placed to dry in the sun for the same drying time on the days when the experiment was conducted. The results obtained using the solar tunnel dryer were compared with those of drying the samples in the sun.

Figure 9 shows the variations in the RDF moisture content versus time for the samples in the sun on the three days when the experiment was conducted in July 2019. The moisture content of the RDF samples placed in the sun decreased from 35.6% to 20.0%, from 28.3% to 12.0%, and from 43.5% to 29.0% on 2, 4, and 6 July 2019, respectively. The quantities of water removed on 2, 4, and 6 July 2019 in the same drying time were 15.6%, 16.3%, and 14.5%, respectively.

The above results showed that in terms of the quantity of water removed, the RDF drying process in the solar tunnel dryer showed improvements of 40.0%, 17.68%, and 50.85% over that in the sun on 2, 4, and 6 July 2019, respectively. Additionally, it was clear that the drying rate as well as the initial and final moisture contents of RDF played important and effective roles in the drying process and this

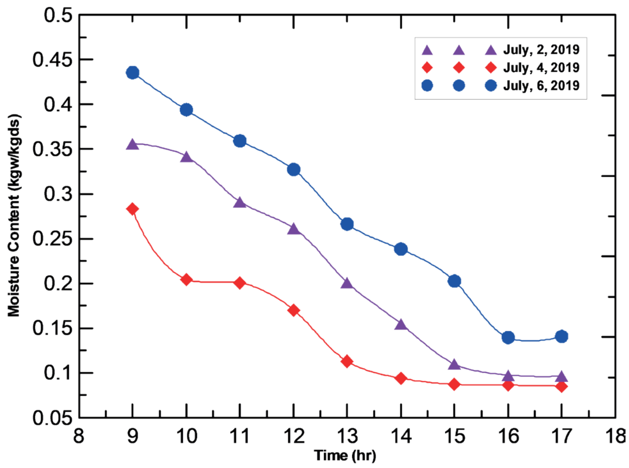

FIGURE 8: Variations in RDF moisture content with time on three days when the experiment was conducted in July 2019 at air mass flow rate of 0.02 kg/s.

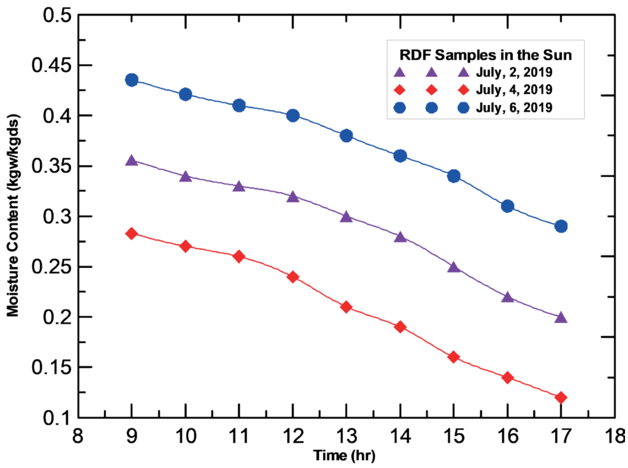

FIGURE 9: Variations in RDF moisture content with time for samples dried in the sun on three days when the experiment was conducted in July 2019.

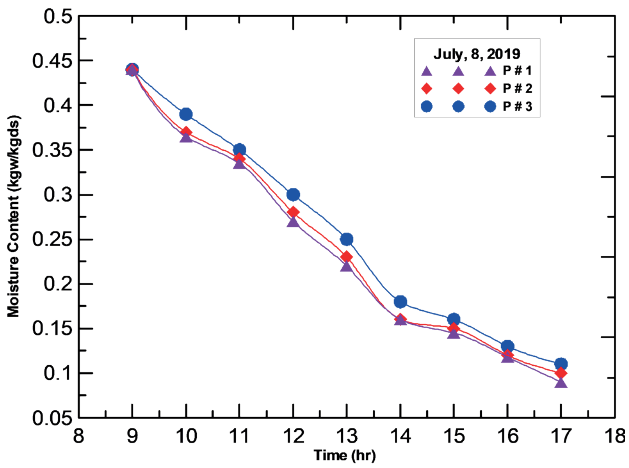

FIGURE 10: Variations in RDF moisture content with time at three different positions on 8 July 2019.

was the main reason for the different values obtained in the two processes.

To evaluate the effect of the drying air stream temperature on the drying process, the solar tunnel dryer was divided into three sections: first (P#1), middle (P#2), and last (P#3). The moisture contents in the three sections were measured and recorded on 8 July 2019. The variations in the RDF moisture content with time in the three different sections are shown in Figure 10. The highest RDF drying rate was observed in P#1, followed by those in P#2 and P#3. This indicated that the drying air stream temperature had an important effect on the drying process. The drying rate of RDF decreased with increase in the length of the piping system owing to the heat transfer losses throughout the system.

Furthermore, the small difference between the three sections in terms of moisture content enabled the piping system to distribute the drying air at a constant rate throughout the solar tunnel dryer.

Figure 11 shows the variations in the dryer efficiency with time on the days when the experiment was conducted in July 2019 at an air mass flow rate of 0.02 kg/s. The dryer efficiency and flat-plate solar collector efficiency exhibited similar variation trends and both varied with the solar radiation intensity. The increase in dryer efficiency may have been owing to the increase in the drying air temperature, which enhanced the evaporative power of air and decreased the drying time. The dryer efficiency varied in the range 14.1-29.5% depending on the drying air stream temperature. The maximum recorded dryer efficiencies were 23.8%, 29.5%, and 25.8% on 2, 4, and 6 July 2019, respectively.

4. CONCLUSIONS

In this study, an experimental investigation was conducted for drying RDF using a solar tunnel dryer. The dryer was manufactured using iron bars and other materials available locally. The climate conditions were measured on

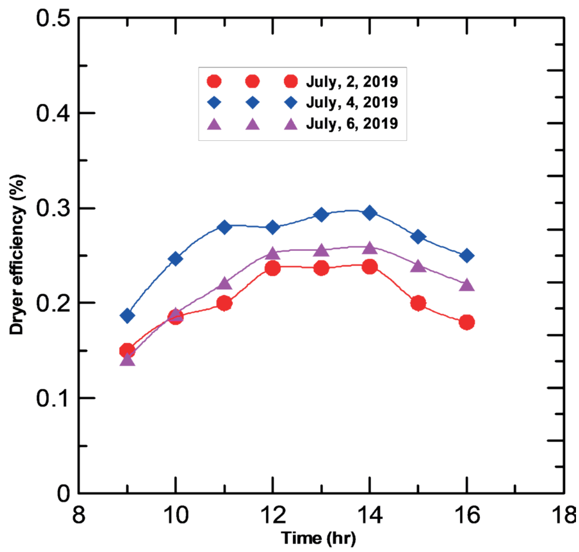

FIGURE 11: Variation in dryer efficiency with local time on three days when the experiment was conducted in July 2019 at air mass flow rate of 0.02 kg/s.

the days when the experiment was conducted and plotted as functions of time. A flat-plate solar collector was used to heat the air stream entering the solar tunnel dryer. The results showed that the drying air stream had an important effect on the drying process and that the RDF moisture content decreased during the day. The flat-plate solar collector efficiency and dryer efficiency were evaluated and calculated according to the experimental results and validated using previously published findings. The maximum recorded values of flat-plate solar collector efficiency were 46.6%, 53.4%, and 40.0% on 2, 4, and 6 July 2019, respectively. The moisture content of RDF decreased from 35.6% to 9.6%, from 28.3% to 8.5%, and from 43.5% to 14.0% on 2, 4, and 6 July 2019, respectively. Finally the dryer efficiency varied in the range 14.1-29.5% depending on the drying air stream temperature, and its maximum recorded values were 23.8%, 29.5%, and 25.8% on 2, 4, and 6 July 2019, respectively.

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CISA

FINAL QUALITY OF A SUSTAINABLE LANDFILL AND POST-CLOSURE MANAGEMENT

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ABSTRACT

Landfill should be designed and constructed in line with the principle of environmental sustainability, guaranteeing over a period of less than one generation (typically considered 30 years) the environmental equilibrium of Final Storage Quality, when waste stability and immobilisation of contaminants is achieved and all active measures of control may be removed without posing any further risk to the environment. The practical definition of FSQ, when a landfill can be released from aftercare, and a procedure for the technical and administrative termination of the post-closure management phase are an evident regulatory strategic need to assure the design of sustainable landfill. The aim of this paper is to provide a criterion to define the FSQ of landfill, based on the control of the analytical emission parameters and of stability indexes related to the residual emission potential. These should concur with the law's acceptance criteria for the landfilling of wastes (LAC), and with the legal limit values (LLV) established for the emission of contaminants into the environment. The interrelationship between, stability indexes (emission potential), analytical emission parameters, LAC and LLV is discussed and reference values are provided. Finally, the paper proposes a procedure for the termination of technical and administrative aspects following the post-closure management in accordance with FSQ.

1. INTRODUCTION

Despite the lack of a conventional definition for a sustainable landfill, in a scientific context a landfill is considered such if the emissions released do not modify substantially the quality of the surrounding environmental matrixes: air, water and ground (Hjelmar & Hansen, 2005; Stegmann et al., 2003).

This concept is closely linked to Final Storage Quality (FSQ). The term "final storage" was first used during the mid-1980s by Baccini, Henseler and other researchers from the Swiss landfill working group (Belevi & Baccini, 1989) in referring to the quality achieved by emissions and wastes at the time in which all active measures of control may be removed without posing any further risk to the environment. This condition of equilibrium should be reached within the time frame of one generation, commonly taken as a period of 30 years, in order to not "compromise the possibility for future generations to meet their needs" (WECD, 1987).

The European directive 1999/31/EC on waste landfilling established that to obtain a landfill permit from the authorities "adequate provisions, by way of a financial security or any other equivalent, on the basis of modalities to be decided by Member States, has been or will be made by the applicant prior to the commencement of disposal operation" (Article 8, point iv). The provisions laid down by the European directive have been implemented to varying degrees in national legislation (as an example, in Italy, landfill operators are required to provide financial provisions for at least 30 years following landfill closure). However, no criteria have been established relating to long-term emission control, with the exception of the placing of physical barriers, the efficacy of which, can be ensured merely for as long as the barriers themselves last. The contamination period of a landfill (intended as the time frame within which emissions may produce negative effects on the environment) however considerably exceeds the lifetime of physical barriers.

A stand-alone time-restricted financial provision would not appear to represent an adequate tool to ensure achievement of FSQ (Fourie, 2003). Moreover, the comprehensive attribution of responsibility to the operator with regard to the period in which the landfill may potentially elicit environmental issues is somewhat unrealistic as this period could be prolonged into centuries, extending beyond the timespan not only of the management company, but also of the relevant authorities (Gronow, 2014). Landfill design therefore should be addressed bearing in mind the role it

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undertakes in a Circular Economy based on the following criteria:

a. A long-term perspective should be taken into account.
b. The limited duration of physical barriers should be addressed and measures taken to extend their lifespan for as long as possible.
c. Limits established by law relating to the emission of contaminants from liquid wastes (including sludges and digestates), from products deriving from recycling (compost, building materials) and the emission of gases into the environment should be taken into account.
d. The time frame required to achieve FSQ should be monitored and reduced to less than 30 years.
e. Legislation should be passed to include a Table of Minimum Objective values (TMO) to be reached prior to post-closure management of the landfill, establishing parameters and values based on scientific knowledge and reliable evidence.
f. Site-specific situations of environmental vulnerability should be considered, where necessary lowering the objective values or raising the degree of protection of individual barriers in the multi-barrier system.
g. All due caution should be exercised in the case of uncertainty when addressing all unsubstantiated aspects (wastes, technologies, etc.).
h. Mono-waste landfills should be preferred and classified based on the homogeneity of long-term behaviour of the specific wastes rather than on classification for inert, hazardous or non-hazardous wastes: e.g., landfills for municipal wastes, landfills for prevalently inorganic wastes, etc.

Accordingly, there is an evident strategic need for the definition of a reference framework for the Final Storage Quality of a landfill.

On the international scenario, the sole legislations to date to adopt a Table of Objective values relating to the achievement of Final Storage Quality by a sustainable landfill was issued by the Lombardy Regional Authorities, Italy (Guidelines for a sustainable landfill design and management, Regional Decree dated October 7th 2014, n. X, 246); while in Germany, several studies put forward FSQ values for use by the German Federal Environment Ministry (Stegmann, et al., 2006, Stegmann, et al., 2003, Stegmann, et al., 2011).

The aim of this paper is to provide a criterion to define the FSQ of landfill, based on the control of the analytical emission parameters and of stability indexes related to the residual emission potential. These should concur with the law's acceptance criteria for the landfilling of wastes (LAC), and with the legal limit values (LLV) established for the emission of contaminants into the environment (into surface waters, onto the soil, onto agricultural land as compost, domestic sludges, digestate, etc., or associated with building products realised with recycled residues). The interrelationship between, stability indexes (emission potential), analytical emission parameters, LAC and LLV is discussed and reference values are provided. Finally, the paper proposes a procedure for the termination of technical and administrative aspects following the post-closure management in accordance with FSQ.

2. CONTROL OF THE RESIDUAL EMISSION POTENTIAL OF WASTES AND FINAL STORAGE QUALITY OF THE LANDFILL

Potential waste contaminants are essentially present in either a mobilizable (s_s) or non-mobilizable form (x_s). The mobilizable fractions contained in landfilled wastes are converted into a non mobile solid form or they are transformed and pass from one phase to another, in line with their characteristics of degradability and leachability, accumulating in the emissions generated (leachate and gas), and potentially polluting the environment. Thus, the mobilizable fraction represents the emission potential of landfill (S_t).

The composition of both raw and pre-treated landfilled wastes varies over time, in particular the ratio between mobile and immobile contaminant fractions, reaching a peak in emission potential (S_{max}) during the operation phase.

Figure 1 graphically illustrates the variation in emission potential of the wastes (associated with the residual mobile fraction of contaminants contained in landfilled wastes) based on the type of treatment undertaken (pre-treatment, in situ treatment carried out during landfill operations or post-closure). The graph is of purely indicative value.

The combination of pre-treatments and in situ treatments should be selected with the aim to achieve an environmentally sustainable value of residual emission potential (S_{sust}), - i.e. such to not disturb the environmental equilibrium (FSQ) - within the time frame of one generation, or however within the time frame covered by the financial provisions provided by the management company (30 years).

RQ represents Rock Quality, when waste is completely mineralized and the emission potential is negligible ($S_s \approx 0$). This condition is achieved over a variable time frame (diagenetic time frame).

To achieve FSQ, a series of quality objectives should be envisaged by the landfill design and management plan. These objectives should be established, in the same way as in numerous other areas of environmental protection, by defining a Table of Minimum Objectives (TMO).

When a landfill is sited on a vulnerable area (i.e. featuring a lower capacity of self-depuration), the local Authorities may pose more stringent values on the project in line with the degree of self-depuration of the environment (TPO – Table of Project Objectives). However, the TPO objectives may also come closer with TMO objectives when the designer includes an increased requirement of physical barriers (versus those established by law). Essentially, in a vulnerable area the landfill designer may intervene either on the quality of wastes (by lowering S_s, S_G, S_L) to achieve more restrictive project objectives (TPO) or on the quality of physical barriers to achieve project objectives resembling those established for TMO. The interrelationship between analytical emission parameters, stability indexes (emission potential), self-depuration capacity, addition of physical barriers (versus those provided for by law) and

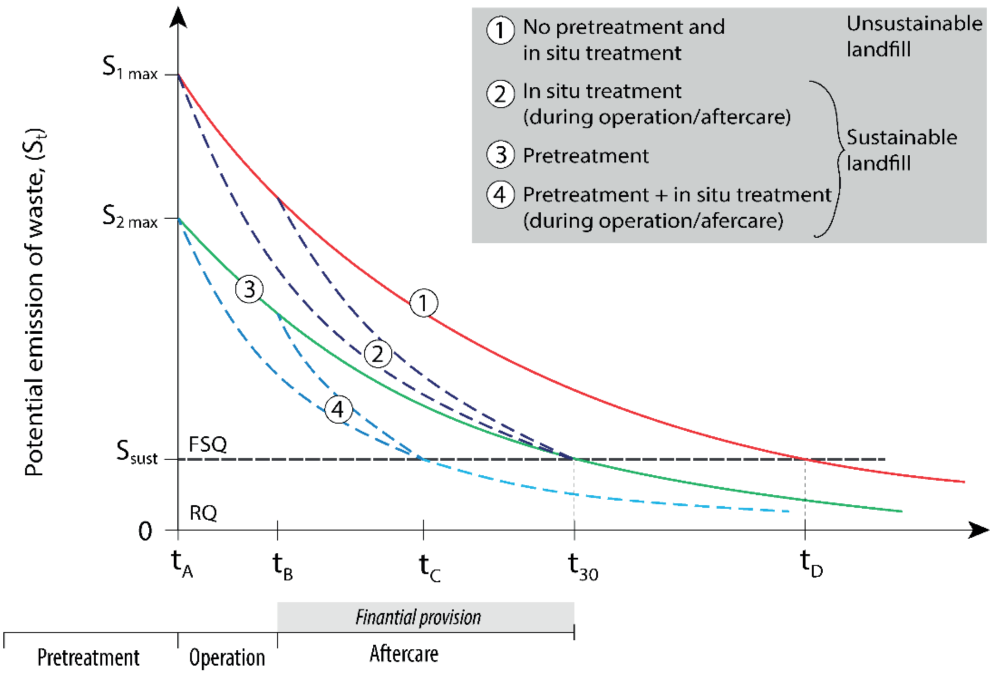

FIGURE 1: Time trend of the emission potential for release of contaminants from a landfill. FSQ= Final Storage Quality, according to which the emission potential reaches a value of S_{sust} in equilibrium with the environment; RQ, Rock Quality, according to which waste is completely mineralized and the emission potential S_s=0; t_A, peak in emission potential during landfill operations; t_B, Closure of landfill operations and commencement of aftercare; t_C, Anticipated achievement of FSQ; t_{30}, Sustainability target and termination of financial provisions; t_D, FSQ beyond the 30-year threshold (unsustainable landfill) (modified from Cossu et al., 2020c).

