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A REVIEW OF RECENT TRENDS TO INCREASE THE SHARE OF POST-CONSUMER PACKAGING WASTE TO RECYCLING IN EUROPE

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ABSTRACT

Through the Circular Economy Package, the EU aims to reduce material imports and environmental impacts of waste management. Ambitious recycling targets should help to achieve these objectives. As a response, the waste industry developed technical and organizational solutions that can intervene at different stages of a waste management system in order to increase the recovery of recycling materials, starting with improved separate collection over the use of modern material recovery facilities up to the enhanced treatment of waste incineration ashes. The large question is how each of these options can contribute to increase recycling rates to achieve a circular economy. By reviewing case studies as presented in the most recent literature published since the year 2010 from European countries on the recycling of post-consumer packaging waste of glass, metals, paper, and plastics, this study contributes to answer this question. In the first stage, the review found 644 articles matching with the search terms and published since the year 2010. Of these, 45 remained for an in-depth analysis, since 599 did not present case studies as defined in the scope of this article. The articles reviewed provide a good overview on the state of knowledge on increasing recycling of post-consumer packaging waste by improved separate collection, material recovery facilities, and waste incineration bottom ash treatment. Additional information and case studies, however, are required, particularly large-scale experiments to test new separate collection systems, large-scale tests with different feedstock material at the most-modern material recovery facilities and waste incineration bottom ash treatment plants.

1. INTRODUCTION

The EU largely depends on imports of raw materials and consumer goods (Bruckner et al., 2012). The wastes generated from the consumption of these raw materials and consumer goods are only partly recycled, while the bulk is incinerated or landfilled (Pomberger et al., 2017). This leads to several environmental problems such as global warming and a reduction in ecosystems' quantity and quality. To reduce both, raw material imports as well as environmental impacts due to production of raw materials and consumer goods as well as disposal of wastes, the EU launched the circular economy package (CEP). The CEP, which consists of several directives, ordinances, and strategy documents, defines not only objectives, but also targets and measures to achieve a circular economy. An important target within the CEP is the achievement of recycling rates for selected recyclable materials in construction and demolition waste (Lederer et al., 2020), municipal solid waste (MSW) and particularly post-consumer packaging waste (PCPW) (Fellner et al., 2018; Fellner & Lederer, 2020; Tallentire & Steubing, 2020). For many countries in the EU, the fulfilment of the recycling rates will pose a large challenge, particularly since the definitions for their calculation were tighten (Weißenbach et al., 2020). It is clear that the CEP targets for recycling rates of PCPW can only be achieved if all sectors in the society will contribute to fulfil this task (Korhonen et al., 2018). For instance, the primary production sector (agriculture, forestry, mining) can increase the prices of primary raw materials, which would put secondary raw materials derived from waste recycling into a more competitive position. Also, the production sector can produce PCPW of longer durability and easier to recycle. Furthermore, the service sector can avoid the consumption of raw materials and PCPW generation by business models that promote multi-use instead of single use of packaging materials (Kalmykova et al., 2018). Despite all of these options the different sectors have, however, it is clear that the

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MSW management sector will also play a complementary role for achieving the recycling rates as defined in the CEP. To do so, the MSW management sector has several adjustment screws to turn, starting with the separate collection of PCPW (Mwanza et al., 2018), treatment in terms of sorting of mixed and separately collected PCPW (Cimpan et al., 2015), and finally the recovery of unburnable PCPW from the residues from MSW incineration, particularly bottom ash from grate incinerators and bed ash from fluidized bed incinerators (Bruno et al., 2021; Šyc et al., 2020).

This article reviews case studies from these stages of MSW management that enable the fulfillment of the CEP recycling rate targets. The research questions are:

What is the current status of MSW management in the EU with respect to the achievement of recycling rate targets as defined in the CEP for packaging wastes?

Which options are available at each stage of municipal solid waste management to increase the recycling rates of post-consumer packaging wastes?

How do they contribute to achieve the CEP recycling rate targets in the EU?

To answer these questions, first the scope of this work is defined in section 2, together with the review methodology. After presenting and discussing the results in section 3, a conclusion is presented in section 4.

2. METHODOLOGY

2.1 Scope definition

2.1.1 Product life cycle stages considered

Beside the selection of waste materials to be considered, the life cycle stages for options to achieve the CEP recycling rate targets covered in this article are restricted, namely to the MSW management sector. Other sectors such as primary raw materials extraction and processing (raw materials phase), design and production of products (design & production phase), and the use of these materials as consumer goods (use phase), are not covered, despite their relevance in achieving a circular economy (Kalmykova et al., 2018). Within the MSW management sector, the focus is on 1) the separate collection of recyclable PCPW (collection), 2) the sorting of PCPW containing MSW streams in material recovery facilities (pre-treatment MRFs/sorting), and 3) MSW incineration and the thereof produced bottom ashes (final treatment). Figure 1 shows the considered stages in the life cycle of a product.

2.1.2 Time period considered

Recycling is practiced by humans since the very beginning of their existence on earth. In modern societies, recycling gained much attention not only in the course of industrialization or in times of raw material shortages be-

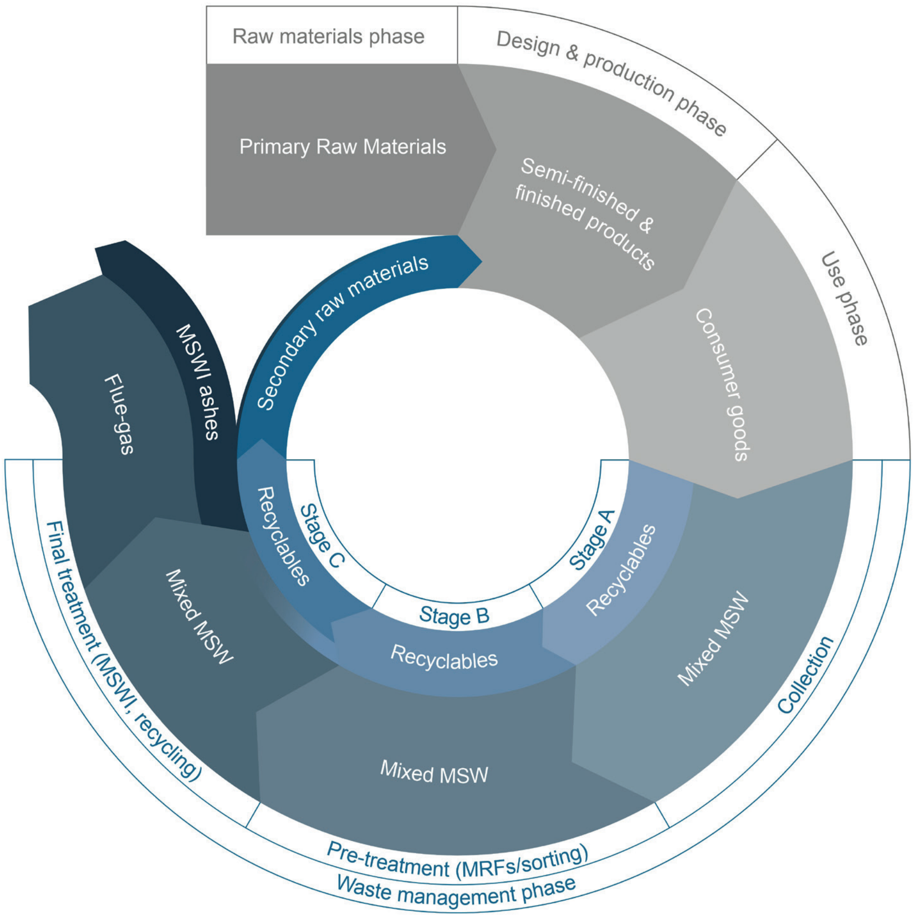

FIGURE 1: Stages in a products life cycle including the waste management phase (blue). Design by solo-ohne© (https://solo-ohne.com/).

tween the two world wars, but also with the emerging environmental discourse in the 1970ies (Keller, 2009; Strasser, 2000; Wilson, 2007). However, since the CEP is a relatively young concept and a lot of progress was made during the last decade, we consider in this article only recent developments. In detail, this means that only studies published between the years 2010 and 2021 were considered.