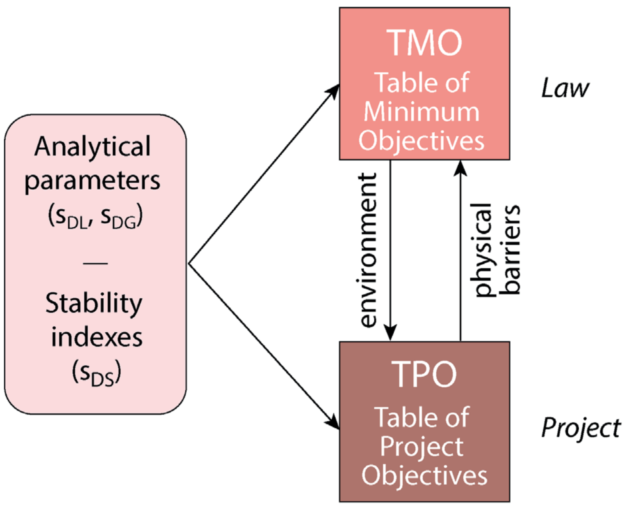

FIGURE 2: Relationship between parameters and stability indexes, self-depuration capacity of the environment and quality/efficiency of physical barriers used in addition to those established in the Table of Objective Project values (TPO), which may be more restrictive or equal to those provided for in the Table of Minimal Objective values established by law. s_{DS}, s_{DG}, s_{DL} = residual concentration of contaminants present in wastes, gas and leachate, respectively.

TMO and TPO is reported in Figure 2.

The parameters considered in the Tables of Objectives (analytical parameters and stability indexes), to be discussed in detail in subsequent paragraphs, are identical to those used to assess waste acceptance criteria (AC – Acceptance Criteria for the Landfilling of Wastes defined by Law), and should also be similar to those established by law for other sectors of environmental protection (Legal Limit Values– LLV). For example, they should concur with the parameters established for the emission of contaminants into surface waters and onto the soil, onto agricultural land (compost, domestic sludges, digestate, etc.) or associated with building products (realised with recycled residues).

Indeed, all operations implying the deposition of wastes on the land are liable to the release of emissions affecting the quality of the environment and therefore, in the same way as landfills, the potential impact produced over time and compliance with Final Quality should be considered, as an uncontrolled release of contaminants, even at low concentrations, could result in an increase of diffuse contamination.

A series of cases of deposition on the land are graphically represented in Figure 3.

The interrelationship and scale of values for the diverse waste quality parameters are graphically illustrated in Figure 4, in which LAC represents the legally established quality parameters as acceptance criteria for the landfilling of wastes, PAC represents the waste acceptance values adopted by the designer, which may be either identical to or more restrictive than LAC values if more stringent pre-treatments than those established by law are applied, TMO and TPO refer to the previously described minimum and project objectives to be achieved in order to terminate the post-closure period of the landfill in line with FSQ, and LLV the legal limit values established for the emission of contaminants into the environment referred to previously.

Essentially, a similar strategy to that applied in the treat-

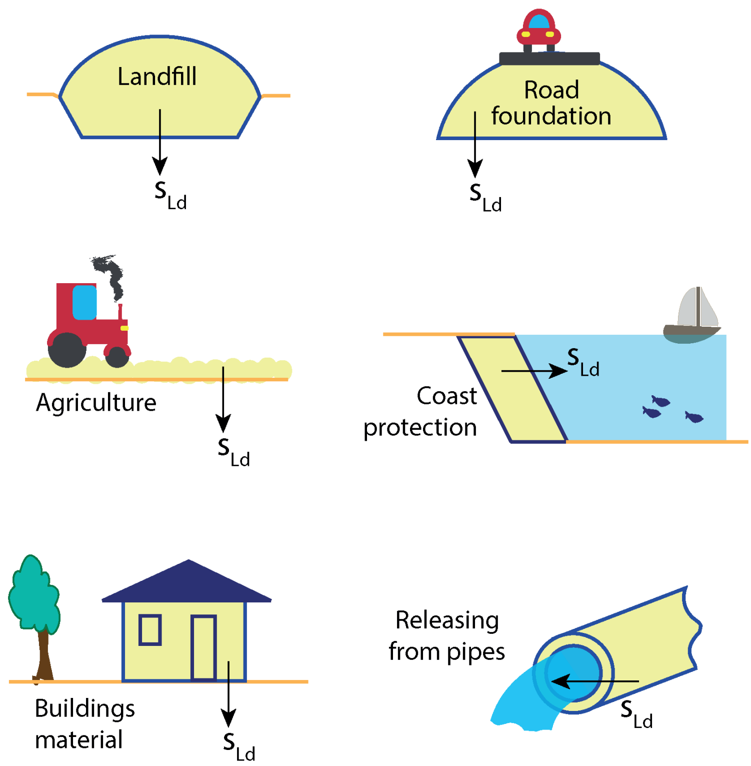

FIGURE 3: Graphical representation of the potential release of contaminants from different forms of waste depositing on the land (Modified from Scharff, 2014; Cossu et al., 2020). s_{Ld}, concentration of mobile contaminants dispersed through leaching.

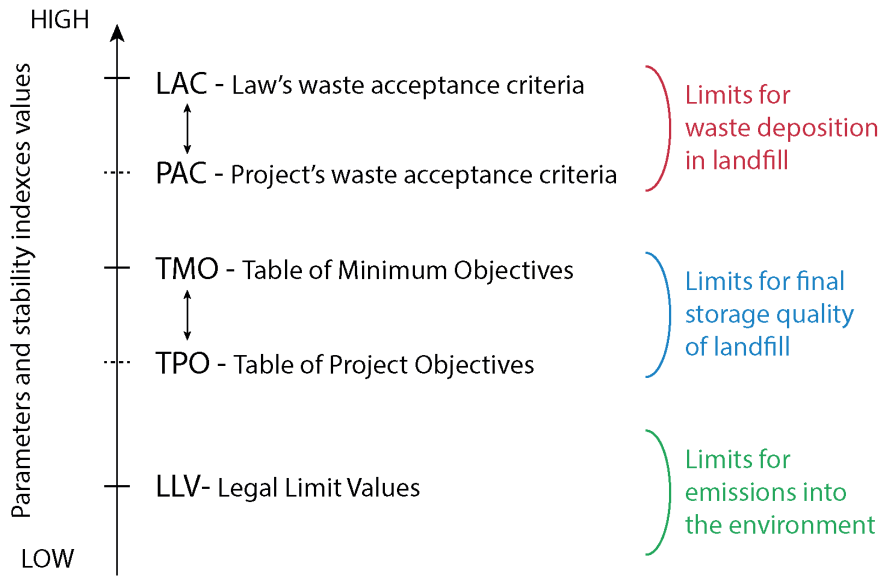

FIGURE 4: Interrelationship and scale of values for parameters and stability indexes that contribute towards defining a series of waste quality objectives.

ment of wastewaters, for example, should be adopted (Figure 5).

Should the need arise to dispose of industrial wastewaters containing 1500 mg/L Total Nitrogen (TN) in a wastewater treatment plant, industry would be required to pre-treat the wastewater to comply with acceptance limits for sewerage systems (LAC, Legislative Decree 152, 2006). While the depuration plant would be dimensioned in order to meet the discharge legal requirements (TMO). However, should the area in which the treated effluent is released be a vulnerable area (water body featuring a poor exchange subject to eutrophication), i.e. the Venetian lagoon, lower concentrations should be present on release, and the plant should be designed, built and managed with the aim of achieving this objective (TPO).

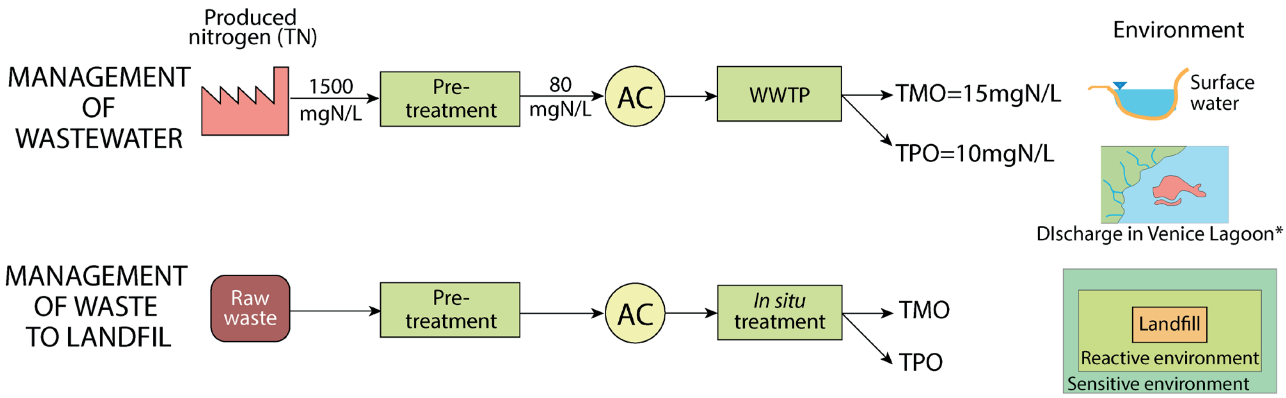

FIGURE 5: Graph parallel between the management of sewage forwarded to a depuration plant and management of wastes forwarded to landfill. WWTP, Wastewater treatment plant. *Limits for discharge into the Venice lagoon laid down in Ministerial Decree 30/7/99.

3. EMISSION PARAMETERS

Gas and leachate emissions that form inside a landfill are the result of complex biochemical, geochemical and chemical-physical reactions and interactions between the three phases co-existing heterogeneously in the waste mass of a landfill. The emissions quality may be characterised by a series of different analytical and operational parameters taken to describe Landfill Quality and changes over time (Heyer and Stegmann, 2018; Cossu and Pivato, 2018).

3.1 Biogas

Parameters used to characterise biogas are:

- Percentage composition (%): CH_4%, CO_2%, O_2%.
- Hourly production of biogas (Nm^3/h): this may be measured by means of extraction tests in the presence of a gas collection system, or using a standardised model of biogas production (Andreottola et al., 2018);
- Specific production of biogas (l/m^2h): this parameter may be used to estimate surface emission of biogas and the eventual oxidative capacity of the top cover; measurements may be taken from static or dynamic surface accumulation chambers and subsequent instrumental analysis of the accumulated gas.

Amongst the above parameters, the percentage composition of biogas may not be significant in assessing the degree of stability reached by wastes due to the fact that the concentration of putrescible organic substances affects the quantity of gas generated but has a negligible influence on quality. The latter is linked solely to the degradation stage reached and potential infiltration of air. With regard to safety, the accumulation of low gas loads may determine the same risks as higher gas loads (fire, explosions, plant asphyxiation).

It might also be of interest to assess the $(CH_4+CO_2)/N_2$ ratio: indeed, the latter yields useful information for the characterisation of biogas quality in both aerobic and anaerobic environments. The ratio is associated with nitrogen which, contrary to oxygen, is not consumed during degradation reactions. This parameter may prove to be of use in assessing the degree of stability reached by wastes and verifying air ingress into the landfill body.

3.2 Leachate

To assess leachate quality, the following parameters are commonly analysed:

- Chemical Oxygen Demand, COD (mgO_2/L);
- Biochemical Oxygen Demand over 5 days, BOD_5 (mgO_2/L);
- Total Organic Carbon, TOC (mg/L). (As this parameter can be analysed both on liquid and solid matrix, it is normally used for carbon mass balance);
- Dissolved Organic Carbon, DOC (mg/L): the fraction of TOC dissolved in water defined as the fraction capable of passing through a 0.7-0.22 filter. As discussed in detail subsequently, DOC has been adopted by the EU as one of the most important parameters to be considered when establishing waste acceptance criteria for landfilling;
- Heavy metals (mg/L);
- Other substances, such as chlorides, sulphates, and fluorides are also taken into account when assessing acceptability;
- Total Kjeldahl Nitrogen, TKN (mgN/L);
- Ammonium Nitrogen, $N\text{-}NH_4^+$ (mgN/L): under anaerobic conditions ammonium cannot be oxidised;
- Nitric and nitrous nitrogen, $N\text{-}NO_2^-$, $N\text{-}NO_3^-$ (mgN/L): the oxidised forms of nitrogen are usually only present in leachate from aerobic or semi-aerobic landfills.

In addition to concentrations of the above-mentioned substances, the ratios between the diverse parameters are highly significant, both in terms of assessment of FSQ and in ascertaining environmental situations featuring the presence of contaminants with different origin. For example:

- BOD_5/COD yields information relating to the residual biodegradability of leachate and an eventual contamination of surface waters or water tables. Generally, as biological stabilisation progresses in the landfill, a rapid decrease in in this ratio is observed due to a rapid uptake of BOD_5, whilst COD rarely reaches values lower than 500-1000 mgO_2/L due to the presence of non-degradable humic substances.
- NH_4/TKN provides an indication of the state of hydrolysis of organic nitrogen and ammonium.

- NH_4/Cl^-. As these substances are both conservative in an anaerobic environment, this ratio allows the origin of ammonium to be determined in the presence of two sources of emission (e.g. leachate from an MSW landfill and runoff from agricultural land fertilised with urea).
- Nitrites and nitrates/TKN. The ratio between the oxidised form of nitrogen and Kjeldahl nitrogen attests to the type of biological processes (aerobic or anaerobic) present within the waste mass.

4. PARAMETERS FOR USE IN MEASURING SOLID WASTE QUALITY AND ASSESSING EMISSION POTENTIAL

Waste quality may be defined based on the following parameters:

- Concentration (expressed as % or mg/kg). This is usually employed to measure the overall quantity of a given substance present in waste either in a mobile (s_s) or non mobile form (x_s), using analytical techniques that vary in line with the substance to be measured.
- Emission potential. This assesses, both directly and indirectly, the amount of contaminant (mobilizable fraction) present in a waste. It may be measured by means of tests providing a standardized, controlled reproduction of the processes which mobilise the contaminant. Tests most frequently applied in the case of wastes destined for landfilling are aimed at assessing the biological degradation of organic substances and leaching.

Cyclical tests facilitate the monitoring over time of the evolution of emission potential based on decay curves such as those represented in Figure 1.

The aim to minimise emission potential and promote long-term stabilisation should therefore be defined on the basis of proven scientific knowledge where possible, or by means of experimental trials when dealing with specific wastes and contaminants for which appropriate information is lacking. A similar procedure is adhered to in the treatment of wastewaters. The performance of biological depuration process applied to domestic wastewaters are widely acknowledged and standardized, whilst wastewaters of a different origin may require a pilot study to be carried out in order to define the necessary design parameters.

4.1 Tests applied in measuring the degree of biodegradation

These tests may be conducted by measuring parameters indicative of biodegradation trend, i.e.:

- O_2, consumed under aerobic conditions;
- Gas, produced under anaerobic conditions;
- CO_2, produced under aerobic or anaerobic conditions;
- Ratio between parameters in eluate from leaching tests.

Tests may therefore be used to assess both the presence of biodegradable substances in the wastes (indirectly) and thier biological stability.

The most commonly used indexes are respirometric and gas production indexes.

The respirometric index measures oxygen consumed by a unit of weight of waste or biomass (expressed as Total Solids –TS or Volatile Solids –VS) over a specific time frame. The test may be carried out under static or dynamic conditions (Cossu et al., 2001; Cossu & Raga, 2008). Static respirometric indexes (RI_4, RI_7) measure oxygen consumed over a 4- or 7-day period and are usually expressed in terms of mg O_2/g solids (TS or VS). Dynamic respirometric indexes (DRI) measure the oxygen consumed hourly by a stabilised air flow that crosses the wastes, expressed as mgO_2/kgVS/h.

GB_{21}, used to measure gas production over a 21-day period, is widely used in the presence of biogas. It may be expressed in terms of NL (gas volume under normal conditions, 0°C, 1 atm) per kg of TS or VS.

The above-mentioned indexes, although very popular, feature a series of drawbacks:

- High cost of equipment (both static and dynamic);
- Low significance of results in the presence of toxic or inhibitory substances (that slow down oxygen consumption) or inert volatile solids (such as paper or plastic) which, being at the denominator, tend to underestimate the index (Cossu et al., 2012).

To overcome these drawbacks (Cossu et al., 2012; Cossu et al., 2017), use of the BOD_5/COD ratio measured on the eluate of a simple leaching test has been proposed (several leaching test conditions have been tested, varying testing mode -static or dynamic-, contact time, liquid to solid ratio and pretreatment).

Figure 6 illustrates the correlation between several diverse indexes as resulting from a series of experimental studies.

The index yields two types of information: the presence of organic substances (COD) and degree of biological stabilisation (BOD_5/COD). Should a finding of null BOD_5 be detected in the presence of a high COD, this would indicate the need to assess the presence of toxic or inhibitory compounds. This index is also advantageous in that measurements are not affected by the presence of paper and plastic and is suitable for use in measuring both fine and coarse material.

Biological stability indexes are indicated for use at different times throughout the life of a landfill for a series of reasons: following mechanical-biological pre-treatment to verify process efficiency prior to waste deposition, to check compliance with waste acceptance criteria, and on termination of the post-management phase to verify achievement of FSQ. The parameters and units of measure used may vary in line with the purposes for which they are applied. As an example, the BOD_5/ COD ratio is highly effective in assessing degree of waste stabilisation following pre-treatment or on termination of the post-management phase, whilst RI_4, measured in mgO_2/gTS, provides a reliable indication of the putrescible content of wastes accepted for landfilling. However, BOD_5/ COD should be measured even when using RI_4 to assess the potential presence of inhibitory factors.