2.1.3 Geographic area considered

Concepts like the Circular Economy were developed in several world regions or countries at different times. Moreover, the EU's CEP spilled-over to countries and regions outside of the EU. However, since only EU member states are bound by the legal documents contained in the CEP, this article first covers case studies from the EU27 member states. In addition, countries with traditional and legislative bounds in the recent past and future, particularly the long-time EU member UK, and the EFTA states Island, Norway and Switzerland, were included.

2.1.4 Waste materials considered

The main focus of the article is post-consumer packaging waste (PCPW), usually contained in MSW. Other wastes of relevance such as agricultural waste, mining waste, construction and demolition waste, industrial waste, or waste from waste water treatment, are not covered. The waste materials considered are the quantitatively most relevant PCPW wastes for which a recycling rate target in the EU's packaging directive exists, namely aluminum, ferrous metals, glass, paper & cardboard, and plastic (EU, 2018). Wood packaging is not covered, due to its comparatively little quantitative relevance in packaging.

2.1.5 Type of studies considered

A large number of different types of studies on the CEP were published, including original research articles, conceptual works and some excellent review articles, for instance by Assi et al. (2020), Astrup et al. (2016), Blasenbauer et al. (2020); Cimpan et al. (2015), Dou et al. (2017), Rousta et al. (2017), Šyc et al. (2020), Verbinnen et al. (2017), Xevgenos et al. (2015), and Zhu et al. (2021). Since the aim of this article is to highlight original research articles, review articles are not covered. Furthermore, studies that provide economic, environmental or social impact assessments of circular economy scenarios that are not based on primary data on how to increase recycling rates, are not considered in this article. Finally, only original research articles which underwent an internationally considered review process by at least two independent reviewers, are selected for review. Studies presented in other sources that did not undergo such a review process, like student thesis, conference contributions, or working papers, are not selected.

2.2 Data collection and analysis

The data in this article are the studies reviewed, collected by using the online database of Scopus©, since these studies usually comply with the criteria for independent review before publishing. The search is a multi-stage one, in which each stage represents one of the most important steps in traditional MSW management systems. The search terms as well as the stage-wise extraction of relevant articles are presented in the subsequent subsection.

2.2.1 Search terms inserted

Stage 1 – separate collection of PCPW: the backbone of traditional recycling is the separate collection of PCPW (Mwanza et al., 2018). The reason for that is that separate collection of PCPW produces secondary raw materials of higher quality than sorting of mixed MSW (Cimpan et al., 2015). To unveil selected literature on this topic, the search terms "waste" AND "recycling" AND "separate collection" were used.

Stage 2 – waste pre-treatment by material recovery facilities (MRF): when separate collection of PCPW cannot further increase the separate collection rate, or when the costs are too high (D'Onza et al., 2016; Feil et al., 2017; Janz et al., 2011), the extraction of recycling materials by MRFs is suggested as a viable option (Cimpan et al., 2016). Currently an even more important role than by these multi-stream MRFs is played by single- or dual-stream MRFs (Antonopoulos et al., 2021). The case studies published in the literature on these topics are aimed to be found by using the search terms "waste" AND "material recovery facility". The term "recycling" is not used since it is metaphorically included in "material recovery facility".

Stage 3 –MSW incineration bottom ash (IBA) treatment: unburnable waste fractions can also be recovered from MSW IBA. From the considered PCPW fractions, this counts for metals and, in theory, glass (Chimenos et al., 1999; Šyc et al., 2018; Šyc et al., 2020). To consider this option, the search terms "waste" AND "incineration" AND "recycling" AND "bottom ash" are introduced.

2.2.2 Reduction procedure and data analysis

First, all articles were analyzed based on their title, keywords, and abstract. Based on that articles that presented a case study outside of the EU or that focused on a waste type not considered, were eliminated. For the remaining articles, also the full text was analyzed and based on that, articles that did not match with the type of studies defined, were eliminated. The final list that remained was then used in this review and they are presented in this work.

3. RESULTS AND DISCUSSION

3.1 Separate collection

The first search gave in total 219 articles published between the years 2010 and 2021 that contained the search terms "waste" AND "recycling" AND "separate collection". After eliminating these with a geographic focus outside of the one defined, a total number of 163 articles remained. A further elimination of these studies that considered a waste material which was not in the scope of this article, for instance biowaste, waste electronic and electrical equipment, or waste textiles, the search yields 119 articles. From these, 15 articles remained for the in-depth content analysis, as the other articles did not contained case studies as desired by the authors. These 15 articles were analyzed in detail.

Five of the articles used the country or national level for their investigation, and eleven city or municipality level. The Netherlands was the country that contained with six the most case studies (Dijkgraaf & Gradus, 2020; Feil et al., 2017; Picuno et al., 2021; Seyring et al., 2016; Thoden van Velzen et al., 2019; Warrings & Fellner, 2019), followed by Italy with five (Bertanza et al., 2021; Del Cimmuto et al., 2014; Romano et al., 2019; Seyring et al., 2016; Warrings & Fellner, 2019), Germany with four (Feil et al., 2017; Picuno et al., 2021; Seyring et al., 2016; Warrings & Fellner, 2019), Austria (Picuno et al., 2021; Seyring et al., 2016; Warrings & Fellner, 2019), Finland (Dahlbo et al., 2018; Seyring et al., 2016; Warrings & Fellner, 2019), Sweden (Rousta et al., 2016; Seyring et al., 2016; Warrings & Fellner, 2019), Spain (Gallardo, Bovea, Colomer, et al., 2012; Gallardo, Bovea, Mendoza, et al., 2012; Seyring et al., 2016) and United Kingdom (Seyring et al., 2016; Wang et al., 2020; Warrings & Fellner, 2019) with three each; Belgium, Czech Republic, France, Portugal (Seyring et al., 2016; Warrings & Fellner, 2019) and Poland with two case studies (Picuno et al., 2021; Seyring et al., 2016). Switzerland (Haupt et al., 2018) as well as all other member states of the EU28 (Seyring et al., 2016) that have hitherto not been mentioned were covered in one case study. With respect to packaging waste materials, of the ones considered, plastics was contained in all but one case study that solely focused on aluminum. Eleven studies considered metals, seven paper & cardboard as well as glass, while beverage cartons were contained in one case study only. Eight case studies provide a comparison between different municipalities or cities, five a comparison of two years between which a historical development took place, and two case studies provided a cross-country comparison. One case study provided an experiment in terms of analyzing the status quo, carrying out an intervention and then analyzing the result of the experiment (Rousta et al., 2016). Table 1 gives an overview and description on the 16 studies analyzed.