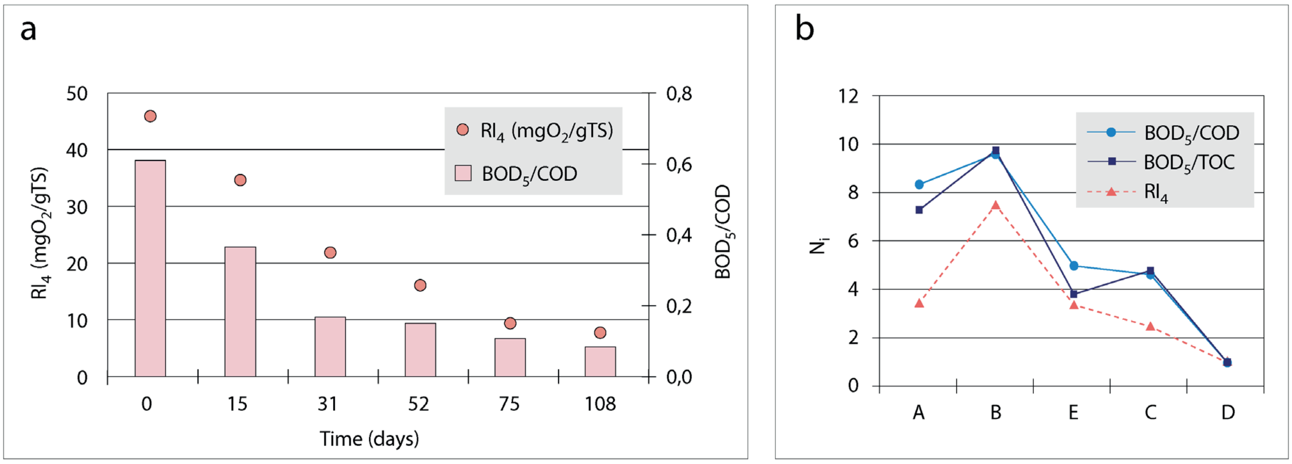

FIGURE 6: Correlation between respirometric index RI4 and BOD5/COD ratio. a. Variation of the two indexes in a lysimeter-conducted stabilisation process of pre-treated municipal wastes (modified from Cossu & Raga, 2014). b. Correlation between normalised values of different biodegradation indexes for various types of waste: A & B = mechanically pre-treated undifferentiated wastes; C = wastes following Mechanical-Biological Treatment; D = compost from anaerobic digestate; E = dried domestic sludge. Ni= Normalised values based on waste stability index for wastes D, taking D as unitary D, Ni = Stability index measured for waste i / Stability index measured for waste D. BOD/COD and BOD/TOC indexes were measured following static leaching test having a time of contact between liquid and solids of 2 h, without shaking; (modified from Cossu et al., 2017).

Although this issue is widely debated throughout the scientific community worldwide, very few countries have included in national legislation indexes for use in assessing the stability of landfilled wastes, as illustrated in Table 1 that provides updated information on this topic.

The European Union has established a limit value for COD, whilst leaving it up to each single nation to deliberate on additional criteria and limits.

German and Austria adopt a 4-day static index (indicated as AT_4), the United Kingdom applies the 4-day dynamic respirometric index (DRI_4) and potential production of methane at 100 days (BMP_{100}), whilst in Italy, the Italian National Institute for Environmental Research and Protection (ISPRA) has adopted the dynamic index.

Generally speaking, the other European countries tend to adopt the COD limit, although at times exceptions are made for municipal wastes.

In several countries such as China and Spain, where no additional checks are carried out, working groups have been set up with the aim of defining possible criteria for future use, although currently limits have only been established by a restricted number of local authorities.

TABLE 1: Verification of the biological stability of wastes prior to landfilling and criteria per assessment applied in different countries.

Country	Verification	Index	Limit value	Reference
Australia	NO	-	-	(Clarke, 2020)
Austria	YES	RI_4, GB_{21}	(7 mg O_2/g TS, 20 Nl/kg TS) [a]	(Binner, 2020)
China	NO	RI_4 [b]	20 mg O_2/g TS	(He, 2020)
Colombia	NO	-	-	(Gandini, 2020)
EU	YES	COD	80 mg/L	(EU- Landfill Directive 31/99)
France	NO [c]	-	-	(Hennebert, 2020)
Germany	YES	RI_4, GB_{21}	5 mg O_2/g TS, 20 Nl/kg TS	(Ritzkowski, 2020)
Japan	NO	-	-	(Ishii, 2020)
Greece	NO [c]	-	-	(Komilis, 2020)
Italy	YES	DRI	1000 g O_2/kg VS*h	(DM 27 set. 2010)
Holland	NO [c]	-	-	(van der Sloot, 2020)
UK	NO	DRI_4, BMP_{100}	-	(Knox, 2020)
Spain	NO [c]	-	-	(Sanchez, 2020)
Sweden	NO [c]	-	-	(Kumpiene, 2020)
USA	NO	-	-	(Thorneloe, 2020)

a. Both limits must be complied with
b. Adopted only on a local level in Shanghai
c. No supplementary indexes used in addition to those indicated by the European directives

4.2 Leaching tests

Leaching tests have been extensively studied and standardized for use in a series of diverse applications both in Europe and the US (van der Sloot et al., 2018). In particular, the European Union has widely endorsed use of these tests in defining waste acceptance criteria for landfilling.

To ensure the appropriateness of leaching tests to act as a technical reference, they should be standardized and officially recognised.

The following variables should be considered with regard to standardization:

- Liquid/solid ratio (L/S);
- Type of contact between liquid and solids (static or dynamic, batch or in leaching columns);
- pH (fixed or variable);
- Quality of leached liquid;
- Duration of contact between liquid and solids;
- Physical conditions of the solids (granular, monolithic);
- Other control parameters (redox potential, complexation capacity, etc.).

Aims of the test: to assess release over time or variation in control parameters (Figure 7).

Whilst taking the above into account, a plethora of standards exist to be used for a wide variety of purposes, thus resulting in the European Committee for Standardization issuing waste characterisation guidelines (CEN TC-292) to facilitate navigation amongst the diverse standards (EN 12920), (van der Sloot et al., 2014).

Without wishing to comment on which test performs best in identifying a Leaching Index to be used in defining a Table of Objectives (TMO) to enhance achievement of FSQ, it should be highlighted how this problem is identical to that presented in the context of landfilling of any other product and/or residue (see Figure 3) or in defining the End of Waste for construction and demolition wastes. Therefore, if a standard is established for these situations, a similar standard should be defined also for FSQ.

5. CRITERIA FOR THE DEFINITION OF OBJECTIVE VALUES RELATING TO FINAL STORAGE QUALITY

The criteria used to define a Table of Minimum Objectives to act as a reference guideline in landfill design should include the following:

- Analogy with parameters of waste characterisation for the purpose of identifying acceptance criteria for landfilling.
- Analogy with parameters and values defined for “End of Waste” products used on soil, i.e. recycled material in road foundation, use of digestate in agriculture, etc.
- Analogy with parameters and limits established for the discharge of wastewaters into water bodies and of gas effluents into the atmosphere.
- Acknowledgement that an increasing level of pollution constitutes one of the major macro-environmental issues and asking questions such as: “Is it preferable to have industrial wastes recycled in bitumen of numerous roads or to deposit them in a sustainable landfill?”
- “End of waste qualification” and “Termination of the post-management phase of a landfill” are two faces of the same coin.
- Limits of concentration for leachate contaminants to be established for TMO should take into account the limited duration of physical barriers.

For the sake of rapid comparison, Table 2 lists the quality objectives established with a view to monitoring environmental impact for use in the measurement of different parameters on a range of liquid (leachate, eluates from leaching tests, wastewaters) and solid matrixes. Intentionally, no Minimum Objective Values (TMO) are proposed for the design of a sustainable landfill as this should be subjected to an institutional assessment conducted ad hoc by a working group set up for this purpose. As a mere example, the TMOs adopted in the Sustainable Landfill Guidelines issued by the Lombardy Regional Authorities in 2014 and the objective values suggested by Stegmann et al. (2006) are reported.

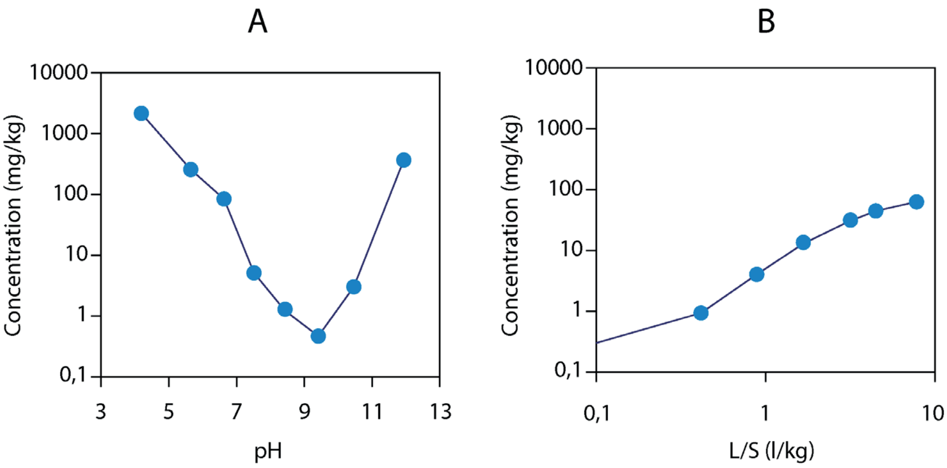

FIGURE 7: Variation of contaminant release in leaching tests according to changes in control parameters (A) or time span (measured in terms of L/S) (B) (modified from van der Sloot et al., 2014)..

TABLE 2: Reference parameters for quality analysis of liquid and solid matrixes provided by European and Italian law for the monitoring of contaminant release into the environment. Objective values established with regard to Final Storage Quality (FSQ) by the Lombardy Regional Authorities and proposed by Stegmann et al. (2006) are also reported.

	AC – Acceptance criteria			LLV – Legal limit values							TMO - Table of Minimum Objectives	
Parameters	European waste acceptance criteria in Landfill –Decision 2003/33/EC			Italian Limit Values for discharging Liquid wastes - Decree n. 152/06			Italian contamination thresholds for underground waters [b] - D.Lgs n. 152/06	Italian contamination thresholds for ground and underground – Decree n. 152/06 (A) agricultural use (B) commercial use/ industrial use	European limits for organic fertilisers[c] and Compost (C) Digestate (D); - Regulation (EC) 2019/1009	Waste recovery[d] - Decree 5/4/2006	Objective values Lombardy region - Reg. Decree 7/10/2014- n.X/2461)	Objective values proposed in Germany (Stegmann et al., 2006)
	Inerts	Non-hazardous (with SNR)[a]	Hazardous	Into surface waters	Into sewers	Onto soil						
Liquid Matrix (mg/L)	EN 12457/ 2&4[e]	EN 12457/ 2&4[e]	EN 12457/ 2&4[e]	Wastewater	Wastewater	Wastewater	Eluate UNI EN 12457-2			Eluate UNI EN 12457-2	Leachate	Leachate
Al	-	-	-	1	2	1	0.2			-	1	-
As	0.05	0.2	2.5	0.5	0.5	0.05	0.01			0.05	0.5	0.1
Ba	2	10	30	20	-	10	-			1	-	-
B	-	-	-	2	4	0.5	1			-	2	-
Be	-	-	-	-	-	0.1	0.004			0.01	-	-
Cd	0.004	0.1	0.5	0.02	0.02	pd	0.005			0.005	0.02	0.05
Co	-	-	-	-	-	-	0.05			0.25	-	-
Total Cr	0.05	1	7	2	4	1	0.05			0.05	2	-
CrVI	-	-	-	0.2	0.2	pd	0.005			-	0.2	0.1
Cu	0.2	5	10	0.1	0.4	0.1	1			0.05	1	1
Fe	-	-	-	2	4	2	0.2			-	2	-
Hg	0.001	0.02	0.2	0.005	0.005	pd	0.001			0.001	0.005	0.005
Mn	-	-	-	2	4	0.2	0.05				2	-
Ni	0.04	1	4	2	4	0.2	0.02			0.01	2	0.2
Pb	0.05	1	5	0.2	0.3	0.1	0.01			0.05	0.2	0.4
Sb	0.006	0.07	0.5	-	-	-	0.005			-		-
Se	0.01	0.05	0.7	0.03	0.03	0.002	0.01			0.01	0.03	-
Sn	-	-	-	10	-	3	-			-	10	-
V	-	-	-	-	-	0.1	-			0.25	-	-
Zn	0.4	5	20	0.5	1	0.5	3			3	3	2
Total cyanide	-	-	-	0.5	1	pd	0.05 (free CN)			0.05	0.5	0.1
Chloride	80	1.500	2500	1200	1200	200	-			100	-	-
Fluoride	1	15	50	6	12	1	1.5			1.5	6	25
Sulphate (SO_4)	100	2000	5000	1000	1000	500	0.25			250	1000	-
Sulphites (SO_3)	-	-	-	1	2	0.5	-			-	1	-
Sulphide (H_2S)	-	-	-	1	2	0.5	-			-	-	-
Phenol	0.1	-	-	0.5	1	0.1	<0.0005-0.18[f]			-	0.5	0.5
BOD_5	-	-	-	40	250	20	-			-	-	-
COD	-	-	-	160	500	100	-			30	1500	-
BOD_5/COD										-	0.1	-
TOC	-	-	-	-	-	-	-	-	-	-	-	150
Total phosphorous	-	-	-	10	10	2	-			-	-	-
$N\text{-}NH_4$	-	-	-	15	30	-	-			-	50	50
$N\text{-}NO_3$	-	-	-	20	30	-	-			50	20	-
DOC	50	80	100	-	-	-	-			-		-

Total Hydrocarbons	-	-	-	5	10	pd	0.35			-	5	-
Aldehydes	-	-	-	1	2	0.5	-				-	-
Aromatic organic solvents	-	-	-	0.2	0.4	0.01	0.0001-0.05 [f]			-	0.2	-
Organic nitrogenous solvents	-	-	-	0.1	0.2	0.01	-			-	0,1	-
Total surfactants	-	-	-	2	4	0.5	-			-	-	-
Organo-phosphate pesticides	-	-	-	0.1	0.1	pd	-			-	0.1	-
Total pesticides	-	-	-	0.05	0.05	pd	-			-	0.05	-
Chlorinated solvents	-	-	-	1	2	pd	10^{-6}-0.81 [f]			-	1	-
Solid matrix												
pH	-	>6										
TOC (%)	<3	<5	<6									
BTEX (mg/kg)	6	-	-									
Mineral oil (C10 to C40) (mg/kg)	500	-	-									
PCB (mg/kg)	<1	-	-									
As (mg/kgTS)									40			
Cd (mg/kgTS)								2(A)-15(B)	1.5			
CrVI (mg/kgTS)								2(A)-15(B)	2			
Cu (mg/kgTS)								120(A)-600(B)	300			
Hg (mg/kgTS)								1(A)-5(B)	1			
Ni (mg/kgTS)								120(A)-500(B)	50			
Pb (mg/kgTS)								100(A)-1000(B)	120			
Zn (mg/kgTS)								150(A)-1500(B)	800			
PAH_{16} (mg/kgTS)									6 (D, C)			
RI_4 (mg O_2/gTS)										Eluate UNI EN 12457-2	2	2.5
DRI (mgO2 / kgSV/h)									25 (D,C) (mmol O_2/kg/h)	-	100	
BG_{21} (Nl/kgVS)									250 (D)(Nl/kgVS)	0.05	5 (Nl/kgTS)	10 (Nl/kgTS)

[a] Europe establishes limit values only for non-hazardous waste, which is landfilled in the same cell with stable, non-reactive hazardous waste. Member States may create subcategories of landfills for non-hazardous waste.
[b] When old waste material is found in soil during excavation or remediation operations, the Italian law (DPR 13th June 2017) establishes that this material is not to be considered as waste (but as byproduct) when, (among others) after leaching test, the concentrations in eluate are below the contamination thresholds for underground waters.
[c] The European regulation 2019/1009 establishes the limit values for organic fertilizers, which shall consist solely of specific component materials (e.g. compost and digestate). Maximum limit values for specific parameters (e.g. stabilization indexes) are additionally fixed for component materials.
[d] The Italian Law (DM 5 aprile 2006, n. 186; DM 5 febbraio 1998) establishes the criteria for the identification of the non-hazardous waste to be recovered.
[e] Europe establishes different limits values according with the adopted leaching test (prEN 14405 or EN 12457/1-4). Here only those for EN 12457/2&4 are reported in terms of mg/L.
[f] Range of values fixed for specific compounds.
pd = prohibition on discharging on soil.

6. PROCEDURES FOR TERMINATION OF POST-CLOSURE MANAGEMENT

The procedure proposed for the termination of technical and administrative aspects following the post-closure management is based on verification of the achievement of the project objectives established in the TPO.

During the post-closure management phase in a landfill, the operators should monitor quality evolution over time. Once conditions meeting the TPO objectives have been satisfactorily reached, the operator should liaise with the relevant Authorities to obtain authorisation to undertake the termination procedure (Figure 8). A gas, waste and leachate sampling schedule is established for a series of collection point at varying depths, and all tests and analyses envisaged by the project are carried out.

If the results obtained meet the limits established by the TPO, the procedure is then repeated for a specific number of times (for example, 3 times) at a pre-established time interval (e.g. 1 month). If values are confirmed, the Administrative authorities may authorise termination of the post-closure phase.

However, if values are not confirmed and the discrepancies are only slight, the Authorities may opt to undertake a risk analysis to assess whether residual values, although exceeding those targeted by the TPO, are compatible with the environmental situation with regard to the quality of emissions and level of residual protection afforded by physical barriers.

Should the risk analysis yield a positive outcome, the post-closure period may be terminated. Conversely, the post-closure management phase should continue, awaiting the onset of suitable conditions to repeat the procedure.

7. DECOMMISSIONING

Once the landfill has achieved final storage quality, compliance with the project objectives has been achieved and post-closure management has officially been terminated, the landfill is then ready for Decommissioning. Decommissioning represents the definitive release of the landfill to nature and to its intended end use in the absence of any further human interventions other than those required to promote end use, in line with the definition of FSQ provided by Baccini & Belevi (1989).

Accordingly, no further monitoring should be required with regard to control of emissions, leachate pumping, maintenance of cover systems or leachate drainage.

To enable the above, the topic should be addressed right from the design stage, as an example, by providing for morphologies that envisage build-up of the waste mass above ground level (see Chapter 3, paragraph 3.15), thus facilitating by reasons of gravity, water runoff from the inside of the landfill. Morphologies that develop the waste mass below ground level, although less critical with regard to mechanical stability, may act as a water storage basin, thus determining a potential uncontrolled outflow of liquid. In this case, appropriate systems of controlled water overflow and drainage of emissions should be implemented.

8. CONCLUSIONS

The practical definition of FSQ, when a landfill can be released from aftercare, and a procedure for the technical and administrative termination of the post-closure management phase are an evident regulatory strategic need to assure the design of sustainable landfill. To achieve FSQ, a series of quality objectives should be envisaged by the landfill design and management plan. These objectives should be established by defining a Table of Minimum Objectives (TMO), which should be used as reference in establishing the Table of Project Objectives (TPO). The procedure for the closure of the aftercare should be based on verification of the achievement of the project objectives. The establishing of TMO and TPO, as well as the aftercare closure procedure should be in line the following criteria:

Minimum Objectives fixed by the TMO may coincide with the Project Objectives (TPO), but in case landfill is sited on a vulnerable area, authorities may pose more stringent values on the project, in line with the degree of self-depuration of the environment and with the further physical barriers (versus those established by law) that the designer might include in the project.

TMO and TPO parameters should be analogous with parameters fixed for waste acceptance for landfilling, for “End of Waste” products used on soil, (i.e. recycled material in road foundation, use of digestate in agriculture, etc.) and for the discharge of liquid and gaseous emission into the environment.

Once compliance with the project objectives has been achieved, post-closure management can be terminated. In case limit values are not achived, risk analysis might be carried out to assess whether residual values, although exceeding those envisaged by the TPO, are compatible with the environmental situation with regard to the quality and level of residual protection afforded by physical barriers.

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MUNICIPAL SOLID WASTE: REVERSE LOGISTICS PERFORMANCE DETERMINES OPPORTUNITY COST OF BULK TIPPING

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ABSTRACT
The opportunity cost concept refers to quantifying the opportunities lost upon choosing one investment option over a more economical alternative. The present study applies the concept to the process of choosing the best investment option for managing municipal solid waste. In a case study in Brazil, the options on the table are bulk collection and tipping versus reverse logistics with selective collection and sale of recovered components. The use of relative monetary values renders the results general and applicable in other scenarios. The bulk tipping option represents the reference cost of 100. The research postulates a linear relation between the opportunity cost of bulk tipping (y) and the efficiency of reverse logistics operations (x). Zero efficiency means bulk collection and tipping of all waste. Full efficiency means capture of all recyclable items, which in the case study amount to 80% of waste. Various intermediate points confirm the relationship that takes the form y=0.968x. The result shows that opportunity cost is dynamic in as much as changes of technologies and administrative procedures move it along that line. It also illustrates to municipal administrations the immediate economic effect of implementation and stepwise improvement of reverse logistics.

1. INTRODUCTION

Different countries apply different solutions to the municipal solid waste problem. Increasing urbanization has increased the competition for space in cities, and its value has risen to a point where landfills are no longer an obvious option for occupying that space. The more advanced options include mechanical-biological pretreatment, source separation, recycling and large-scale composting. The present research is concerned specifically with the situation prevailing in Brazil, where those options are not yet the rule. The administration of municipal waste is in the stage of transition from dumpsites to landfills, mandated by federal law (Brazil 2007, Brazil 2010). The law requires of municipal administrations that they create and implement local plans to eradicate dumpsites and provide collection service to all residences. The transition has progressed to 52%, which means that 52% of the municipalities are already operating sanitary landfills. In this context, tipping all municipal waste at the landfills presently represents the solution to the waste problem. Diversion efforts are in their infancy. Of the 1.05 kg per person per day of waste collected, only 37 grams find their way into the reverse logistics chain (Brazil 2014). There is demand for research in order to move forward from this modest result. The present study uses the concept of opportunity cost accounting to identify economically viable alternatives to the landfill for the destination of municipal waste. The starting point is the expense reported for mixed collection and landfilling of municipal waste (Brazil 2014), whose average stands at BRL 109.96 per person per year. With the average national waste production of 1.05 kg per person per day, this translates into BRL 286.91 per tonne (109.96*1000)/(1.05*365). The opportunity cost concept can point to other applications of this amount that are economically and environmentally more interesting.

The concept of opportunity cost originated in the School of Vienna in the early 20th century. It is defined as the value of a productive resource in economic theory (Pereira 1990, Wieser 1926). It is the study of scarcity with its resulting phenomena, like the need for choosing the best use of resources among various mutually exclusive options. It reflects the obligation of making choices (Santos 2000, Beuren 1993), and represents the value of a sacrificed alternate option of investment (New Oxford 2019). The value is time-dependent. If at the time the investment decision has to be made, only one destination of resources

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is known and considered urgent, there really is no choice and no sacrificed alternative. Consequently, there is no opportunity cost (Buchanan 2015). Even if at a later date, other opportunities appear for allocation of the same resources to the same end, the original decision may not be reversed, because it has been executed. As a result, opportunity cost is a dynamic and forward-looking concept. Buchanan (2015) also contents that any application of the opportunity cost concept to non-market settings requires analytical clarification in each particular case.

The present case study uses data from the municipality of Ituiutaba in central Brazil. Prior to 2005, the municipality operated a dumpsite. At the time, this was the only option available, and consequently had zero opportunity cost. With federal legislation asking for the elimination of dumpsites, those sites suddenly acquired opportunity costs as the viable and compulsory alternative of landfills arose. Consequently, the municipality constructed a landfill in 2005. Again, this was the only option on the horizon, and thus initially, had zero opportunity cost. The next viable alternative to appear in the municipality after the landfill was reverse logistics, which is the subject of the present study. Reverse logistics does not function instantaneously. It passes through sequential stages of success, during which the opportunity cost of the landfill increases until reaching a value that corresponds to the highest possible diversion rate. The opportunity cost of the reverse logistics procedure accompanies its stages of success. Every new stage burdens the previous one with an opportunity cost. The eventual future option will be "zero-waste", advocated by the Zero Waste International Alliance (ZWIA 2019), which is the logical endpoint of reverse logistics or its final stage of success. It has already been experimented with in small communities (Fehr 2012, Fehr 2014). That sequence of events illustrates the time dependence of opportunity costs, as mentioned earlier. The sequence may also be accelerated. Research is reported on that considers overriding certain steps, like progressing from dumpsites directly to reverse logistics and ignoring the landfill option (Veiga 2019). This would apply to municipalities who failed to advance during the time period when the landfill was the best viable alternative. The opportunity cost of dumpsites in the era of reverse logistics is enormous. It accumulated during the time of omission to construct the landfill. The opportunity cost of an investment option always reflects the time distance the decision maker remains behind the most modern technical and administrative options known and available.

The main objective of this communication is to provide a value for the opportunity cost of the bulk tipping option at a new landfill to be constructed in the municipality of Ituiutaba MG Brazil, with 103945 inhabitants (2014 census), considering reverse logistics as the best viable alternative at its various stages of success or efficiency. The originality resides in showing that the opportunity cost is dynamic, and is a function of reverse logistics efficiency.

The complementary objectives are as follows: Show the existence of hidden costs and lost opportunities inherent in the bulk tipping option. Provide an economic analysis to quantify the hidden costs. Quantify the benefits achievable with progressive reverse logistics. Challenge the paradigm of bulk tipping as solution to the waste problem. Illustrate the reasoning with a specific case and its specific data.

2. METHODS

The research needed to collect a great variety of information and data as basis for the calculation of opportunity costs. The qualitative part of this investigation consisted of establishing contacts within the community of stakeholders, through which it was then possible to arrive at the quantitative data on waste movement. The interaction was quite dynamic. As explained by Godoy (1995), qualitative research is not limited to asking questions and writing down the answers. The authors temporarily joined the teams of municipal administrators, real estate agents and operators of waste collection and tipping in order to create familiarity with the situation, gain insight and open up space for exchange of ideas. This type of procedure is in accordance with research schemes developed by Alencar and Gomes (1998) and by Foote and Whyte (1990). It also involved the participation of the authors in meetings and seminars. Eventually, it led to the identification of the correct persons for obtaining specific quantitative data on waste movement and costs. Through this interaction, the authors also gained access to specific documents within the municipal administration, like reports and budgets. Gil (2006) refers to this method as documental study.

The use of a case study was necessary in order to proceed to considerable depth in terms of a real-life situation that could use and expand existing theory, as mentioned by Yin (2001). In the case at hand, the sources of information were the various departments of the municipal administration, the autonomous municipal sanitation authority, the landfill operator and a cooperative of recyclers. The types of waste presently tipped at the landfill were identified as domestic waste, gardening discards, street cleaning output, sewage treatment sludge and institutional and commercial garbage. The time span of this research goes from 2007 to the publication of the first research report in 2016. Opportunity cost calculations involved the comparison between bulk collection and tipping as practiced now and a reverse logistics scheme as the most viable alternative. Various progressive stages of that scheme produced the corresponding opportunity costs of bulk tipping. A predictable outcome resulted, defined by a mathematical relationship between opportunity cost of bulk tipping and efficiency of reverse logistics. The following components entered the opportunity cost determination: landfill construction, equipment and its life span for the total tipping option, value of land occupied by the landfill and its vicinity, market values of residues tipped, costs of collection and landfill operation, costs of reverse logistics.

As for the calculation procedure, the authors followed the ideas of Buchanan (2015) and New Oxford (2019). As opportunity cost only exists at the moment of an investment decision, this study determined it to support the decision that will be on the table in 2020 when the present landfill in Ituiutaba reaches its capacity and has to be closed. Data collected for the last 10 years, both on collection and landfill costs, and on the increasing reverse logistics operations, allowed for presenting and quantifying two options for the 2020 decision. Both options contain a landfill as inevitable component. Option 1 uses bulk collection and tipping of all trash. Option 2 invests in reverse logistics to divert part of the trash from the landfill and to tip only the rest. In order to render the results general and universal, the authors consider the bulk tipping cost as reference value, and determine all other costs and benefits relative to it.

3. RESULTS

3.1 Present Situation in numbers

The data collected on waste movement in Ituiutaba during the ten-year period from 2007 to 2016 show 218,725 tonnes collected and tipped. This means 60 tonnes per day. On an individual basis, it comes to 0.578 kg per person per day, which is well below the national average of 1.05 kg per person per day. The raw data obtained from the various sources consulted would only justify using three significant digits. In the following calculations, the authors retained as many digits as thought fit to allow the reader to accompany and verify the results, keeping in mind the three digit precision limit. Table 1 shows the composition of municipal waste. The first column represents "inert" components. The term "rejects" refers to mixed material that cannot be separated and allocated to inerts and biodegradables.

The composition report provides the recycling potential as 80.5% of waste. This value forms the natural final target for stepwise improvement of the reverse logistics operations over planned time intervals, and receives the designation of 100% reverse logistics efficiency in the subsequent calculations.

The expenditures for the ten-year period, obtained from the municipal budget and reduced to cost per year are detailed in the subsequent paragraphs.

3.1.1 Tipping costs

- Collection and tipping operations BRL 96.94/tonne.
- Landfill construction and maintenance operations BRL 59.45/tonne.
- Total BRL 156.39/tonne.

This value is below the national average estimated earlier.
Expenditure with bulk collection and tipping in the ten-year period:

- BRL 156.39/tonne * 218,725 tonnes/10 years = BRL 3,420,640.28/year.

TABLE 1: Composition of waste collected and tipped by weight.

Paper, cardboard, tetrapak	11.4 %			
Plastics	16,4 %			
Glass	2.1%			
Metals	1.8%	31,7%		
Biodegradables		48,8%	80,5%	
Rejects			19,5%	100%

Source: Municipal Department of Urban Services

3.1.2 Productivity of land occupied by landfill

The average agricultural productivity of farmland around the city obtained from real estate agents and farmers is BRL 547.69 per hectare per year. The present landfill occupies 21.5 hectares (ha). The value sacrificed by occupying this area with the landfill amounts to 547.69*21.5 = 11,775.34 BRL/year.

The sum of the last two items BRL 3,420,640.28 + 11,775.34 = BRL 3,432,415.62/year forms the reference of 100 for opportunity cost calculations.

3.1.3 Landfill diversion

A functional reverse logistics scheme at its sequential stages of success or efficiency will divert increasing fractions of waste from the landfill and recycle them. The subsequent calculations relate the credit derived from those fractions to the reference value of 100.

3.1.4 Value of recycled material

The average income for dry material from selective collection as reported by the recyclers' cooperative was BRL 434.07/tonne, considering the composition report of Table 1 and their operating expenses. This material represents 31.7% of the waste stream (Table 1) of 21,872.5 tonnes/year.

The average income for compost produced from biodegradable residue has been reported by Fehr (2016) as BRL 240.00/tonne of residue. This material represents 48.8% of the waste stream (Table 1) of 21,872.5 tonnes/year.

3.1.5 Expenditure with reverse logistics

The municipal contribution to reverse logistics consists of providing selective collection infrastructure, and administrative and real estate support to a cooperative. The value obtained from the Municipal Department of Urban Services was BRL 254,024.88 per year when 4% of waste were actually recycled. The subsequent calculations will adjust the value to different fractions of recycle.

3.1.6 Taxes paid by the operators of reverse logistics

The recyclers' cooperative with its present infrastructure and its present production rate pays taxes of BRL 58,321.52 per year. In order to remain conservative, the calculations will not change this value for future situations.

3.2 Opportunity cost

Based on the data provided so far, Tables 2 to 7 detail the calculation of the opportunity cost of bulk tipping for various stages of reverse logistics efficiency.

Table 2 shows the division of waste between the landfill and recycling operations for various degrees of reverse logistics efficiency.

Table 3 defines the case of zero reverse logistics. The cost of the bulk tipping operation forms the reference value of 100 for opportunity cost calculations.

Table 4 illustrates the calculation procedure for arriving at a relative opportunity cost from a given value of reverse logistics efficiency. In this case, the reverse logistics

TABLE 2: Material movement as function of reverse logistics efficiency for the waste composition of Table 1. Units: % of waste.

Reverse logistics efficiency	destination	inerts	biodegradables	rejects	total
0%	recycled	0	0	0	0
	tipped	31.7	48.8	19.5	100
35%	recycled	11.1	17.1	0	28.2
	tipped	20.6	31.7	19.5	71.8
60%	recycled	19.0	29.3	0	48.3
	tipped	12.7	19.5	19.5	51.7
80%	recycled	25.4	39.0	0	64.4
	tipped	6.3	9.8	19.5	35.6
100%	recycled	31.7	48.8	0	80.5
	tipped	0	0	19.5	19.5

Explanation of Table 2. Example 80% reverse logistics efficiency. Inert material: Capture and recycle 80% of 31.7 or 25.4. Tipping 31.7 – 25.4 = 6.3. Biodegradable material: Capture and recycle 80% of 48.8 or 39.0. Tipping 48.8 – 39.0 = 9.8. Rejected material totally tipped 19.5.

TABLE 3: Determination of the reference value for opportunity costs. Cost of waste handling with zero reverse logistics.

Expenses	lost land productivity	11,775.34/3,432,415.62 =	0.003431
	reverse logistics cost	0	0
	tipping cost	3,420,640.28/3,432,415.62 =	0.996569
	total expenses		1.000000
Credits	value of inert material	0	
	value of compost	0	
	taxes paid	0	
	total credits		0

*Reference expenses are 1 * 100 = 100 = 3,432,415.62 = actual expenses. Opportunity cost = reference expenses – actual expenses = 1 – 1 = 0*

TABLE 4: Cost of waste handling for 35% reverse logistics efficiency relative to reference value (according to Table 2, reverse logistics captures 28.2% of waste).

Expenses	lost land productivity	11,775.34*0.718/3,432,415.62 =	0.002463
	reverse logistics cost	254,024.88*(0.282/0.04)/3,432,415.62 =	0.521743
	tipping cost	156.39*21872.5*0.718/3,432,415.62 =	0.715537
	total expenses		1.239754
Credits	value of inert material	21,872.5*0.111*434.07/3,432,415.62 =	0.307030
	value of compost	21,872.5*0.171*240.00/3,432,415.62 =	0.261521
	taxes paid	58,321.52/3,432,415.62 =	0.016991
	total credits		0.585542

Net expenses 1.239754 – 0.585542 = 0.654212
Opportunity cost = reference expenses – actual expenses = 1 – 0.654212 = 0.345788
Opportunity cost referred to 100 = 34.6
Relation: opportunity cost / reverse logistics efficiency = 34.6 / 35 = 0.989

*Explanation of Table 4. Lost land productivity. Only 71.8% of waste is tipped. Reverse logistics cost adjusted from the experimental value for 4% recycle to 28.2% recycle. Tipping cost: 71.8% of 21872.5 tonnes of waste tipped at a cost of BRL 156.39 per tonne. Inert material: Recycle is 31.7*0.35 = 0.111 of 21,872.5 tonnes at BRL 434.07 per tonne. Value of compost: Recycle is 48.8*0.35 = 0.171 of 21,872.5 tonnes at BRL 240.00 per tonne.*

efficiency of 35% leads to the relative opportunity cost of 34.6%. The relationship is 34.6/35=0.989.

Explanation of Table 4. Lost land productivity. Only 71.8% of waste is tipped. Reverse logistics cost adjusted from the experimental value for 4% recycle to 28.2% recycle. Tipping cost: 71.8% of 21872.5 tonnes of waste tipped at a cost of BRL 156.39 per tonne. Inert material: Recycle is 31.7*0.35 = 0.111 of 21,872.5 tonnes at BRL 434.07 per tonne. Value of compost: Recycle is 48.8*0.35 = 0.171 of 21,872.5 tonnes at BRL 240.00 per tonne.

Table 5 repeats the procedure for 60% reverse logistics efficiency leading to 58% relative opportunity cost. The relationship is 58.0/60=0.967.

Table 6 repeats the procedure for 80% reverse logistics efficiency leading to 76.9% relative opportunity cost. The relationship is 76.9/80=0.961.

Table 7 depicts the situation for 100% reverse logistics efficiency with the relative opportunity cost of 95.6%. The relationship is 95.6/100=0.956.

Admitting an error range of 2%, the relation of opportu-

TABLE 5: Cost of waste handling for 60% reverse logistics efficiency relative to reference value (according to Table 2, reverse logistics captures 48.3% of waste).

Expenses	lost land productivity	11,775.34*0.517/3,432,415.62 =	0.001774
	reverse logistics cost	254,024.88*(0.483/0.04)/3,432,415.62 =	0.893642
	tipping cost	156.39*21872.5*0.517/3,432,415.62 =	0.515226
	total expenses		1.410642
Credits	value of inert material	21,872.5*0.190*434.07/3,432,415.62 =	0.525548
	value of compost	21,872.5*0.293*240.00/3,432,415.62 =	0.448103
	taxes paid	58,321.52/3,432,415.62 =	0.016991
	total credits		0.990642

Net expenses 1.410642 – 0.990642 = 0.420000
Opportunity cost = reference expenses – actual expenses = 1 – 0.420000 = 0.580000
Opportunity cost referred to 100 = 58.0
Relation: opportunity cost / reverse logistics efficiency = 58.0 / 60 = 0.967

Explanation of Table 5. Same procedure as Table 4.