Some of the studies analyzed selected determining factors for separate collection rates. One finding was that the higher the population density and population numbers are, the lower the separate collection rate is. Examples come from Germany and the Netherlands (Feil et al., 2017), Poland (Połomka et al., 2020), but also from analyzing data for the EU-28 (Seyring et al., 2016). In their report for the European Commission, Seyring et al. (2015) showed that the average separate collection rate of packaging materials and biowaste is, with few exceptions, much lower for the capitals of the EU-28 than for the regarding countries. This negative correlation between population density and separate collection is shown for the EU-28 in Figure 2. Contrary to that, Romano et al. (2019) found that in the case of Tuscany region in Italy, the separate collection rate increased with higher population density. For this reason, Romano et al. (2019) suggested to introduce actions that raise awareness and encourage pro-environmental behavior of inhabitants of low-density, rural areas. Since population density can hardly be influenced by waste management, only one of these studies that found a negative relation between separate collection rates and population density, namely Feil et al. (2017), suggested to overcome this gap by introducing post-sorting of recycling materials from mixed MSW by MRFs in areas with high population density and low separate collection rate. Figure 2 shows the separate collection for the EU-28 countries and their capitals, based on data from Seyring et al. (2015).

A more often investigated factor that determines the separate collection rate is the collection system, for instance if it involves a deposit-refund system (DRS) for single use beverage containers or not. All studies that investigate DRS agree that it leads to higher separate collection rates than systems without DRS (Dahlbo et al., 2018; Picuno et al., 2021; Seyring et al., 2016; Warrings & Fellner, 2019). However, the case of aluminum beverage cans shows that a sophisticated separate collection system (e.g. in Italy) or post-sorting of mixed MSW or IBA (e.g. in Netherlands, Belgium, France, Austria) reduces the gap to countries with DRS (Warrings & Fellner, 2019). Also, pay as you throw (PAYT) programs positively influenced the separate collection rate (Seyring et al., 2016; Thoden van Velzen et al., 2019).

Another option to increase separate collection is to reduce the distance for users to the separate collection container by shifting from drop-off to kerbside or door-to-door collection. This had positive effects in many cities and countries (Bertanza et al., 2021; Dahlbo et al., 2018; Dijkgraaf & Gradus, 2020; Gallardo, Bovea, Colomer, et al., 2012; Seyring et al., 2016; Wang et al., 2020). Particularly impressing are the increase in the separate collection rate in Brescia by a factor of 2-3 within three years (Bertanza et al., 2021) and Nottingham with a factor of 1.5 within one decade (Wang et al., 2020). With respect to the quality of the collected material, for instance in terms of extraneous material contents, Bertanza et al. (2021) pointed out that a decrease in the quality was expected. At the paper sorting facility that receives the separate collected materials from Brescia, however, no increase in the rejects was recorded. Information on whether there was an increase in rejects in the paper mill was not available to the authors. The result of Brescia is also displayed in Figure 4, based on data from Bertanza et al. (2021).

In addition, two studies specifically deal with the question of number of drop-off points and distance to households. Gallardo, Bovea, Mendoza, et al. (2012) found a clear negative correlation between distance to drop-off points and separate collection rate for a selection of Spanish municipalities with more than 50,000 inhabitants. Haupt et al. (2018) showed for the example of PET beverage bottle collection in Switzerland that the number of drop-off points increases much faster than the amount of PET beverage bottles collected and recycled. This result which indicates a ceiling for separate collection rates even when the collection system is expanded, is shown in Figure 5 based on data from Haupt et al. (2018).

Finally, some of the studies claim that comparison of different separate collections systems should consider the composition of the collected fractions. Gallardo, Bovea, Colomer, et al. (2012) concluded in their comparison of Spanish municipalities larger than 5,000 inhabitants that a door-to-door collection of commingled recycling materials (metals, paper and cardboard, plastics) is the most

TABLE 1: In-depth analyzed articles on separate collection of waste (n=15).

No.	Authors	Level	Description of area	Materials	Content of study	Comparison
Sc01	Picuno et al. (2021)	Country	AT, GER, NL	plastics	Comparison of collected, sorted, recycled plastics in AT, DE, NL	Cross-country
Sc02	Bertanza et al. (2021)	City	Brescia, IT	glass, metals, plastics, paper	Historical development of separate collection rates in Brescia over 30 years, particularly after changing from bring to kerbside collection	Historical development
Sc03	Dijkgraaf & Gradus (2020)	Country	NL	plastics	Comparison of plastic amounts for recycling from 99 municipalities in NL with different collection systems, including MRFs for mixed MSW	Cross-municipality
Sc04	Połomka et al. (2020)	Municpality	22 Municipalities in Marszów, PL	LWP (metals, plastics)	Comparison of the development of separate collected LWP (metals, plastics) in selected rural and urban municipalities in PL	Cross-municipality
Sc05	Wang et al. (2020)	City	Nottingham, UK	glass, metals, plastics, paper	Separate collection rates in Nottingham between 2006 and 2016, after changing from drop-off points to kerbside collection	Historical development
Sc06	Romano et al. (2019)	Municipality	Municipalities in Tuscany, IT	glass, metals, plastics, paper	Comparison of separate collection rates of municipalities with different organization form (public / private operator)	Cross-municipality
Sc07	Thoden van Velzen et al. (2019)	Municipality	Oosterhout & Waal-wijk Municipality, NL	beverage cartons, metals, plastics	Separate collection rates of beverage cartons, plastics, and metals from 21 households in two municipalities in NL; based on that, re-calculation of separate collection rate in 13 municipalities in NL	Cross-municipality
Sc08	Warrings & Fellner (2019)	Country	AT, BE, CZ, FR, GER, GR, IT, NL, PT, SE, UK	metal (aluminum)	Comparison of separate collection, post-sorting (from incineration bottom ash) and disposal rates of aluminum beverage cans in 11 European countries, including countries with deposit system	Cross-country
Sc09	Haupt et al. (2018)	Country	CH	plastics (PET)	Separately collected and recycled amounts of PET beverage bottles in CH over 25 years, considering extension of collection points (bring system) and PET bottle content in collected amounts	Historical development
Sc10	Dahlbo et al. (2018)	Country	FI	plastics	Separately collected and post-sorted plastics in FI before and after extension of separate collection and installing an MRF for mixed MSW	Historical development
Sc11	Feil et al. (2017)	Municipality	Northrhine-West-phalia, GER; NL	plastics	Separately collected plastics in municipalities of different population densities in GER and NL and estimation of potential of plastics from automatic sorting in a material recovery facility for mixed MSW	Cross-municipality comparison
Sc12	Seyring et al. (2016)	City	28 capital cities in Europe	glass, metals, plastics, paper	Comparison of separate collected glass, metals, paper, plastics in 28 EU-capitals, distinguishing between separate collection system	Cross-municipality
Sc13	Rousta et al. (2016)	Municipality	Borås Municipality, SWE	glass, metals, plastics, paper	Comparison of separate collected glass, metals, paper, plastics in a pilot area in Borås before an after interventions (information campaigns, decreasing distance to drop-off centers)	Experiment
Sc14	Gallardo et al. (2012)	Municipality	115 Municipalities, pop. >5,000, ES	glass, metals, plastics, paper	Comparison of the separate collection rate of glass, metals, paper, plastics of cities with different collection systems (kerbside, drop-off)	Cross-municipality
Sc15	Gallardo et al (2012)	Municipality	45 Municipalities, pop. >50,000, ES	glass, metals, plastics, paper	Comparison of the separate collection rate of glass, metals, paper, plastics of cities with varying distance to drop-off points	Cross-municipality

favorable option, not mentioning the potential losses during sorting in an MRF. These losses, however, can be substantial, depending not only on the MRF technology, but also the quality of the input material (Antonopoulos et al., 2021). Some authors argue that this quality of separately collected PCPW decreases also when extending the separate collection system. Haupt et al. (2018) conclude that the extension of the PET-beverage bottle collection system in Switzerland lead to a higher amount of extraneous materials disposed-off in the collection containers. A similar finding was made by Thoden van Velzen et al. (2019) who showed that higher amounts of recycling materials separately collected does not necessarily mean a better performance, since undesired fractions may increase

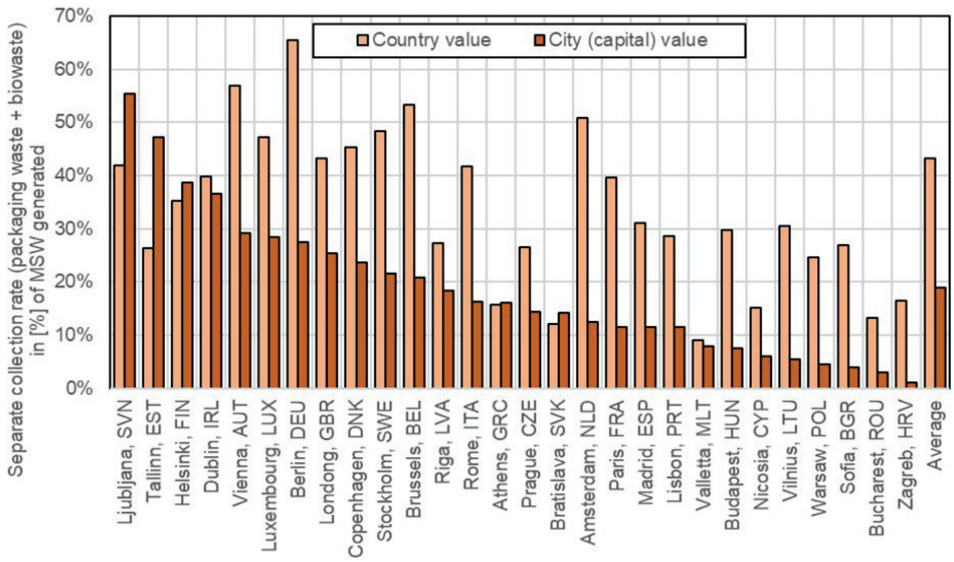

FIGURE 2: Separate collection rates of packaging waste and biowaste in the EU-28 countries and their capitals (cities). Data from Seyring et al. (2015).

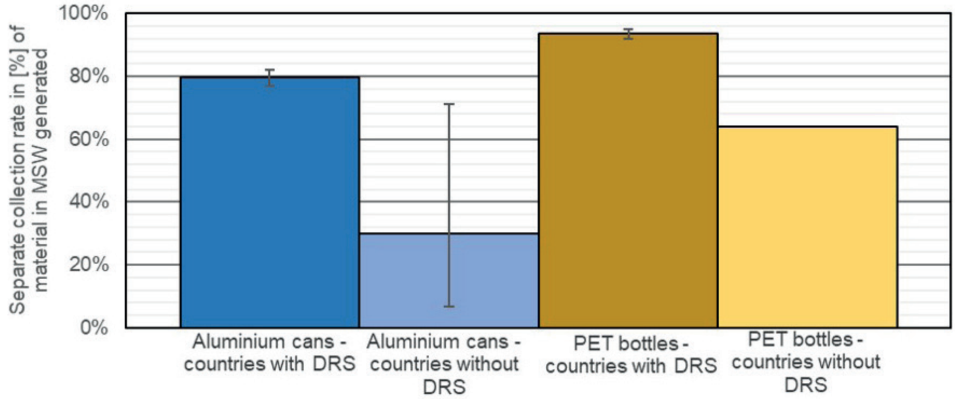

FIGURE 3: Separate collection rates of aluminium cans and PET bottles with and without deposit-refund system (DRS), based on Warrings and Fellner (2019) for aluminium and Picuno et al. (2021) for PET.

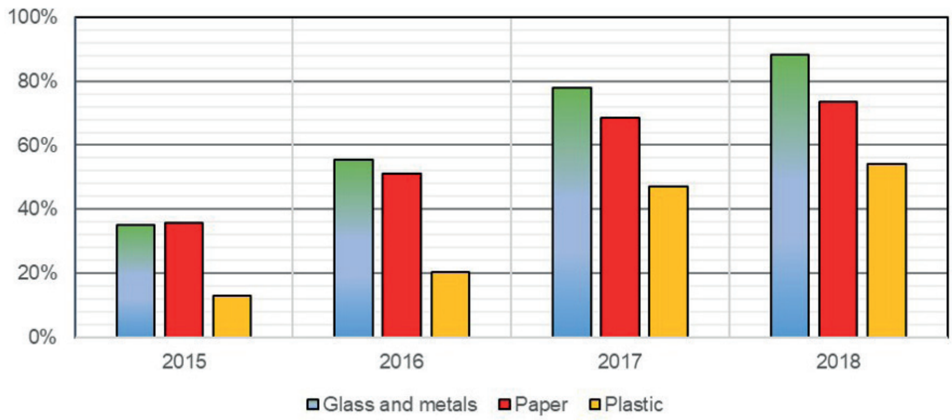

FIGURE 4: Development of separate collection rates of glass & metals, paper & cardboard, and plastics in Brescia after shifting from drop-off to door-to-door collection. Data from Bertanza et al. (2021).

over-proportionally. Also, Połomka et al. (2020) found for their Polish region under investigation that the share of undesired fractions in separately collected lightweight packaging waste increased with extension of the collection service and amounts, while the desired fractions decreased. Furthermore, Wang et al. (2020) showed that in Notting-

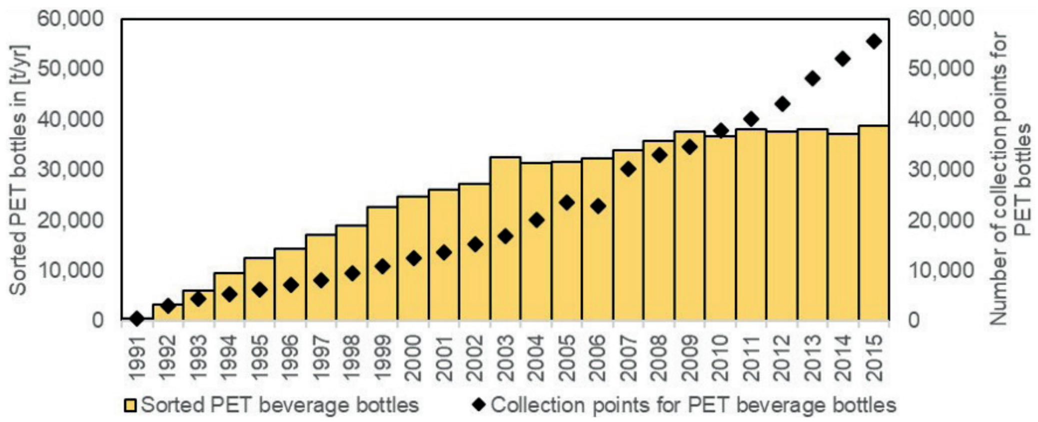

FIGURE 5: Development of the number of drop-off collection points and the separately collected and sorted amounts of PET beverage bottles in Switzerland. Data from Haupt et al., (2018).