TABLE 6: Cost of waste handling for 80% reverse logistics efficiency relative to reference value (according to Table 2, reverse logistics captures 64.4% of waste).

Expenses	lost land productivity	11,775.34*0.356/3,432,415.62 =	0.001221
	reverse logistics cost	254,024.88*(0.644/0.04)/3,432,415.62 =	1.191523
	tipping cost	156.39*21,872.5*0.356/3,432,415.62 =	0.354779
	total expenses		1.547523
Credits	value of inert material	21,872.5*0.254*434.07/3,432,415.62 =	0.702575
	value of compost	21,872.5*0.390*240.00/3,432,415.62 =	0.596451
	taxes paid	58,321.52/3,432,415.62 =	0.016991
	total credits		1.316017

Net expenses 1.547523 – 1.316017 = 0.231506
Opportunity cost = reference expenses – actual expenses = 1 – 0.231506 = 0.768494
Opportunity cost referred to 100 = 76.9
Relation: opportunity cost / reverse logistics efficiency = 76.9 / 80 = 0.961

Explanation of Table 6. Same procedure as Table 4.

TABLE 7: Cost of waste handling for 100% reverse logistics efficiency relative to reference value (according to Table 2, reverse logistics captures 80.5% of waste).

Expenses	lost land productivity	11,775.34*0.195/3,432,415.62 =	0.000669
	reverse logistics cost	254,024.88*(0.805/0.04)/3,432,415.62 =	1.489403
	tipping cost	156.39*21872.5*0.195/3,432,415.62 =	0.194331
	total expenses		1.684403
Credits	value of inert material	21,872.5*0.317*434.07/3,432,415.62 =	0.876834
	value of compost	21,872.5*0.488*240.00/3,432,415.62 =	0.746328
	taxes paid	58,321.52/3,432,415.62 =	0.016991
	total credits		1.640153

Net expenses 1.684403 – 1.640153 = 0.044250
Opportunity cost = reference expenses – actual expenses = 1 – 0.044250 = 0.955750
Opportunity cost referred to 100 = 95.6
Relation: opportunity cost / reverse logistics efficiency = 95.6 / 100 = 0.956

Explanation of Table 7. Same procedure as Table 4.

nity cost (y) to reverse logistics efficiency (x) is constant at 0.968. This leads to the equation of a straight line: y=0.968x from which to predict the opportunity cost.

The total economic cost of choosing bulk tipping is the sum of the expense actually occurred and the savings sacrificed for each stage of reverse logistics efficiency. For example, in the case of 80% reverse logistics efficiency, the economic cost is 100 + 76.9 or 176.9% of the investment actually made. Here lies the power of the opportunity cost concept. It shows hidden items that are not visible in the municipal budget. In fact, the budget only reports an expense of 100 for bulk tipping, whereas the real cost of this option is 176.9 because it sacrifices savings of 76.9 by not choosing the alternate option available. This is the message to the municipal administration for the decision process on schedule for 2020.

4. DISCUSSION

The present landfill in the city under study has been in operation since 2005 and will reach its projected capacity in 2020. At that time, it will have to be closed, and a new decision on the continuity of municipal solid waste management will have to be made. In 2005, bulk collection and tipping was the only option on the table in the city. The reason: Federal law at the time asked for moving from dumpsites to landfills, and there was no experience with reverse logistics.

In the last ten years, however, reverse logistics operations with selective collection appeared on the scene and produced numbers on recycling, which were missing heretofore. Consequently, in 2020, there will be at least two options of investment. The first option is to construct a new landfill and continue with the bulk collection and tipping scheme. The second option is to invest in reverse logistics and target the ideal recycling rate of 80.5% of all waste, or else target intermediate recycling rates at the choice of the administration for planned time intervals. The present research collected data on both options during the last ten years, and compared the two investment choices. As an original contribution to the decision process, it developed an equation relating the efficiency of reverse logistics to opportunity cost. The municipal administration can now use the results for making the decision in 2020. In relative yearly terms, the first option requires and investment of 100 monetary units per year. The ideal second option solves the same problem with an investment of 4.4 (100 – 95.6) monetary units per year. It consists of organizing reverse logistics in order to recycle 80.5% of all waste, i.e. sell the selected inert components and sell compost produced from the biodegradable components. The difference of 95.6 monetary units becomes available for other uses in the municipal budget and thus represents the opportunity cost of choosing the first option. In economic terms, the real cost of choosing the first option is a surprising 195.6 monetary units, which consist of the investment proper of 100 and the credits of 95.6 sacrificed by ruling out the second option. The merit of this research resides in providing data for the imminent decision on the continuity of waste management in the city, on the table for 2020 when the present landfill will be closed, and in quantifying the opportunity cost of the landfill at a time when administrative and technical means are available to operate a reverse logistics scheme.

The data and calculations specifically refer to the case study, but the method is general. The most impressive contribution of opportunity cost accounting to investment decisions concerning municipal waste management resides in the fact that it identifies credits possible with alternate procedures, which are not visible in the budget. In the case at hand, the budget shows the investment of 100 for the new landfill, but does not show the savings of 95.6 obtainable with the alternate procedure. The study confirmed the dynamic nature of opportunity costs. Within the ten-year period considered, the cost of building and operating the landfill stayed constant at 100, but its opportunity cost rose from zero to a maximum of 95.6 as a consequence of the appearance of a viable alternative with a price tag of only 4.4 units.

As a case study, this research provided specific data on the situation in the municipality of Ituiutaba in 2016. The data are specific, but the procedure can possibly be extrapolated and thus be useful in other municipal contexts. The idea of an equation relating opportunity cost to reverse logistics efficiency is both original and generally applicable, although its precise form may vary from locality to locality. The equation represents a very visible guide for starting and expanding reverse logistic programs that move any municipality beyond strict legal requirements to eliminate the dumpsites.

5. CONCLUSIONS

The study attained its main objective, which was to provide a value for the opportunity cost of tipping all waste at a landfill in the municipality of Ituiutaba MG Brazil, with 103945 inhabitants (2014 census), in view of an investment decision to be reached in 2020 when reverse logistics will be available as the best viable alternative. The study also attended to complementary objectives. It identified the hidden costs of a landfill that do not appear in the municipal budget and refer to benefits derived from recycling part of the waste and to liberating part of the land occupied by the landfill.

The investment figures for the various items of the available options were translated into relative monetary units in order to make the results more comprehensible to an international audience.

The investment required in the landfill as the only destination was considered the reference of 100 in the context of working with relative monetary units.

The alternative of reverse logistics at different stages of efficiency required investments in the range of 100 (0% reverse logistics) to 4.4 (100% reverse logistics) to solve the same problem, namely manage municipal waste. Consequently, the opportunity cost of the landfill is dynamic and increases with improving reverse logistics. The research produced the equation $y = 0.968\ x$ to predict the opportunity cost y resulting from any reverse logistics efficiency x.

The economic value of the investment in the landfill for 100% reverse logistics efficiency was 195.6 comprising the sunk money invested and the opportunities lost by excluding the better alternative.

The study illustrated the dynamic nature of opportunity costs by showing their time dependence. An opportunity cost of an investment decision only exists if at the time of the decision a better alternative for applying the resources is known, available and technically mature for implementation.

In the context of the case study, the landfill constructed in 2005 had no opportunity cost because no better alternative for managing municipal waste was obvious in the city. During the following ten years, reverse logistics operators gained experience and produced numerical information. Consequently, at the time of the next investment decision, due in 2020, the opportunity cost of a landfill in the city will be in the range of zero to 95.6 compared to its budget cost of 100, depending on the target chosen for reverse logistics efficiency. In essence, the research challenged the landfill as destination of municipal waste by showing the viability of a cheaper alternative.

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PARTICIPATORY SOLID WASTE GOVERNANCE AND THE ROLE OF SOCIAL AND SOLIDARITY ECONOMY: EXPERIENCES FROM SÃO PAULO, BRAZIL

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ABSTRACT
Waste governance is emerging as transdisciplinary and inter-sectoral approach to waste management and policy, overcoming primarily prescriptive engineering perspectives of waste. The process of governing waste involves the articulation of different structures, institutions, policies, practices and actors. Paying attention to issues of power, scale, and equity are important in the search for more democratic practices. Innovative forms of governance are emerging as decentralized, participatory and inclusive, focused on waste reduction and resource recovery. Social and Solidarity Economy (SSE) is an innovative alternative in generating work and income and a response in favor of social and labor inclusion. It can also be considered as a new, more humane and inclusive development model. With this article we aim to provide practical knowledge on the contributions of grassroots organizations and networks in waste management, supporting the discussion of waste governance in the context of the SSE. We present different experiences of waste picker organizations in the metropolitan region of São Paulo, Brazil to showcase their assets and to discuss prevailing challenges. Employing the SSE as a new development model allows us to address everyday issues of waste generation, management and governance in Brazilian cities and in other parts of the world; particularly from the perspective of organized waste pickers in associations, cooperatives and networks. This is a development paradigm which goes beyond just economic considerations, as highlighted with examples from waste management.

1. INTRODUCTION

Cities worldwide generate between 7 and 10 billion tonnes of waste every year from households, commerce, industry and construction; a number that is expected to double by 2030, given the rapid urban growth trend (UNEP & ISWA, 2015). Particularly in cities within lower income countries, waste collection is limited and typically only reaches high income neighbourhoods. Approximately 2 billion people in lower income countries are without access to solid waste collection and even if collected, the waste often ends up in open dumps. Worldwide, this represents 33% of urban waste mounting to a serious and global health hazard (Kaza, Yao, Bhada-Tata & Van Woerden, 2018). Basic waste management challenges include open dumps, uncontrolled dumping, open burning and inadequate access to waste services and unsafe recycling practices, particularly in informal neighbourhoods (Wilson, et al., 2015).

Local governments encounter multiple financial and technical constraints as well as numerous challenges in waste management, with the increase in volume, the growing material complexity or the difficulties with source separation. Mismanaged waste becomes a global wicked problem, particularly noticeable when it washes up in places far from its origin, causing environmental contamination, entanglement for animals and affecting the food chain. Many everyday challenges, such as poverty, pollution, flooding, poor health, littering and environmental degradation are closely related to the lack of proper answers to waste issues. Environmental governance (Lemos & Agrawal, 2014) plays a key role in addressing these challenges, involving all scales (global, national, regional and local) and all sectors from government, and business to civil society at large (UN Environment, 2019). Complex environmental problems are always closely related with other policy areas (Jordan & Lenschow, 2010), yet efforts to overcome the separation and boundaries as a result of sector approaches in gover-

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nance have not made sufficient progress (Adelle & Nilsson, 2015). A better understanding of the reality, the hurdles, the missing links and the assets in waste governance in different countries and cities is required to achieve better waste management outcomes.

A key step towards reducing the environmental and health impacts of waste is to shift the perspective from regarding waste as a threat to seeing it as a resource that must be recovered. That is how waste pickers all over the world have always perceived waste, as a resource to support their livelihoods (Wilson et al., 2006, 2012; Chaturvedi, 2009; Samson, 2009; Scheinberg, Simpson & Mol, 2011; Linzner & Lange, 2013; Gutberlet, 2008; 2016). Today an estimated 15 to 20 million waste pickers work in waste recovery worldwide (ILO, 2013) and while these waste pickers are key environmental ambassadors operating in the circular economy, most of them work under unacceptable conditions (Wilson et al., 2015; Duan, Li & Liu, 2017), exposed to health risks (Binion & Gutberlet, 2012; Gutberlet & Uddin, 2018), social exclusion and stigmatization.

Participatory waste governance, the focus of this paper, draws the attention to the integration of waste pickers in formal recycling, of which few examples have so far been discussed in the literature (Jaligot et al., 2016; Wilson et al., 2012; Jacobi & Besen, 2011; Sembiring & Nitivattananon, 2010).

Our case study is located in the metropolitan region of São Paulo (MRSP), Brazil, with 21 million inhabitants. The region has sanitary landfills and most of the formal city has regular waste collection services, but little to no formal recycling programs. Informal settlements are generally excluded from regular collection and open dumping becomes widespread. Very few municipalities in the MRSP have started programs for selective waste collection and recycling, and are working in co-production with organized waste pickers. These programs service mostly the central parts of the city, while in the periphery recycling happens informally, involving a majority of autonomous waste pickers.

Good governance and the sustainable management of solid waste links up with all of the 17 Sustainable Development Goals (SDGs) established under Agenda 2030. Particularly, goal number 12 proposes to ensure sustainable production and consumption patterns and specifically target 12.5, demands a substantial reduction of waste generation through prevention, reduction, recycling and reuse, by 2030 (ONUBR, 2015). This requires consolidating a multi-scalar governance approach and the alignment of international, national and local actions (UN Environment, 2019). Strategies tackling sustainability through packaging design and product lifecycle, for example, that aim to reduce the impacts on natural resource extraction, can promote a 10 to 15 per cent reduction in global greenhouse gas emissions just from recycling and improvements in solid waste management, and between 15 and 20 per cent by including measures to prevent or reduce the production of waste (Wilson et al., 2015, p. 12).

One of the key objectives of the paper is to demonstrate the potential of participatory waste governance, specifically with the inclusion of waste pickers (organized in cooperatives, associations or networks) in selective waste collection. We, further want to investigate the role of the Social and Solidarity Economy (SSE) (Moulaert & Ailenei, 2005; Moulaert & Nussbaumer, 2005) as an instrument supporting the integration of waste pickers and possibly as a driver for a new social and economic development model.

The next section introduces the two main concepts used as analytical framework: (a) waste governance and (a) social and solidarity economy. We then outline the methodology and describe the qualitative research strategy involving six waste picker networks and the national waste picker organization in the metropolitan region of São Paulo, providing a rational for the choice of our study region. The following section presents and discusses the results, drawing specific attention to the role of participatory governance and Social and Solidarity Economy in expanding the capabilities of waste pickers. We conclude with reflections on the assets waste picker organizations can bring in shaping participatory, environmental governance and some of the challenges organized waste pickers are facing, given the current situation in waste management in the region and beyond.

2. ANALYTICAL RESEARCH FRAMEWORK

2.1 Waste governance

Governance is about the organization of society or groups within it, as well as their decision-making structures and accountability. It is a broad, diverse and complex concept which takes different shapes in different geographic contexts (Andrews, 2008). "Governance determines who has power, who makes decisions, how other players make their voice heard and how account is rendered" (Institute on Governance, n.d.). The need for governance exists anytime a group of people come together to address a specific issue, e.g. waste management. Governance also concerns the "performance of the government, including public and private sectors, global and local arrangements, formal structures, informal norms and practices, and spontaneous and intentional systems of control" (Ezeah & Roberts, 2015, p. 122). Governance practices are shaped by the interactions of different social actors, which often come with diverse, conflictive, antagonistic interests, views and values, which governance is supposed to settle or reconcile. In that process, rules able to regulate and intermediate negotiation processes are crucial, allowing to arrive at a common understanding on the problems to be solved and at possible strategies to overcome these problems.

Waste governance takes an inter-sectoral approach to waste management with a focus on policies and regulations, as well as institutional arrangements that delimit how decisions are made, who participates in the deliberations and how work is carried out. Thus, from a waste governance perspective, there are examples where some governments are currently moving away from a primarily prescriptive, engineering approach (Bulkeley et al., 2005; Moore, 2012) towards forms that are more inclusive of different actors, different methods and diverse perspectives.

Waste is a noticeable proof of inefficiency and wrong decision making in any environment and requires innovative governance solutions. Good waste governance requires the reconceptualization of waste as a resource, it takes an inclusive approach to waste management that allows e. g. grassroots waste actors (waste pickers, small-scale waste entrepreneurs, community-based organizations, non-governmental organizations, or citizens at large) to participate and it encourages innovative approaches to prevent the generation of waste and wasted resources.

The process of governing waste involves the articulation of different institutions, structures, technologies, practices and actors; and issues of power, scale, and equity are important in shaping the outcomes. Innovative forms of governance are decentralized, participatory and inclusive, focused on waste reduction and resource recovery. These attributes allow for the socio-productive inclusion of waste pickers and the achievement of greater sustainability (e.g. zero waste, circular economy). Good waste governance also addresses: poverty reduction, builds community resilience, tackles climate change and increases environmental sustainability; besides working towards reduction, reuse and recycling.

In the global context waste governance is one of the biggest environmental and urban challenges (Hoornweg Bhada-Tata, 2012), regardless of city size and political priority (UN - Habitat, 2010). Waste is intrinsically related to environmental health issues and environmental governance is relevant specifically given the challenges cities are facing, with climate change, raising poverty, ecosystem degradation or pollution. Environmental governance is defined as “interventions aiming at changes in environment-related incentives, knowledge, institutions, decision making, and behaviors” (Lemos & Agrawa, 2006, p. 298).

The state has an important role to play in shifting towards inclusive environmental governance (Adelle & Nilsson, 2015), e.g. by ratifying and implementing conventions, promoting research, designing new policies, laws and regulations, and by supporting vulnerable populations, e.g. through Social and Solidarity Economy programs (Alvarez, 2015). In lower income countries, including Brazil, waste management problems are amplified by rapid urban growth, pressured and insufficient facilities, poor service delivery, lack of resources and often also lack of political will (Ezeah & Roberts, 2015). Furthermore, the diversity of materials in solid waste and their specific forms of management makes governance more complex, insofar as it also involves unknown materials and new actors. Organic material mixed within inorganic waste also creates a wide range of other problems.

Waste pickers formalized in cooperatives or associations are important actors in waste governance (Asim et al., 2012; Dias, 2016; Fei et al., 2016; Gutberlet, 2015; Scheinberg, 2012; Velis, 2017; Velis et al., 2012; Wilson et al., 2006). Particularly in Latin America, their participation in municipal selective waste collection programs has not only created better local waste management and circular economy results, but has also generated ‘socio-productive inclusion’, which is defined as implementation of policies (SSE) that promote the organization of waste pickers and their integration into municipal waste management, generating income and providing better working conditions (Fracalanza & Besen, 2016). Selective waste collection operated by waste pickers is considered a successful ‘social technology’ (Rutkowski & Rutkowski, 2015). Moreover, we argue that participatory governance is a prerequisite to effectively address urban waste problems in the global South.

As we will demonstrate ahead, the transversality of SSE actions, especially those related to economic, social and environmental contexts, constitutes a theoretical and practical platform closely related to the issue of waste governance in practice. The generation of work and income, combined with a greater involvement with work practices of recycling cooperatives and associations, for example, in addition to the participation in political decision-making bodies (such as public councils, when they exist) and the generation of environmental externalities, allow for the activity of many recycling cooperatives and associations to be connected with economic, social and environmental aspects.