ham, the extension of separate collection not only lead to higher amounts of PCPW collected, but also that the share of waste input that was sorted out for recycling declined, from 99.6% in the year 2006/2007 to 81.8% in the year 2016/2017. All of this indicates a decrease in the quality when separate collection systems were extended, and for this reason, data that solely rely on the separately collected amount of recycling material not considering its composition, can hardly serve to assess the efficiency of a separate collection system (Thoden van Velzen et al., 2019).

3.2 Material recovery facilities

The first search yielded 150 articles published in 2010-2021 containing the terms "waste" AND "material recovery facility". By eliminating 88 articles outside of the defined geographic area, 62 articles remained. Further eliminating these which are out of the scope by either a a focus on a different material than PCPW or from the method used (e.g. review) or data presented (e.g. solely secondary data already published in one of the other articles selected), solely 14 articles, shown in Table 2, remained for the in-depth content analysis.

Of the 14 articles selected, one was at country level, six investigated MRFs as part of the MSW management system of a city or region, six were on plant level only, and one was at the level of a paper factory that received waste paper from different MRFs. Four studies were related to the UK (Kirk & Mokaddam, 2021a, 2021b; Miranda et al., 2013; Wang et al., 2020) and Spain (Ip et al., 2018; Istrate, Galvez-Martos, et al., 2021; Istrate, Medina-Martos, et al., 2021; Miranda et al., 2013), and three to Italy (Ardolino et al., 2017; Gadaleta et al., 2020; Mastellone et al., 2017). One study each was on Portugal, Germany, Austria and without any reference (Cimpan et al., 2016; Dias et al., 2014; Griffiths et al., 2010; Warrings & Fellner, 2021). With respect to input flows, eight studies consider MRFs for mixed MSW, seven for lightweight packaging (LWP) of plastics, metals, and sometimes lightweight beverage cartons (LBCs), and two on commingled packaging waste (CPW) that also include paper or glass. Most studies analyzed MRFs that aim to recover metals (ten), plastics (nine), glass (seven), paper (four), and LBCs (two). However, the most detailed analysis was by most studies carried out on plastics.

Two important indicators for the sorting process in an MRF are the sorting efficiency per material defined as the amount of material in the output flow that is sent to recycling in mass percent of the amount of the material in the input flow, shown in Table 3, and the purity of the output flow defined as the content or grade of the desired material, shown in Figure 6.

The studies agree in the point that sorting efficiencies are generally higher for MRFs feed with commingled (COM) or lightweight packaging PCPW (LWP), as well as for metals, PET and HDPE (Cimpan et al., 2016; Gadaleta et al., 2020; Ip et al., 2018; Istrate, Medina-Martos, et al., 2021). For the latter materials, the values are in the range of other sources (Antonopoulos et al., 2021; Brouwer et al., 2019; Brouwer et al., 2018; Van Eygen et al., 2018). However, the data of Cimpan et al. (2016) suggests that there is a large difference between plants as designed and in their typical operation. This finding by Cimpan et al. (2016) is shown in Table 3.

With respect to the purity of the material for recycling produced, better values were achieved for PET and HDPE than for LDPE and PP. Values were also good for metals and lightweight beverage cartons (LBC), however, there were not many case studies for these materials (see Figure 6). This lack of data was also a problem that Warrings and Fellner (2021) faced in their study. Nevertheless, the input data, i.e. the sorting efficiency on which the authors calculate the increase of the sorted aluminum rates, was not that clearly expressed. Figure 6 shows the purity of the desired material fraction extracted from the MRFs, based on data from Mastellone et al. (2017), Gadaleta et al. (2020), and Ip et al. (2020).

For paper PCPWs, little data was available, particularly with respect to the quality of the output flows in terms of purity. However, the study of Miranda et al. (2013) shows that paper particularly from suppliers from the UK operating small MRFs for commingled PCPW was of low quality. Compared to paper and the other materials, there was even less data on the sorting and the quality of glass, suggesting that it is neither collected with other PCPW, nor sorted from mixed MSW that often, even though there is a large

TABLE 2: In-depth analyzed articles on material recovery facilities (n=14).

No.	Authors	Level	Area	Input	PCPW	Content of study
MRF01	Istrate et al. (2021)	Plant, City	Madrid, ESP	MSW COM	glass metals plastics paper	Material flow analysis (MFA) of MRFs for mixed MSW and commingled PCPW. Both sorted 1% / 23% glass, 44% / 66% iron, 12% / 33% aluminum, 7% / 20% paper, 13% / 20% cardboard, 3% / 43% carton, 5% / 82% PET, 5% / 82% HDPE, 5% / 76% LDPE related to the input (MSW MRF / commingled).
MRF02	Istrate et al. (2021)					
MRF03	Warrings & Fellner (2021)	Country	AUT	MSW LWP	aluminum	Scenarios to increase aluminum recycling concluding that mixed MSW MRFs and improved MSW IBA treatment were the best measures.
MRF04	Kirk & Mokaddam (2021a, b)	Plant, City	London, GBR	MSW	glass metals plastics paper	Sampling input material or mixed MSW MRFs, showing high variation depending on the company delivering the MSW, but also the impact on operation costs and revenues of the MRFs.
MRF05						
MRF06	Gadaleta et al. (2020)	Plant, Region	Bari, ITA	LWP		Determining the sorting efficiency (recovery index RI) and quality (purity index PI) of plastic PCPW from an MRF called ASM. Both indicators were high for PET, HDPE, PP and LDPE >A3, but low for LDPE<A3. All indices were higher than in the MRF Bedonia analyzed earlier by the authors.
MRF07	Wang et al. (2020)	Plant	Notting-ham, GBR	MSW LWP	glass metals plastics paper	MRFs for commingled PCPW and mixed MSW + other waste extracted 82% and 9% of PCPW for recycling, respectively. The commingled MRF contributed to 93% of glass, 95% of paper, 23% of metal and 78% of plastic, and the mixed MSW MRF to 26% of metals and 4% of plastics sorted for recycling. Separate collection contributed 7% glass, 5% paper, 17% metals, and 18% plastics, and MSW IBA treatment 34% metals for recycling.
MRF08	Ip et al. (2018)	Plant	Toledo, ESP	LWP	glass metals plastics	MFA of a commingled PCPW MRF. Grades (purity) of outputs and sorting efficiencies were high for iron, aluminum, PET, HDPE, LBC.
MRF09	Ardolino et al. (2017)	Plant, City	ITA	MSW	metals plastics	An LCA was performed for an mixed MSW MRF, focusing on biowastes. The mixed MSW had 26% plastics and 4% iron, of which 0.9% of plastics and 0.01% of iron were sorted by MRF.
MRF10	Mastellone et al. (2017)	Plant	ITA	LWP	metals plastics	MFA of an MRF for LWP (plastic, ferrous metals, aluminum) was performed by collecting samples of the input and output materials. Based on that, different indicators were calculated representing the efficiency of sorting and the purity of the output material. These were calculated as a time series to show temporal variations.
MRF11	Cimpan et al. (2016)	Plant	DE	LWP	metals plastics LBC	A study that shows great discrepancies between sorting efficiency as designed and in real operation, including an economic analysis.
MRF12	Dias et al. (2014)	Plant	POR	MSW	glass	Amount and content of glass in the heavy output of five mechanical-biological treatment plants in Portugal. Glass dry matter contents 33-83%.
MRF13	Miranda et al. (2013)	Factory	ESP, GBR	Single Com-in-gled	paper	Comparison of the quality of waste paper from old and modern commingled PCPW MRFs, showing better values of the latter. Qualities are worse than in separate collection.
MRF14	Griffiths et al. (2010)	Plant	None	LWP MSW		Engineering principles of MRFs applied to separately collected PCPW (clean MRF) and to mixed MSW (dirty MRF).