Within these enterprises, it is expected that collective organizations along the lines of the SSE allow for the development of participatory and democratic governance, giving those involved the power of voice towards their working conditions and the enterprise in general. From an operational point of view, as we know, the activity of thousands of women and men in the collection, sorting and processing of recyclable materials has been increasingly recognized by society, in recent years. This is due, in large part, to the recognition of its economic and environmental relevance, in addition to the potential for social inclusion of a large contingent of people excluded from other work opportunities; people who live in social marginality and vulnerability (Millar, 2018). In Brazil, the collective organization of waste pickers in cooperatives and associations represents an important part of the SSE (IPEA, 2016). This fact allows for reciprocal strengthening of the SSE and the democratic and participatory management practices in waste picker cooperatives and associations.

2.2 The Social and solidarity economy

The SSE is understood as alternative economy generating work and income, as response to the demand for social and labor inclusion. For many, the SSE resembles an innovative, more humane and inclusive development model compared to the capitalist market driven economic development model (Morais, Dash & Bacic, 2017). The SSE framework sees a diversity of economic and social practices, carrying out different activities including the production of goods, services, solidary finances, exchange of goods and services, fair trade and solidary consumption (Morais, 2013, 2014). According to Vieira, “as a concept, [SSE] has a triple nature. At the same time that it is an empirically verifiable object, it is also a social movement and a propositional theory of socioeconomic change” (2005, p. 56).

The existing terminological diversity was described by Moreno (1996), highlighting the most frequent terms: third sector; voluntary or independent sector; 'non-profit, philanthropic or charitable sector; non-governmental organizations (NGOs); intermediate sector; tax-free sector; and social and solidarity economy. Tremblay (2009) lists country-specific terminologies, such as Social Economy (USA and Canada), Solidarity Economy (Argentina, Brazil, Chile and Quebec), People´s Economy (Asia), Associative Movements (Senegal and Turkey), Civil Society (South Africa) and Community Economic Development (Australia, New Zealand and Anglo Saxon Canada). The large number of different terms reflect the multiple modes of generation, performance and behavior that this sector manifests in different geographies, based on the specific historical, cultural, political or legal contexts (UN Inter-Agency Task Force on Social and Solidarity Economy, 2018). The following Table 1 summarizes some of the more widespread definitions, operating principles and respective entities of the SSE.

Alvarez synthetizes SSE as a set of socioeconomic practices that "combine cooperative entrepreneurship with the association of people seeking to meet needs" (2018, p. 6). SSE organizations can offer comparative advantages to address social, economic and political challenges around the world, including social cohesion, empowerment and the recognition of a pluralistic economy. SSE is therefore becoming more important at the present time, given the global economic crisis, when solutions will require, among other aspects, a more inclusive and sustainable development paradigm, as is also expected under environmental participatory governance (Adelle & Nilsson, 2015; Morais, Dash & Bacic, 2017). SSE organizations share key features that set them apart from conventional enterprises. They are often bottom-up, have a significant participation of volunteers, who often play an important role in the start-up phase of the organization, consequently, their governance structure also tends to be more inclusive and democratic. This is a new type of community-based development approach through which beneficiaries gain more direct control over project decision-making, implementation and evaluation processes. It supports collective action, community empowerment, and demand-driven local service delivery. These are the seeds for innovative, increasingly self-sustaining local development processes.

There is a wide spectrum of different actors and legal arrangements for SSE in Brazil, however, cooperatives are easiest to find and are the ones with the greatest tradition in the country. Except for the studies on cooperatives, the scientific discussion on SSE is relatively recent (Morais & Bacic, 2018; 2019). In Brazil, Solidarity Economy (SE) constitutes a mode of production, which differs from capitalism, aspiring for more democratic decision making. According to Paul Singer SE "was created by workers, in the beginning of Industrial Capitalism, as a solution to poverty and unemployment" (Singer, 2002, p. 83). For Singer, the weapon available to those who are deprived of capital is solidarity and SE emerges as reaction to the deprivations that the dominant system refuses to address.

In general terms, the key feature of SSE enterprises is that they produce goods or services with 'social value' and are not eminently guided by profit. Profit (or surplus), however, is essential for the sustainability and development of SSE organizations and enterprises. However, it is not considered as ultimate goal, in addition, it is used and distributed according to the specific rules inherent to the previously agreed and defined legal structures. Profit and productive surplus are necessary for the socio-economic viability of the SSE and are generally used to expand business and improve the human resources and infrastructure of the actors involved. Cooperatives, for example waste picker cooperatives, operate in a capitalist environment and are thus subject to many challenges and constraints. In his classical text Stewart Perry (1978) gave a detailed account of some of the limitations that cooperatives experience, e.g. caused by competitive pressures that affect the organization and that can distract from the original values and principles of the cooperative. Egan (1990) remarks that cooperatives are compelled to operate within the logic of capitalism, producing and selling in the market, "thereby reproducing the commodification of use values [which further] …forces cooperatives into competition with each other" (p. 72). Our research will demonstrate how particularly waste picker organizations in Brazil are committed to core values and principles of the SSE and have distinguished themselves from many other examples of cooperatives.

TABLE 1: Different definitions, operating principles and forms of the SSE.

Definitions of SSE	SSE Operational Principles	SSE Entities
Third sector Voluntary sector Non-profit sector Philanthropic sector Charitable sector Non-governmental organizations (NGO) Intermediate sector Tax-free sector Social economy Solidarity economy Social and solidarity economy People´s economy Associative movements Civil society movements Community economic development Social innovation Collaborative economy Corporate social responsibility Corporate citizenship Circular economy Common good economy	Social value Fair trade Responsible and conscious consumption Democratic and popular management Social, political and gender empowerment Local development Production with social, political and economic impacts of communities Collectivity Reciprocity Mix of market and non-market resources Social, political, economic and cultural transformation of territories Social cohesion Popular economy	Cooperatives Mutual benefit societies Associations Foundations Social enterprises Recovered factories Community banks Exchange clubs Solidarity economic enterprises

According to a broad international debate that takes place, within the International Labor Organization (ILO) and involves international SSE scholars and institutes (Borzaga, Salvatori & Bodini, 2017) we can affirm that:

a) SSE refers to specific forms of organizations and companies, with the most common types being cooperatives, mutual societies, associations, community organizations, social enterprises, foundations, Non-Governmental Organizations (NGOs), solidarity economy enterprises etc. It is, therefore, a group of dynamic and evolving organizations;
b) SSE organizations have common characteristics that differentiate them from public economy and traditional private economy organizations, as they share specific operating principles based on voluntary participation, solidarity, reciprocity, innovation, collective ownership and self-management;
c) The existing range of names that cover the SSE are divergent but yet related concepts. They all have certain geographical origins and theoretical backgrounds that emphasize particular dimensions of this economic form;
d) SSE organizations can offer comparative advantages to address social, economic and political challenges around the world, including social cohesion, 'empowerment' and the recognition of a plural economy;
e) Despite their diversity and heterogeneity, SSE has other fundamental characteristics that differentiate them from traditional companies, such as the fact that their organizations, in large part, are conceived within local communities in response to common opportunities or needs, as well as greater inclusion and democracy in its governance and decision-making processes;
f) SSE is, therefore, acquiring more importance at the present moment, in view of the global economic crisis, since the solutions will require, among other aspects, a more inclusive and sustainable development model.

In general, SSE can be defined as "a concept that refers to enterprises and organizations, in particular cooperatives, mutual benefit societies, associations, foundations and social enterprises, which specifically produce goods, services and knowledge while pursuing economic and social aims and fostering solidarity" (ILO, 2011, cited by Borzaga, Salvatori & Bodini, 2017, p. 14).

Greater multi-stakeholder participation in governance is a goal in environmental governance which can be facilitated under the SSE and depends on the synergies between governments and different government levels as well as between civil society organizations (UN Environment, 2019). We will further discuss the proposed approach to waste governance in the context of the MRSP. The next section examines waste picker organizations and networks, supported by the SSE and their impact on waste governance.

3. METHODOLOGY

Our research uses a qualitative approach, involving document analysis, in-depth key informant interviews and participatory observation. The initial stage involved the search and review of relevant literature on waste governance and the social and solidarity economy, with emphasis on lower income countries. Data collection happened between September and November 2018 and entailed semi-structured, in-depth interviews including both closed- and open-ended questions, involving six waste picker networks operating in the metropolitan region of São Paulo, as well as the National Waste Picker movement (Movimento Nacional de Catadores(as) de Materiais Recicláveis, MNCR). We chose this particular region for our case study because of our long-term trusted relationships with some of the waste picker organizations. Another reason was to explore the large diversity of different working situations among organized waste pickers in the MRSP, where we highly organized and well-equipped groups but also small-scale and precarious organizations.

All interviews were either held in the administrative unit of the network or cooperative. The interviews took between 2 and 3 hours on average. They were taped and later transcribed by the local research assistant (Solange Dias), who was also present during all interviews.

Except for one interview (Catasampa) which was conducted with just one representative of the network, several other members of the network always joined our conversations. The semi-structured interviews were designed to develop a deep understanding of the complex issues that surround the networks, and followed themes that captured the history of the network, their geographic and thematic scope, objectives and governance structure, as well as information related to the social innovations driven by the network. The interviews were manually theme coded to capture key ideas. Interview methodologies, however, do have their drawbacks, as the perspectives and experiences of individuals are not always accurate representations of actions or facts (Knox-Hayes, 2008). Rigour and credibility, therefore, were enhanced through the triangulation of our results across all interviews. The information was compared and complemented with a wide array of document sources (reports, academic papers, website information). The researchers have obtained approval from the University of Victoria Human Research Ethics Council for this project (Protocol # 17.193).

4. RESEARCH CONTEXT AND RESULTS

4.1 Waste picker organizations in Brazil

Estimates for the number of waste pickers in Brazil vary between 400,000 and 600,000, depending on the source (IPEA, 2013). The 2010 official census (IBGE, 2012) provides a number of 387,910 self-declared waste pickers. According to the census, almost 39 per cent were organized in associations, cooperatives or networks. 31 per cent of all waste pickers were female, however, women were the majority of organized waste pickers. The average age of waste pickers were 39 years, most of them were Afro-descendent (66 per cent) and a significant number (20 per cent) was still considered illiterate. Only 25 per cent of all waste pickers have completed their basic education (IBGE, 2012). Currently there are six networks and 95 organized groups in the metropolitan region of São Paulo (verbal communication MNCR, 2019).

Many waste pickers are part of the National Waste Pickers' Movement (MNCR), their main political voice, providing new formulations on waste management, disrupting existing assumptions and preconceived ideas about waste pickers. The MNCR was created in 2001, with the goal to expand inclusive solid waste management programs throughout the country and to integrate the struggle of waste pickers for self-determination and inclusion in the praxis of handling solid waste, which also means better access to funding and credit lines. The MNCR is also a member of the Latin American recyclers' movement (RedLacre) and the global network of waste picker organizations (GlobalRec).

In Brazil, the SSE has been instrumental in the formation of solidarity networks of waste picker enterprises, a recent phenomenon in the literature and in the praxis of waste picker organizations (Mota, 2018). Tirado-Soto and Zamberlan (2013) consider that networks of waste pickers are a strategy that allows them to access credit lines and complementary resources, as well as to improve administrative practices (Boeira, Campos & Ferreira, 2007), adding value to the recyclable materials (Aquino, Castilho Jr. & Pires, 2009). Not all waste picker cooperatives comply with the high standards of solidarity principles, their leadership might not be democratically established or they might not follow the rules established by the members of the cooperative (e.g. regular meetings, democratic election, etc.) and they are known as 'false cooperatives' among the waste pickers.

Waste picker cooperatives are not immune to the contradictions evident in the capitalist environment under which they have to operate. They struggle to make their decision-making processes democratic and transparent, in line with the values and principles of the SSE. While we also agree with Millar (2018) that formalisation not automatically implies a transition from a precarious to a secure workplace, and that many of the vulnerabilities and insecurities still persist in the cooperative environment, our research results emphasise the opportunities for emancipation and inclusion created through the organization of the waste pickers, through their cooperatives, networks and the social movement. There are different forms of formalization and 'inclusion', which do not comply with these standards (e.g. forced creation of cooperatives or associations after landfill closure) (Millar, 2018). In this article we refer to the 'cooperative model', which is based on autonomy of the workers and inspired by solidarity and cooperation. Other models of formalization might not be built on the same foundation and the risk is high that by merely transferring waste pickers into recycling plants, as a form of social inclusion, the paternalistic social state is just rearranged, as described by Hare (2019) for the case of waste pickers in Uruguay. We do not want to romanticize or idealize workers owned cooperatives and we recognize the many contradictions, challenges and conflicts that persist and we agree that "by including just a small part of the informal labour force, state initiatives [.....] risk re-enacting centuries-old forms of dispossession that rest on the fracturing and partial delegitimization of the working class" (Hare, 2019, p. 43).

4.2 Waste picker cooperative networks in the metropolitan region of São Paulo

This section presents the results from the in-depth interviews with leaders of six waste picker networks and the National Movement providing insights to their history, current situation and struggles.

Table 2 provides information about the networks participating in the study (Table 2). The first waste picker network in Brazil was Cataunidos, established in Belo Horizonte/Minas Gerais, in 2002 (Rutkowski, 2013). Out of the 6 networks participating in our study, Rede CATASAMPA was the first to be created in 2006, serving as an example for many of the other networks that followed in the region. COOPCENT ABC was the second network to become formally established in 2008. The other networks that took part in our research were formalized after 2012. The largest network is Rede Paulista, with 41 members and Rede CATASAMPA, with 20 members, while the others have between 7 (COOPCENT ABC) and 15 members (FEPACORE).

All networks revealed that their main objective for creating a network was collective commercialization, which would allow them to sell directly to the industry and get better prices. Collective commercialization at Rede Solidaria Catavida, e.g. happens in the following way: the cooperatives that are members call when they have a freight ready. Then the network collects the materials with their truck and centralizes in their storage space. With sufficient quantity the industry sends a truck to pick up the materials. All interviewees further listed many other reasons for forming a network, listed in Table 3.

Networks, such as Rede CATASAMPA, provide assistance to other cooperatives that are not yet well organized and still work under precarious conditions, helping them solve issues related to work space, lack of infrastructure, or precarious working conditions. Furthermore, as networks they are more respected, able to engage in public policy formulating and have bargaining power to negotiate contracts with their local governments. Sometimes even geographically distant groups are included in a network, demonstrating shared affinities and visions among these groups (Figure 1).

Mota (2018) has studied three waste picker networks in the state of São Paulo and also concluded that networks are generally born with a commercial focus and then evolve towards other purposes and benefits. Starting collaborations with a commercial focus seems to facilitate the consolidation of these groups. The relations between middle men and waste pickers in the Brazilian context are mostly of dependency, subjugation and exploitation. Pádua Bosi (2015) describes how waste pickers in Brazil have week bargaining power and are systematically exploited by the middlemen (atravessadores). Waste pickers frequently comment that they are cheated when it comes to weighting their materials and sometimes the middlemen even fix their scales to register a lower weight (Medina, 2007). Very differently, in the case of India middleman have been reported as important intermediaries, offering a kind of a safety net to waste pickers and also supporting their strive for recognition (Gill, 2010).

TABLE 2: Networks participating in the research.

Name of the network	Mandate	Geographic region	Affiliated groups	Waste pickers	Date of creation
Rede Paulistana	Commercialization for the two mega-recycling centers (megacentrais).	Municipality of São Paulo	41	1200	2016
Rede Solidária Cata-Vida	Collective commercialization. Solidarity networking.	Southwest of RMSP	14	800	2001
Rede Sul	Collective commercialization. Improve recycling quality. Support cooperativers in administrative, technical and legal issues.	South of MRSP and Campinas	13	800	2012
Rede Catasampa	Collective commercialization. Capacity building, Collective purchasing, service contracts.	11 municipalities (Guarulhos, São Paulo, Southern coast (Santos, Itanhaem, Mongagua)	20	750	2006
Coopcent	Collective commercialization. Political organization of the member groups.	ABC region	7	208	2008
Fepacore	Professionalization of the waste pickers and collective commercialization.	State of São Paulo	15	670	2013
Rede Verde Sustentável	Collective commercialization. Capacity building.	Santana de Parnaiba, Itapevi, Cotia, Taboão da Serra, Embu das Artes, Osasco, São Paulo	10	500	2007
		TOTAL	120	4928	
MNCR	Political organization of the member groups. Professionalization of waste pickers. Capacity building. Participation in policy making	National level	n.a.	n.a.	2001

Networks are configured as an important organizational arrangement in the face of the complex challenges presented by the recycling market. All networks had benefited from federal government funding through the Cataforte program, promoted by SENAES, the National Secretariat of Solidarity Economy (Secretaria Nacional de Economia Solidária) and destined for the socioeconomic inclusion of waste pickers. This program was incorporated in the National Development Plan PPA 2012-2015, under the thematic program 'Number 2067' on Solid Waste, and was instrumental in the rise of waste pickers. Fundação Banco do Brasil, a foundation linked to Bank of Brazil, was another important funding source, mentioned by the networks, to support many actions required for the socio-productive inclusion of waste pickers.

Several networks had initiated or promoted social and technical innovations among their member cooperatives. The network Rede Solidária CataVida, e.g. has set up a

TABLE 3: Waste picker organizations and networks.

Major benefits from creating a network	
Skill development	Cooperative administration skills
	Workers health and safety
	Public policy and waste management
	Challenges in material recycling (quality standards, minimizing the waste of resources, new materials)
Gender specific aspects	Generating gender awareness
Project development	Developing joint funding applications
Community outreach	Environmental education interventions
Knowledge exchange among waste pickers	Pricing, conflict resolution, legal compliances
Consultancy	Contracts with local governments, business and industry

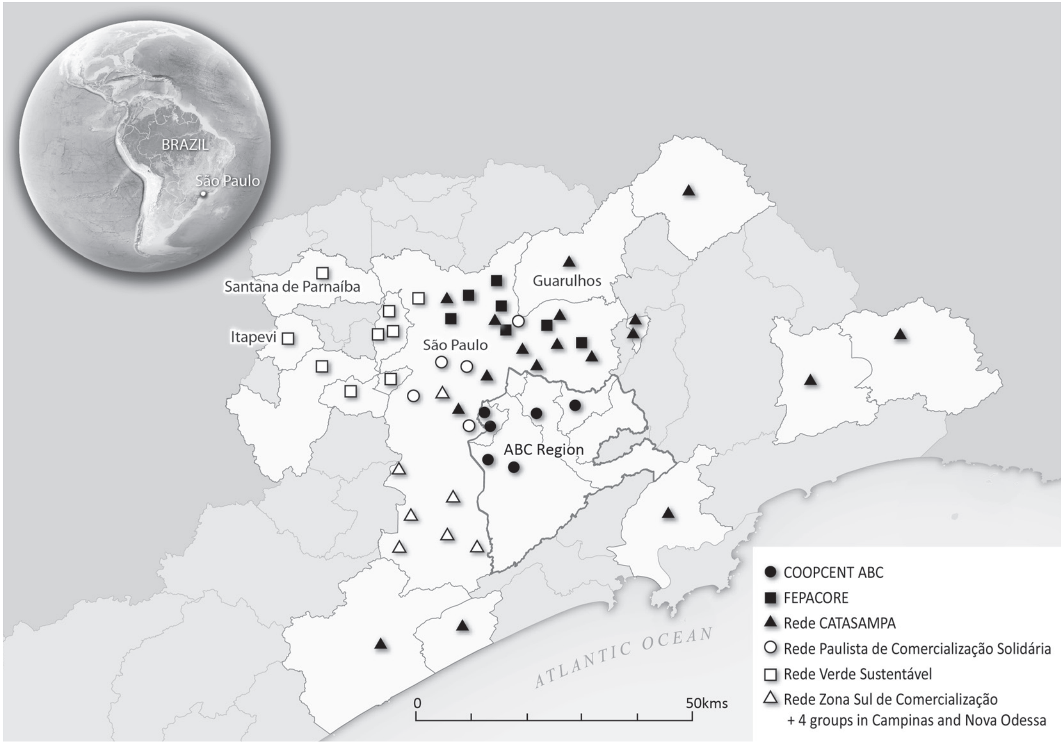

FIGURE 1: Networks participating in the research.

Polymer factory for plastic recycling and has adapted a process to transform cooking oil into soap and animal feed; benefiting all members of the network. Their leader reiterated the importance of waste pickers in adding value and not just collecting and separating materials. Yet, they mentioned that it is difficult to convince all members to invest in making products of higher value. We were told that occasionally cooperatives had left the network because they still preferred to sell to middlemen for a lower price, however receive the payment immediately. The interviews revealed that a downside of collective commercialization is that it can take up to 15 days to receive payment and not all cooperatives have enough cashflow to survive this waiting period.

Nevertheless, collective approaches to commercialization can improve earnings significantly. Rede Solidária CataVida for example sells white High-Density Polyethylene (PEHD) at R$ 1.80 to the industry, while middlemen are paying only R$ 0.50 to R$ 0.80. The largest challenges currently are the lack of recyclable materials, linked to increased competition from new small to medium-scale recycling industries (not waste picker cooperatives). Another bottleneck is the lack of working capital to be able to pay the cooperatives immediately for their materials contributed to collective commercialization (as briefly highlighted before).

Rede CATASAMPA provides public and private collection services, issuing records (certificados) based on the invoices received from the industry who purchases their sorted materials. In Mogi das Cruzes, Rede CATASAMPA services the municipal recycling program, Reciclamogi. The program survived the change in local government, demonstrating resilience and the ability of the network to provide high quality service. The waste pickers involved in the program receive a fixed value for collection and sorting, based on operational costs. The workers benefit from subsidized transportation, food, health coverage, holidays and paid annual leave. Their monthly income is approximately 1 minimum salary (R$ 1,045 or 209 US$) plus benefits. The creation of waste picker networks is a recent phenomenon and still lacks research. Our study confirms that networks are created to support and further consolidate waste picker cooperatives, providing them with capacity building opportunities, technical, administrative and financial training or services and facilitating collective commercialization to increase their income (Tirado-Soto & Zamberlan, 2013). Network leaders revealed that their networks support the cooperative members during contract negotiations with the city or with industry partners, strengthening their position. The network also just helps disseminate relevant information to the cooperatives (e.g. related to new legislation, price developments, capacity building and funding opportunities, etc.).

The literature shows that cooperatives involved in collective commercialization are better equipped for the challenges of the recycling market (Pisano et al., 2019). In addition, our research highlights that network articulation is not limited to providing a direct economic benefit. Bringing cooperatives together as a network has the potential to strengthen the social assets of the cooperatives, empowering them politically and economically, be it by selling directly to the industry, by providing services or capacity building activities, by exchanging their experiences and knowledge among each other, or by helping in the negotiation of contracts with local governments.

Waste picker networks are important organizations to deal with new challenges. These challenges are e.g. related to the volatile global recycling market and fluctuating prices, or the emergence of new materials, for which there is no market. The interviewees also recorded growing competition over recycling, with new entrepreneurs (not guided by SSE principles) arriving in this previously undervalued market and they recognized that organizing is critical for waste pickers to be able to participate in reverse logistics. According to the network leaders we interviewed, some cooperatives were facing internal management hurdles and also financial difficulties related to accumulated debts from unpaid fines, which puts them at risk of losing the agreement with the city hall. Many groups are not able to comply with the legislation and get fined for that. The leaders mentioned that the cooperatives always need to demonstrate to a city hall or company that they are competent and able to do the service. Sometimes, political change interrupts the pay for the collection service, leaving the waste pickers with lower income. There still seems to prevail a strong stigma against waste pickers among local government and continuous dialogue with Government is needed, to overcome prejudice and marginalization. Networks also mentioned that they are often understaffed to assist in policy making. However, they all reiterated the necessity for waste pickers to work together and find consensus, which will then strengthen the cooperatives. Some interviewees also were concerned with increasing the number of cooperatives affiliated to the network, to increase the collective power.

5. DISCUSSION: PARTICIPATORY WASTE GOVERNANCE IN THE METROPOLITAN REGION OF SÃO PAULO

The period of more inclusive waste governance and supportive public policies, in Brazil, coincides with the governments of President Luiz Inácio Lula da Silva (2003 to 2011) and President Dilma Rousseff (2011 to 2016). Since 2002, the profession waste picker (catador) has become legal. Several additional laws have been enacted since, in support of inclusive solid waste management and decent working conditions for waste pickers. Table 4 summarizes some of the public policies and actions, instrumental in changing the working conditions and creating opportunities for cooperatives to be included in the recycling chain.

One of the key driving forces for the socio-economic inclusion of waste pickers was the creation of the federal agency for Solidarity Economy (SENAES), in 2003, under the Ministry of Work and Employment. At the same time the Brazilian Forum of Solidarity Economy (Fórum Brasileiro de Economia Solidária, FBES) was constituted, with the objective to articulate and mobilize SE and to maintain a direct communication channel with SENAES. Since 2008, SENAES has narrowed the relations with MNCR, which allowed organized waste pickers to capture funding for infrastructure and education. Programs such as the Urban Solid Waste Program (Programa Resíduos Sólidos Urbanos) have helped organize waste pickers and have given them more visibility. In 2010, SENAES created the Pro-Catador program to integrate and coordinate the actions of the federal government supporting waste picker organizations, with actions to improve working conditions and expand selective waste collection, reuse and recycling through the inclusion of waste pickers. The fact that the federal law includes solid waste as part of sanitation has furthermore extended funding opportunities for this sector.

The 2010 Brazilian National Waste Management Legislation (PNRS) emphasizes selective waste collection and recycling and requires municipalities to integrate organized waste pickers (Besen & Fracalanza, 2016; Gutberlet, 2018). The PNRS has established several targets for the reduction of waste disposal at landfills, by 2014 (Brazil, 2010; Tavares Campos, 2014), most of these targets have not yet been met. In 2018, the formal material recovery rate from

TABLE 4: Brazilian Legislation supporting waste pickers activities.

Law/Decree/Action	Main objectives
Federal Law No. 5,764 of December 1971	Establishes the National Policy on Cooperatives
In 2002, the Ministry of Labor and Employment creates the professional category: catador collector of recyclable materials and includes it in the Brazilian classification of occupations (CBOS), under the Code 5192-05 (MTE. Classificação Brasileira de Ocupações)	Legal and formal recognition of the occupation of collector of recyclable materials, setting parameters for the development of this activity
Decree No. 5,940, 25 October 2006	Requires public institutions to separate and donate the recyclable fraction of their solid waste to recycling associations and cooperatives
Federal Law No. 11,445, of 5 January 2007: National Policy on Basic Sanitation	Authorizes the municipalities to hire recycling associations and cooperatives to collect, process and market recyclable or reusable municipal solid waste
Federal Law No. 12,017 of August 2009 and published the annex VII of the D.O.U, 13.8.2009, extra Edition	Changes the law of the budget guidelines, allowing the direct transfer of resources to cooperatives, without intermediation of municipalities or social organizations of public interest
Federal Law No. 12,305, July 2010 and its regulation through Decree No. 7,404 of December 2010	Establishes the National Solid Waste Policy and creates the Inter-ministerial Committee of the Brazilian solid waste Policy and the Steering Committee for the implementation of the reverse logistics systems
Federal Decree No. 7,405, 23 December 2010, published in D.O.U. of 23 December 2010	Institutes the Pro-Catador program. It creates the joint inter-ministerial Committee for social and economic inclusion of the collectors of reusable and recyclable material
Federal Law No. 12,690, of 19 July 2012 published in D.O.U., 20 July 2012	Rules on the organization and functioning of Workers' Cooperatives

municipal selective waste collection programs in Brazil is estimated at 7.3 per cent of dry, recyclable household waste (Brazil, 2019). All formal programs rely mostly on the integration of waste picker organizations. The census identified a total of 27,063 waste pickers working in 1,232 associations or cooperatives, in 827 municipalities. Many waste pickers still operate informally and are often not integrated in local selective waste collection programs.

In 2018, 38.1 per cent of the 3,468 Brazilian municipalities that had participated in the survey (Brazil has a total of 5,570 municipalities) claimed to have a municipal selective collection system in place, with different levels of coverage (Brazil, 2019). With the advent of increased pressure to close landfills and quickly solve the surmounting problem of solid waste accumulation, public private partnership (PPP) funding waste incineration and waste-to-energy has recently risen as a new threat to the livelihoods of waste pickers (Rodrigues, Azevedo & Gutberlet, 2015; Gutberlet, Bramryd & Johannson, 2020).

The challenge of reducing the generation of waste and of managing the over 180,000 tons of municipal waste that are produced on the national level, every day in Brazil, requires integrated public policies that are articulated between the main actors and involve efficient management systems that incorporate transparency and accountability geared towards building co-responsibility among citizens and other waste generators (Reis, Conti & Correa; 2015; Jacobi & Besen, 2017). The following figure (Fig. 2) shows the diversity of possible actors involved in waste management.

Dias and Samson (2016) showcase some experiences of transformative modes of waste governance in various cities in Brazil. The research demonstrates how important government funding (grants, microcredit) is, acting as a cushion to fall back on, particularly during times of instability and low income. The study shows how the lack of such programmes increases the vulnerability of waste pickers. The authors also identified main factors that support waste pickers in different contexts of formal integration into solid waste management, as well as, the various roles all levels of government play in inclusive waste management.

The discussion on public policies unveils gains and setbacks and refers to the necessary adaptation of the political actors to constantly changing situations. In some cities, such as Belo Horizonte, the formal integration of waste pickers has persevered over a long period of time (21 years), which according to Dias and Samson (2016) is mainly due to two factors: 1) the power of the organizing process in the city (most member-based organizations are active participants in the national movement of waste pickers and also participate in several socio-governmental platforms), and 2) the commitment of dedicated officers at the municipal sanitation agency who pressure the mayor's office to continue with inclusive solid waste management policies. Often these programs do not survive in the absence of local government support and the political will of decision makers.

Diadema, in the MRSP, was the first municipality to create, in 2005, a law that establishes remuneration for the service provided by waste pickers (municipal law 2.336/2004 and Decree 5.984/2005). However, this municipal law is currently not respected given the lack of political will of the current government and the weak participatory governance structures (Jacobi & Besen, 2011; Gutberlet, 2015). Waste pickers have been continuously demanding for the implementation of the law but without success. Other cities, like Ribeirão Pires, São Caetano do Sul, Guarulhos and Mauá in the metropolitan region of São Paulo as well as Ourinhos, Ribeirão Preto in the interior of São Paulo have implemented and maintained their service payments. The city of Ribeirão Pires pays R$ 527 (US$ 105) per ton of separated material to Cooperpires and Ourinhos pays R$746 (US$ 149) per ton to the cooperative Mãos Dados. In both cases the networks (Coopcent-ABC in the case of Ribeirão Pires and Rede Anastasia in the case of Ourinhos)

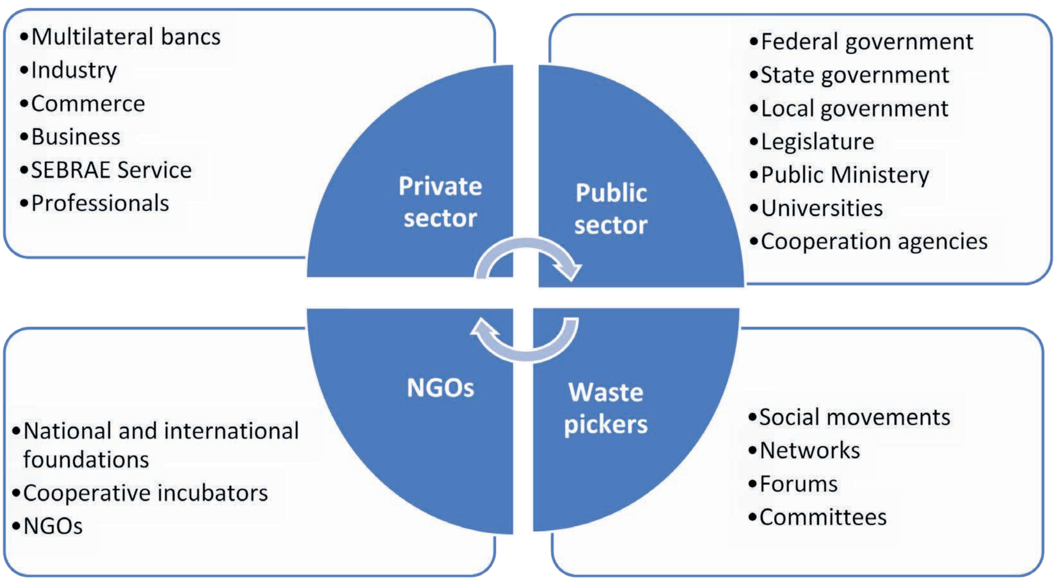

FIGURE 2: Actors involved in the socio-productive inclusion of waste pickers.

were instrumental in the negotiation between the cooperative and the municipality, establishing these contracts. Similar ideas are tested out by municipal governments in Bolivia, Argentina and Colombia, with moving towards the integration of waste pickers and the payment for separate waste collection and recycling services (Tavares Campos, 2014; Murakami et al., 2015; Rutkowski & Rutkowsky, 2015; Gutberlet, 2016). The monetary service recognition makes a significant difference in the income of the waste pickers.

The research has helped understand the role of collective commercialization as important starting point for better articulation between waste picker organizations and local governments, as also described by other authors (Azevedo et al., 2018; Pisano et al., 2019). Collective commercialization is a concrete and tangible goal opening the dialogue among the groups; however, there needs to be enough cash flow to immediately pay the cooperatives for their work. Those networks that have succeeded today economically have also been able to diversify their benefits, supporting them in their relations with government and industry. Our results show that expanding the political role of waste pickers through networking is important to promote inclusive environmental waste governance. Our examples demonstrate how essential the support provided by the SSE is, allocating funds to waste pickers to better equip them with knowledge and infrastructure.

Some of these tangible socio-economic advancements of the SSE and participatory governance in Brazil are challenged by recent changes in the federal government. With President Temer, between 2016 and 2018, the National Secretariat for Solidarity Economy (SENAES) was reduced to a Sub-secretariat in the Ministry of Labour, receiving significantly less funding, thus restricting the scope and action of the SSE. The gradual dismantling of SENAES further continued with the present government (Bolsonaro) extinguishing the Ministry of Labour and SENAES. The SSE in Brazil has now become a 'directory board', allocated within the recently created Ministry of Citizenship (provisional measure 870/2019). This new structure expresses both the extinction of SENAES and the changes in policies, associating the SSE with the policies of social assistance and not any more with 'productive inclusion'. The withdrawal of the state from SSE has weakened the process of socioeconomic inclusion of vulnerable actors, such as the waste pickers. The advances detected in our empirical study are on verge of being lost, due to the recent changes in federal waste governance, marked by authoritarian, top-down decision-making and the dismantling of inclusive and participatory governance structures, including the SSE.

6. CONCLUSIONS

Waste progressively gains more attention among scholars and policy makers, and new actors and organizations have emerged over time. This article introduces a case study involving several waste picker networks located in the Metropolitan Region of São Paulo and analyses the cases where a collective approach to waste management has changed waste governance.

Social and technological innovations since the late 1990s have introduced important changes to waste governance in Brazil, and new ways of organizing waste pickers into cooperatives, associations and networks, allowing for new models in waste governance to evolve. The emergence of large numbers of waste picker organizations and networks is also linked to the expansion of the Social and Solidarity Economy in Brazil, since the early 2000s. SSE allows for diverse socioeconomic practices to emerge, generating jobs and opportunities for practicing democracy. Similar levels of organization are also developing in other countries in Latin America, in Asia and in Africa in particular.

SSE proves to be a viable development model as there are currently about 21 thousand solidarity economy enterprises in Brazil, involving more than 1.5 million members, of which almost 7 thousand are cooperatives operating in 13 branches of activity, with 14, 6 million members (OIBESCOOP, 2019). Approximately 10 per cent (40 to 60,000) of all waste pickers in Brazil are organized and are members in cooperatives and associations, making up a total of 1,100 waste picker organizations in the country (Brazil, 2011). Many have organized as networks, allowing them to collectively sell their materials and to coordinate other actions that enhance their capacity in waste management. Based on this development model, the objective of the cooperative is to generate work, income and to improve the working and living conditions of its members. Waste picker cooperatives are self-managed and practice solidarity, e.g. by including also vulnerable and underprivileged individuals (e.g. suffering from mental or physical health or from addiction problems), who otherwise would not have paid work. Organized waste pickers demonstrate that there is honour in the work they perform and that their workplace democracy has not given in to what Steward E. Perry (1978) has called 'the iron law of oligarchy'.