potential as well as content in certain output fractions of mixed MSW MRFs (Dias et al., 2014).

3.3 Recovery of recycling materials from waste incineration bottom ash

The first search yielded 275 articles published in 2010-2021 containing the search terms. Then, 121 articles not in the defined geographic area were eliminated and 154 articles remained. A large number of 138 article were eliminated as they focused on other materials or topics. The most important topic therein was the recycling of the mineral fraction of MSW IBA in construction materials like cement, concrete, or as base-layer in road constructions. At the end, 16 articles remained for the in-depth analysis, as shown in Table 4.

All articles dealt with IBA from grate incinerators, but none on fluidized bed incinerators or rotary kilns. Two of the articles investigated recycling of PCPW from MSW IBA at European level (Abis et al., 2020; Bruno et al., 2021), another two at national level (Grosso et al., 2011; Warrings & Fellner, 2019). All other articles investigated the topic at the level of MSWI or MSW IBA treatment plants. Therein, Italy is represented in six articles (Abis et al., 2020; Biganzoli & Grosso, 2013; Biganzoli et al., 2014; Biganzoli et al., 2013; Bruno et al., 2021; Warrings & Fellner, 2019), Germany and Denmark in five (Abis et al., 2020; Allegrini et al., 2014; Bruno et al., 2021; Gökelma et al., 2021; Holm & Simon, 2017; Huber, 2020; Warrings & Fellner, 2019), Austria, the Netherlands and Belgium in four (Abis et al., 2020; Bruno et al., 2021; Hu et al., 2011; Huber, 2020; Van Caneghem et al., 2019; Warrings & Fellner, 2019), Spain and the UK in three (Abis et al., 2020; Bruno et al., 2021; del Valle-Zermeño et al., 2017; Gökelma et al., 2021), Switzerland and all other member countries of the EU in two (Abis et al., 2020; Bruno

TABLE 3: Sorting efficiencies of MRFs (Cimpan et al., 2016; Gadaleta et al., 2020; Ip et al., 2018; Istrate et al., 2021). For PET, the non-weighted mean of different fractions was used.

		Istrate et al. (2021b)	Istrate et al. (2021a)	Cimpan et al. (2016)	Cimpan et al. (2016)	Cimpan et al. (2016)	Cimpan et al. (2016)	Cimpan et al. (2016)	Cimpan et al. (2016)	Gadaleta et al. (2020)	Gadaleta et al. (2020)	Ip et al. (2020)
	Input material	MSW	COM	COM	COM	COM	COM	COM	COM	LWP	LWP	LWP
	Operation	typical	typical	typical	typical	typical	designed	designed	designed	typical	typical	modelled
	Plant type / name	Madrid	Madrid	basic	medium	advanced	basic	medium	advanced	ASM SSC	Bedonia SSC	Toledo
Glass	Glass	1%	23%									
	Aluminium	12%	33%	30%	35%	40%	60%	70%	80%			82%
	Ferrous	44%	66%	40%	44%	48%	80%	88%	95%			92%
Paper	Paper	7%	20%	25%	30%	35%	50%	60%	70%			
	Cardboard	13%	20%	25%	30%	35%	50%	60%	70%			
	Carton	3%	43%	25%	30%	35%	50%	60%	70%			
	LBC			35%	39%	43%	70%	78%	85%			90%
Plastics	PET	5%	82%			40%			80%	94%	57%	98%
	HDPE	5%	82%		27%	27%		54%	54%	97%	97%	88%
	LDPE small	5%	76%	25%	30%	35%	50%	60%	70%	30%	21%	
	LDPE large									85%		
	PP				27%	27%		54%	54%	97%	78%	

et al., 2021) and all other EFTA and EU candidates hitherto not mentioned in one (Bruno et al., 2021). All articles focused on metals, except two that at least mentioned another glass too (Bruno et al., 2021; del Valle-Zermeño et al., 2017).

The guiding research question in almost all articles deals with the recovery of metals from MSW IBA. For packaging metals, which are either iron or aluminum based, the latter received much more attention than the prior. This may have to do with the already high efficiency and thus little losses of iron through magnetic separation, if compared to aluminum (Allegrini et al., 2014; Huber, 2020; Mehr et al., 2021). The latter is basically lost in two ways of oxidation (thermal oxidation during incineration, or oxidation by wheathering in the IBA treatment), or due to low sorting efficiencies. Thermal oxidation takes place twice, during the incineration process itself, but also in the metal smelting. That these losses can be substantial is shown by a number of studies. Biganzoli et al. (2014) estimated a loss of 54-63% during incineration. Allegrini et al. (2014) stated personally communicated losses in metal smelting of between 19-34%, depending on the grain size, while Mehr et al. (2021) reported 14%. Iron is mainly lost in the treatment only, and oxidation seems to be in a lower range (Mehr et al., 2021). The reason for that is that aluminum packaging occurs much more as flexible packaging with thin layers, making it more exposed to temperature (Biganzoli & Grosso, 2013; Biganzoli et al., 2014; Biganzoli et al., 2013; Gökelma et al., 2021). With respect to the sorting efficiency of IBA treatment plant, Figure 7 shows some values from literature. These do not contain oxidation losses during smelting.