The Brazilian experience illustrates how a prolonged period of progressive social policy can help built a robust social and solidarity economy, creating unseen opportunities for a sector which was previously excluded. Yet, the example also shows how fragile institutional frames are, depending on who is governing; as we observe with the current political situation, where SSE institutions have suffered neoliberal dismantling. In particular, we understand that these issues also have international parallels, particularly, related to conflicts associated with economic crises, unemployment, environmental challenges and the exacerbation of neoliberal and austerity pressures. In this sense, we believe that there is a significant contribution to the discussion on the role of this type of grassroots social organization in confronting these political forces in a global context, where the level of organization of the waste pickers and the strength of the cooperation with local governments will determine the resistance and persistence of inclusive waste management arrangements.

Particularly since 2010, waste pickers in the MRSP and beyond have begun organizing into networks, signing contracts with city administrations and expanding their activities into co-production of urban services including environmental education and reverse logistics programs. Despite many difficulties and constraints attached to co-production in waste management (particularly due to working with vulnerable populations), there are gains for city administrations, communities, the waste pickers and the environment. The obvious gains from waste co-production are (1) social: building more inclusive communities (reducing unemployment providing low barrier jobs, particularly for women and vulnerable individuals) and (2) environmental: contributing to urban sustainability (increasing resource recovery and the circular economy). Innovative experiences and collaborative arrangements as discussed in this paper demonstrate these opportunities and multiple social, economic and environmental benefits for cities and their populations.

The most successful cases of cooperative recycling are those with co-production arrangements, where local governments sign contracts with organized waste pickers to pay for the collection and separation services provided and were reverse logistics programs are in place. Public policies formalizing these inclusive arrangements are crucial to guarantee a successful and lasting programme, but they are not enough. The organization into networks provides the groups with a stronger political voice, pressuring municipalities to include them in waste management. A shift towards integrated, collaborative environmental governance is vital to make the co-produced service work.

We have given examples from the metropolitan region of São Paulo for singular technological innovations and new forms of waste governance involving membership-based waste picker organizations and have shown the relevance and roles of waste picker networks in supporting their members, increasing their income (through collective commercialization but also by supporting local groups to establish service contracts with local governments) and offering experiences to skillfully and successfully run a waste picker cooperative. The Brazilian case highlights how the SSE with waste picker organizations (cooperatives, networks, the National Waste Pickers' Movement) have been able to develop an institutional and policy frame in support of waste pickers. However, we also learn that there are setbacks, specifically tied to prevailing neoliberal political contexts.

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Extra contents

COLUMNS AND SPECIAL CONTENTS

This section comprises columns and special contents not subjected to peer-review

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Info from the global world

INFORMAL RECYCLING IN VANCOUVER: BINNERS' CHALLENGES AND OPPORTUNITIES

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The Covid-19 pandemic, has emphasised the need to consider the interconnectedness of our planet, and the importance of highlighting new, and previously underrepresented perspectives on global waste management issues. The new corner, "Info from the global world" wants to collect thoughts and impressions from different parts of the world, with the aim of contributing to a more innovative and inclusive waste management studies discourse. The column will promote cultural intersections on issues affecting circular waste management, environmental protection and human health. We will highlight contributions from diverse expert authors who discuss, among a number of topics, how gender inequality and environmental racism can be combated through truly sustainable waste management and how the circular economy and Sustainable Developing Goals can contribute to combating poverty and mitigating waste inequalities.

The second issue of the Column features the work of Dare Sholanke and Jutta Guterblet of the University of Victoria, Canada. Their discussion centres the lives and livelihoods of informal recyclers in Western Canada- a topic which has traditionally been contextualised within Global South settings. Sholanke and Guterblet's reflection is both empirically interesting, as they provide a vivid snapshot of the quotidian vulnerabilities of this group, but also conceptually valuable, as the theoretical framework they utilise could be readily adapted for scholarly use within other contexts. Their conclusions challenge the inclusivity of local waste management systems for informal recyclers, and the further recommendations that continue to come out of this project should be of great international interest.

Maria Cristina Lavagnolo, *University of Padova, Italy*

Informal recycling is mostly studied in the context of global south cities. This article explores a North American perspective on the topic, demonstrating major shared challenges that these vulnerable populations encounter in the course of their daily endeavour – recovering recyclables or binning, as the activity is termed at the West Coast of Canada. We draw from Mitchell and Heynen's (2009) concept of "geographies of survival" to explore the lived experiences of binners in Vancouver, B.C. Using a mixed methods approach (in-depth interviews, photovoice and survey), we investigate the struggles and pathways of survival that binners in Vancouver, navigate daily. We also examined the roles of a grassroots innovation in enhancing binners' livelihood strategies and concluded that grassroots innovations are integral in enhancing binners survival in an arduous waste management landscape, thus, promoting an inclusive waste management system.

Introduction

In North America's seemingly organized waste management system, the informal sector – often referred to as binners, canners or diverters at the Canadian West Coast, are generally excluded from both waste management (selective waste collection for recycling) and waste governance (decision-making processes). Meanwhile, recovery of primarily beverage containers for refund deposit is a major livelihood strategy of these individuals, particularly those who do not receive government social assistance, and suffer from homelessness, addiction or mental health. The denial of the binners' right to collect these materials through the enforcement of bylaws by city officers has continually impacted their ability to survive, leading to tensions between binners and other stakeholders in the waste management system. To explore the many challenges of binning, we used Mitchell and Heynen's concept of geographies of survival, which they define as the "spaces and spatial relations that structure how people may live and whether they may live" (2009, p. 611). The concept was drawn from Lefebvre's notion of the 'right to the city', understood broadly as the right to urban life with a decent quality of life and specifically addressing the right to habitat and inhabit for the most dispossessed, the homeless, ethnic minorities and people in situation of poverty (1996). Both concepts are helpful in pointing out the inequalities and injustices inscribed in the urban by the capitalist system suggesting a necessary change towards more participatory, inclusive and community-based governance. The discussion on these notions is particularly relevant to addressing the pressing urban social and environmental challenges that demand more sustainable solutions. Social actors that have previously been excluded need to be invited into the

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dialogues and decision-making processes, contributing with their experiences, social innovations and alternative forms of city-making.

Homelessness has evolved into a significant problem in most North American cities. According to Statistics Canada, in 2014, more than 235,000 Canadians experienced homelessness within the year and "approximately 2.3 million Canadians (representing 8% of the overall population aged 15 and over) reported that, at some point in their life, they had to temporarily live with family, friends, in their car, or anywhere else because they had nowhere else to live". In part this situation is the results of the withdrawal, during the 1990s, of the federal government's investment in affordable housing, the reduction in social benefit (pensions and social assistance) as well as declining wages (e.g. the minimum wage has not kept up with inflation in any jurisdiction in Canada). These developments have placed many Canadians at risk of homelessness. In 2017's local homeless count, about 15% of the homeless in Vancouver responded that binning was their main income source (The Homelessness Services Association of BC, Urban Matters, and BC Non-Profit Housing Association, 2018). The majority of binners in Vancouver experiences the absence of the right to habitat (a place to reside that can be called home). As binners are often banned from using public spaces, the same could be said about the right to inhabit (to exist or be in a place with decent quality of life). Too often binners are still harassed, stigmatized or even fined for working with the recovery of recyclables. The deprivation of rights, such as the right to collect recyclable materials, has been a major nightmare for informal recyclers globally, including North America. Another fact not well known is the contribution of binners to the local community and to the environment. In Vancouver, the binners project alone, working with 168 binners, diverts about 7,000 kg of waste per month, a number which has been growing by 10% every year (The Binners' project, 2020).

Informal Resource Recovery

In response to the many challenges involved in binning in the global North, we examined the roles of the Binners' Project, a Vancouver-based social grassroots innovation, in empowering binners to navigate the treacherous terrain of the waste management system. We understand grassroots social innovations essentially as democratic, contextual and political processes whereby ideas, products, processes, alliances, public policies or methods are designed and developed, involving vulnerable and/or marginalized groups such as waste pickers and binners for the purpose of creating more inclusive recycling programs and opportunities for social change. The Binners' Project, a community-based organization that exist primarily to assist less privileged individuals who lack access to the superior form of rights often through the development of novel ideas and innovations that challenge the neoliberal policy and urban inequalities through different forms and levels of empowerment (Arruda, 2008). In our study, we were seeking answers to the following two questions: what are the challenges for the binners in Vancouver and how has the grassroots organization helped address these issues?

We adopted a community-based research (CBR) design to foster active involvement and collaboration with members of the binners community (the Binners' Project and the United We Can Bottle Depot, both in Vancouver), who served as co-researchers. A mixed methods approach was applied, which included in-depth interviews, a survey and a photovoice engagement workshop. In-depth interviews were conducted with five stakeholders to identify the binners' contributions to municipal waste management and/or governance, experiences and challenges encountered while working and how grassroots innovation could help to solve these problems. These stakeholders included two City Government Officials from the Engineering Department, one Manager of the United We Can bottle depot, one Manager of Encorp Pacific, and the Manager of the binners' grassroots initiative (the Binners' project). Interviews were recorded, transcribed and then manually analyzed using thematic content analysis to identify themes, sub-themes, and patterns that emerged from the text. The survey was applied by six binners who had previously been trained as co-researchers, to a total of 60 binners within the City of Vancouver The survey included a wide range of questions which helped provide insights into the demographic and socio-economic situation of binners. In addition, a photovoice workshop was held with some of the binners. Photovoice is widely applied as a research method to promote democratic knowledge development where participants are actively involved in the research process particularly in terms of collecting data relevant to their lived experiences. Participants were compensated with an honorarium for the time involved with training and taking the pictures. An iPad was provided for the participants to take five to ten photos. Once we had all photos, a focus group was held to discuss the content of the photos. We created large posters from the photos which were then exhibited at a community recycling event, the 'Coffee Cup Revolution' – an annual event where binners recover disposable coffee cups within the Downtown Eastside of Vancouver. This is an important annual event organized by the binners' Project to create more awareness about disposability and recycling.

The Challenges of Binning

Drawing from our data we explored the geographies of survival of binners in two distinct facets, which included the challenges binners are facing and the roles the Binners' Project has played in empowering them to navigate through these challenges. One of the major challenges mentioned by binners is the reduced access to recyclables. This problem started already in June 2005 after the city of Vancouver made a proposition for the enforcement to lock larger dumpsters (at least one cubic yard in size) (City of Vancouver, 2018). Not only has the locking of dumpsters limited binners' right to access recyclable materials, but also has it threatened their right to survival. The survey revealed that about 60 percent of the binners were unemployed and 34 percent did not receive any government social assistance. Among this population, over 60 percent of the them earned less than $20/day, which would barely suffice to purchase two meals, not to mention other daily expenses. Reducing the access to refundable beverage containers limits their access to a fundamental resource that supports their livelihoods.

Occupational health hazards pose another challenge for binners. Highlights from the survey revealed that most binners frequently experience cuts (over 50 percent) and physical soreness (about 50 percent) during their work. This result was reiterated by several other studies that showed that binning exposes these individuals to several occupational hazards, which include headaches, soreness, cuts, infections and musculoskeletal problems among others (Tremblay, 2007; Uddin & Gutberlet, 2018).

Tension between binners and government officials or private company workers, essentially due to the bylaw that prohibits the collection of recyclable materials from the curbside, was reported as a major problem. The curbside is still a contested space of ownership with regards to the recovery of recyclable materials, as most governments contract out waste collection, thus transferring ownership of these materials to private companies which then results in the exclusion of informal collection and the criminalization through the bylaw enforcement. While the municipal government and the private sector consider waste an objects to be managed or governed, binners consider waste as a common pool resource with market value (Moore, 2012) to sustain their livelihoods. These variations in rationale over ownership claims are a significant source of tension for these individuals, as demonstrates the quote from some binners: The city union has pretty much screwed that up to saying no, this is ours to do. This is all our job and you guys stay out of it (M. binner). They (private companies) laugh at us… they don't even want to talk to us. We're they're their biggest pain in the ass because we are a competition. They don't give a shit about anything… But they cut into their bottom line and they make more money by giving people fines for fucking it up (M. & J. binners). For long waste picker collectives in the global South have argued for waste to be recognized as an urban common pool resource. Waste pickers contributing significantly, yet hidden, to waste management, to the generation of jobs and income among the poor and to reducing carbon footprints of cities, defending their right to produce, use, and appropriate waste (Zapata & Zapata Campos, 2014).While these issues are widely discussed for the context of the global south these very similar questions are hardly addressed in the global North.

In response to these challenges in Vancouver, the Binners' Project was established in 2014 by Ken Lyotier – a former binner and the founder of United We Can bottle depot – to create economic opportunities, destigmatize binning, and promote social cohesion among binners. Strategically located in the heart of the Downtown East Side (DTES) of Vancouver, the Binners' Project is a grassroots organization led by a core group of binners and supported by a steering committee, staff, special advisors and volunteers, all working collaboratively in making decisions on the direction of the project and developing initiatives to enhance binners' survival.

Based on our findings, the organization has developed several strategies to empowering binners. These include professionalization – playing the role of an advocacy between the binners and the government and also by providing a means of identification such as identity cards and uniforms with their logo crested thereon, similar to what waste picker organizations in the global South are already doing (Gutberlet, 2008). One of the government officials noted that: …professionalizing binning is the easiest route to overcoming the negative perceptions that binning has among some residents" (SF). The Binners' Project seeks to empower binners by destigmatizing their work through public campaigns/engagement and waste recycling education at events such as the Coffee Cup Revolution. This has helped distinguishing binners as professional informal recyclers and part of a formal organization. Training and skill development is also offered by the organization to build binners' capacity to contribute and perform optimally in the formal waste management setting, leading to an increased engagement and interest from the city government as well as other private companies and also increasing the access to recyclable materials. As one of the government

FIGURE 1: Littering resulting from locked bins.

FIGURE 2: The abode of the homeless.

FIGURE 3: The Binners' Project Team.

officials noted: So, really, the challenge is if there isn't a binners project, it's very difficult for us to work directly with the community (S.F).

Moving forward

The concept of the geographies of survival (Mitchell and Heynen, 2009) has served as a lens to examine the challenges informal recyclers are facing in the global North, based on a case study conducted in Vancouver. Grassroots social innovations play a key role in improving binners' survival strategies. We presented several challenges which mostly include reduced access to recyclable materials through locked bins (Figure 1), occupational health hazards, stigma and tensions with other stakeholders (Figure 2). We highlighted the strategic approach by the Binners' Project in resolving these challenges (Figure 3). Our finding strongly supports the notion of inclusive and participatory waste management. Grassroots organizations are critical in promoting positive change in the lives of informal recyclers across the globe, as well as benefitting the environment by diverting recyclables into the circular economy. Further research is necessary to expand these findings. Another example where binners have organizing is Montreal, Canada. Here the initiative started out as Projet Consigne, created in 2004 by two volunteers (Marina and Marica). This project offered free collection points for refundable containers within businesses of downtown Montreal, and gradually expanded to rise to the cooperative, called Les Valoristes, in 2012. Very recently another initiative was created in Victoria, called the Diverters Foundation, reflecting the distinguished local name diverter for binner. All these projects draw the public attention to the social benefits associated with improved deposit-refund systems, among other benfits. We highly recommend that similar organizations be established in other cities and regions where informal recyclers are still largely unrecognized and stigmatized.

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DETRITUS & ART / A personal point of view on Environment and Art

by Rainer Stegmann

Artists seldom provide an interpretation of their own work; they leave this to the observer. Each of us will have his/her own individual view of a specific piece of art, seeing different contents and experiencing a range of own feelings and emotions. Bearing this in mind, I created this page where you will find regularly selected masterpieces from different epochs and I express my thoughts on what the work conveys to me personally. My interpretation will refer specifically to the theme "Environment". Any comments or suggestions regarding this column should be addressed to stegmann@tuhh.de.

RENÉ MAGRITT / The Lovers, 1928.

The term Surrealism comes from French language and means "above realism". In surrealism artists did not only depict reality in their paintings but also dreams, the unreal and fantastic and combine these different worlds. To a certain extent this art direction was a reaction to World War I. René Magritt (1898-1967) is one of the main representatives of surrealism.

I selected his painting "The Lovers", which exists in several versions. I chose it because I think we also live in a somewhat surrealistic time, a time that human society only imagined in movies and books but not in reality.

When I saw the masked faces on Magritt´s painting immediately Corona came into my mind. A man and a woman in love are separated by the pieces of cloth that cover their faces. The prints of the faces on the pieces of cloth and the way the heads are positioned show that the persons are in love but cannot come closer to each other, the cloth separates them. The analogy to our situation during the Corona pandemics imposed itself on me and I might "translate" René Magritt´s title to "Love in times of Corona", as a symbol of the difficult time when people are separated and cannot come together.

One may also find an analogy to living in times of climate change, which to a certain extent is also surrealistic, the effects as changing weather conditions, rising ocean water tables, intense melting ice in the pole areas are real but the relation of human action to climate change –although they can be scientifically explained- remains surreal, unreal.

When looking not only on the faces but on the entire painting of Magritt I see light on the two faces and a partly blue sky with fair weather clouds, which give a kind of mellow and not a depressing atmosphere. May be we can also find some "positive" effects of Corona pandemics. Especially during the first lock down in spring, nature benefitted from less tourists, fewer airplanes in the air, emptier roads, less consumption. There were many reports about the recovery of nature, of cleaner air and water (e.g. dolphins in the channels of Venice, cleaner air in big cities, less NO_2 in the atmosphere above China). I was surprised how fast nature in some areas recovered and this gives hope and should encourage all of us to reduce planet´s CO_2 footprint and strive for sustainability. What we can learn from Corona pandemics is the necessity to take drastic measures now; otherwise we will not succeed.

In order to make in the future the pieces of cloth around our heads redundant new pandemics have to be avoided. Transforming human society to sustainability also by implementing animal welfare, avoiding pollution and increasing biodiversity will help to reduce the chances for new pandemics on our globe. In a healthy environment virus has less chances for finding appropriate living conditions.

Next issue: In my next column I will present Paul Cézanne´s beautiful painting of the landscape of "Anvers du Cote´du Valhermeil". Paul Cézanne (1839-1906) is the pioneer and one of the main representatives of expressionism.

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