The results shown in Figure 7 suggest that there are huge differences in the recovery of PCPW metals from MSW IBA. This also means that there is still a large potential available, not only from packaging, but also from non-packaging metals which are sometimes in practice incorrectly assigned to packaging (Van Caneghem et al., 2019). An even larger potential than for metals is present for glass PCPW, which is only mentioned by two of the reviewed studies (Bruno et al., 2021; del Valle-Zermeño et al., 2017). Neither of these two studies, however, actual-

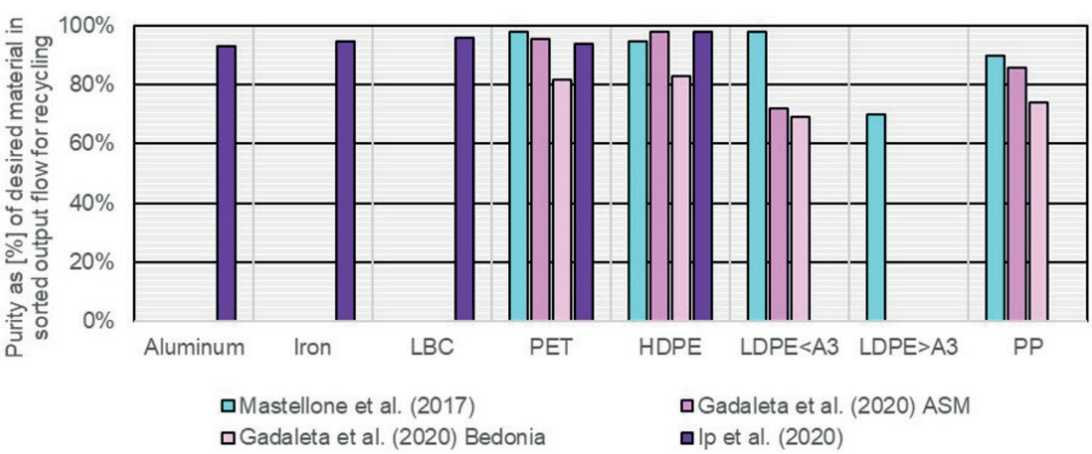

FIGURE 6: Purity of in MRF produced material flow for recycling, defined as the content or grade of the desired material, based on literature (Mastellone et al., 2017; Gadaleta et al., 2020; Ip et al., 2020). ASM and Bedonia stands for two different MRFs.

TABLE 4: In-depth analyzed articles on separate collection of waste (n=16).

No.	Authors	Level	Area	Input	PCPW	Content of study
IBA01	Bruno et al., (2021)	Continent	Europe (EU27, EU candidate countries, EFTA, UK)	MSW	metals	Scenario on the effects of reducing the amount of untreated IBA to 0 and implementing state-of-the-art IBA treatment in 38 European countries on recycling of minerals and metals. Metal reco-very could increase from 1.83 Mt/a to 3.77 Mt/a.
IBA02	Gökelma et al., (2021)	Plant	8 samples from USA, UK, DK	MSW	aluminum	Recyclability of aluminum in IBA, determined by oxide layer thickness, metal yield and coagulation efficiency after re-melting. 76-93% of aluminum was recovered as metal.
IBA03	Mehr et al., (2021)	Plant	CH	MSW	metals	LCA based on updated data of the dry IBA treatment fa-cility in Hinwill. Data for metals including iron, stainless steel, and aluminum >0.3 mm grain size. Expressed in extraction efficiency and recycling efficiency (i.e. substi-tution of primary raw material).
IBA04	Abis et al., (2020)	Continent	EU 28	MSW	metals	Analysis of the nexus between recycling and MSWI, highlighting the relevance to improve recovery of metals and minerals from IBA.
IBA05	Huber, (2020)	Plant	GER, AT, IT	MSW	metals	Modelling the material flows of metals and minerals based on data from 5 IBA treatment plants from AUT, DEU and ITA. Recovery rates and transfer coefficients are provided.
IBA06	Van Cane-ghem et al., (2019)	Region	Flanders, BEL	MSW	metals	MFA of packaging and non-packaging metals in Flan-ders, based on IBA sampling. Validation of recycling rates of metal packaging.
IBA07	Warrings and Fellner, (2019)	Country	AT, BE, CZ FR, GER, GR, IT, NL PT, SE, UK	MSW	aluminum	Comparing sorting rates for aluminum packaging waste in 11 EU countries, considering separate collected and IBA recovered metals. The latter are assumed to be higher than reported.
IBA08	Haupt et al., (2017)	Plant	CH	MSW	iron	Investigating the quality of steel scrap from IBA and other scrap, showing lower quality and higher recycling energy demand of IBA scrap.
IBA09	del Valle-Zer-meño et al., (2017)	Plant	ES	MSW	glass	Determining the impact of separate collection of glass on the glass content in IBA by sampling inputs and outputs of an IBA treatment plant
IBA10	Holm and Simon, (2017)	Plant	DE	MSW	metals	Comparing two dry and one wet treatment plant for IBA with a focus on use of the mineral fraction for construction, thereby establishing data on recovery rates for metals.
IBA11	Biganzoli et al., (2014)	Plant	IT	MSW	aluminum	MFA of aluminum in two italian MSWI plants with attached IBA treatment. 21-23% of aluminum was recov-ered. This can be increased by improvement to 28-38%. 54-63% are oxidized in MSWI and thus lost to recovery, but contributing to 1% of energy production.
IBA12	Allegrini et al., (2014)	Plant	DK	MSW	metals	MSW IBA treatment in a plant in DK was analyzed. 85% of iron and 62% of aluminum in IBA was found to be recovered
IBA13	Biganzoli and Grosso, (2013)	Plant	IT	MSW	aluminum	MFA of aluminum in two MSWI plants with attached IBA treatment in Italy, focusing on packaging aluminum. 81% of cans can be recovered, but only 51% of tray, 27% of mix and 47% of paper laminated foils.
IBA14	Biganzoli et al., (2013)	Plant	IT	MSW	aluminum	Testing of improved aluminum recovery from IBA fine fraction <4mm by H2 production. 15% was metallic and of this, 21% were recovered.
IBA15	Hu et al., (2011)	Plant	NL	MSW	aluminum	Comparing the recovery rates of different aluminum packaging. These were around 90-95% for cans, 85% for foil containers and 77% for thin foils, expressed in metallic aluminum.
IBA16	Grosso et al., (2011)	Country	IT	MSW	aluminum	Modelling aluminum recovery for scenarios in Italy for the MSW management system, including IBA treatment.

ly focused on recovering glass from IBA, even though del Valle-Zermeño et al. (2017) showed that there are large amounts of glass in IBA. One reason for that might be that past experiences with glass recovery from MSW IBA produced in a grate incinerator with wet discharge were not that successful (Šyc et al., 2020).

3.4 Towards a recycling-based circular economy from a systems perspective

The studies reviewed showed not only that there is already a lot of literature and thus knowledge available to increase the amount of PCPW available for recycling. They also suggest some important future directions of research.

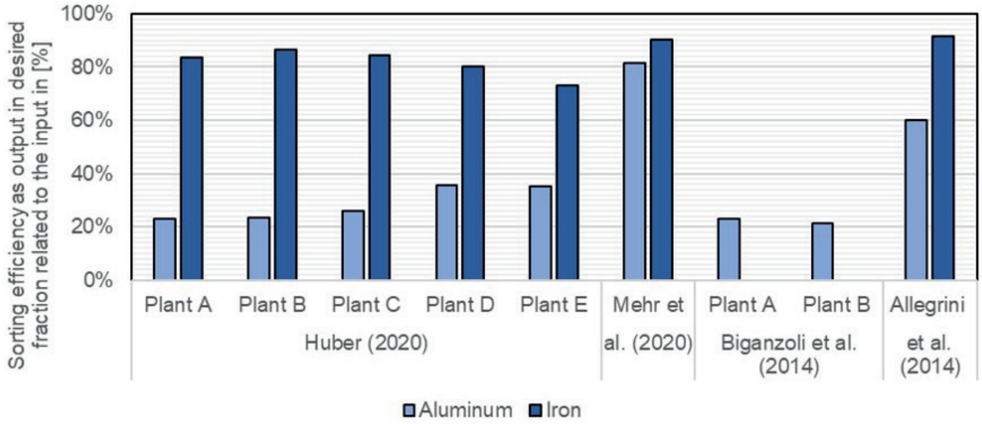

FIGURE 7: Sorting efficiency of IBA treatment plants. In Huber (2020), Plant C refers to the plant described by Allegrini et al. (2014) and Plants D and E to the plants in Holm and Simon (2017), all adapted to the Austrian situation.

For separate collection, which will very likely be the backbone of any recycling system also in future, a number of different systems are available, from single stream over combined light weight packaging (lightweight beverage cartons, metals, plastics) or mixed packaging (glass, metals) to commingled systems collecting all PCPW in one container (Cimpan et al., 2015). Yet, the question is which system fits best under certain circumstances and contexts. Such a context can be the population density or the size of a municipality. For instance, in dense populated areas where average housing area per capita is too scarce to reserve a lot of space for a large number of different bins for recyclable PCPW, commingled systems might be one alternative which should be tested before implementation, for instance in large-scale experiments that also involve the most important stakeholders of separate PCPW collection (Lederer et al., 2015; Pedersen & Manhice, 2020). This would mean that the decision on PCPW separation is shifted from the consumer to central sorting facilities, i.e. MRFs (Gundupalli et al., 2017). The recent progress in automatic sorting allows this option, even up to a level that mixed MSW can undergo such a recovery process (Feil et al., 2017). Event though the quality of the recycling material produced in these mixed MSW MRFs is expected to be lower for materials like paper and plastics, they provide a cost-effective measure to recover materials that otherwise would be incinerated (Janz et al., 2011). Nevertheless, additional post-treatment steps might be required to improve the quality of PCPW sorted from mixed MSW, but considering the size of the problem current societies are facing, it can be expected that the technological progress to deal with these quality issues will be available to recycling systems in the near future. For unburnable materials, MSW IBA treatment is the last option before the material is lost for recycling, and a lot of progress for recovering metals from IBA was made in the last years, as the increase of recovery rates in Figure 7 as well as literature shows (Šyc et al., 2020). Nevertheless, it is hardly imaginable that even the most sophisticated IBA treatment can replace separate collection or MRFs to recover PCPW metals (Warrings & Fellner, 2019). This counts particularly for aluminum and its high oxidation rates (Biganzoli et al., 2014). Moreover, some unburnable materials, and therein particularly glass, is often not even attempted to be recovered. This is interesting since glass makes a considerable portion of IBA (Chimenos et al., 1999; del Valle-Zermeño et al., 2017; Huber et al., 2020). However, probably, it is solely the firing technology that avoids the recovery of glass even from IBA. Considering the high contents of glass in the heavy fraction from mechanical biological treatment plants in Portugal (Dias et al., 2014) and that in some countries like Austria, this fraction is often burned in fluidized bed incinerators (Bösenhofer et al., 2015; Purgar et al., 2016), it is likely that IBA from fluidized bed incineration is more suitable for glass extraction and recycling. Beside the glass-rich feedstock, these incinerators usually burn with lower temperatures (Lecker & Lind, 2020) and furthermore, they discharge the IBA in dry form. Dry discharge also means that valuable materials like glass or metals are not present in clusters by sticky, wet fine particles of ash, making it easier to extract these materials by automatic sorting processes (Šyc et al., 2020). This makes it inevitable to investigate and consider not only the recovery potential of glass, but also of easily oxidizing metals (aluminum) and even the mineral fraction of different incineration technologies. The example of fluidized bed incinerators also shows that circular-based MSW management system need to be designed from a systems perspective. One argument against fluidized bed incineration were the high costs associated to the provision a mixed MSW splitting plant upstream of MSWI plant (Leckner & Lind, 2020). If an MRF for mixed MSW is needed anyway in order to comply with the recycling targets for plastics or paper, as presented and discussed in Section 3.2, however, it can also serve as such a splitting plant. Thus, from a systems perspective, solutions which were unfeasible in the past, might be beneficial in future, in order to establish a recycling-based circular economy, as shown in Figure 8.

4. CONCLUSIONS

The negative consequences of the life style of modern societies requires a mammoth project in order to provide a sustainable development for present and future generations, and the circular economy package of the EU is exactly the kind of project that might bring societies a step further into the right direction. Waste management can contribute a lot to achieve the objective of the circular economy package, but only if all potentials at each step of a waste management system are fully exploited. Both, societal and technological innovations to do so are available. These must be tested in different situations in order to implement them successfully, for the achievement of circular and sustainable society.

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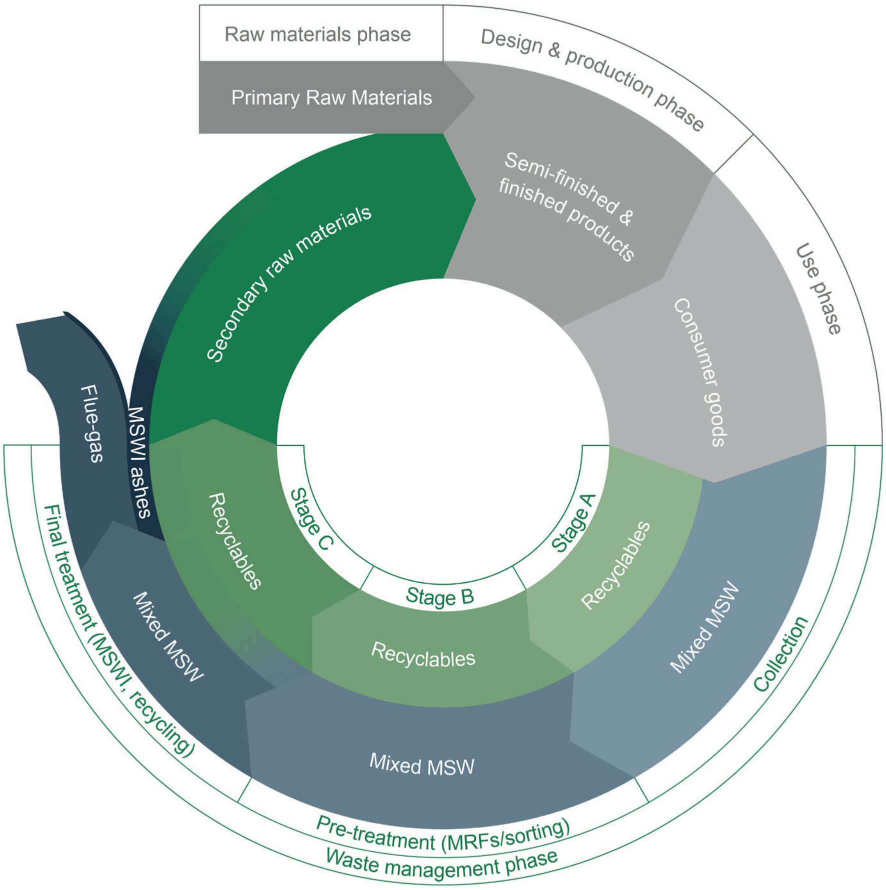

FIGURE 8: The life-cycle of a product in a more circular economy, based on enhanced municipal solid waste recycling. Design by solo-ohne© (https://solo-ohne.com/).

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