Resources, Conservation & Recycling 203 (2024) 107461

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

journal homepage: www.elsevier.com/locate/resconrec

# The contribution of waste and bottom ash treatment to the circular economy of metal packaging: A case study from Austria

Jakob Lederer [a,1,*], Dieter Schuch [b,c,1]

[a] Christian Doppler Laboratory for a recycling-based Circular Economy, Institute of Chemical, Environmental and Bioscience Engineering, TU Wien, Getreidemarkt 9, 1060, Wien, Austria
[b] Institute for Water Quality and Resource Management, TU Wien, Karlsplatz 13/226, 1040, Vienna, Austria
[c] Altstoff Recycling Austria AG, Mariahilfer-Straße 123, 1060, Vienna, Austria

ARTICLE INFO

Both authors equally contributed to this article. The order of authors is alphabetically.

*Keywords:*
Recycling rate
Metal packaging
Aluminum
Ferrous metals
Material flow analysis
Reverse modelling

ABSTRACT

The EU defined recycling rate targets for aluminum and ferrous metal packaging wastes. According to the European Commission, aluminum and ferrous metal packaging recovered through waste sorting and bottom ash treatment can also be considered for target achievement, in addition to separate collection. Lack of data causes that many countries don't report these recovered amounts. This study determined the aluminum and ferrous metal packaging recovered by waste sorting and bottom ash treatment and the contribution of these amounts to the recycling rate in Austria by material flow analysis. Results show that waste sorting and incineration bottom ash treatment increases the recycling rate of aluminum packaging from 33 % to 58 % and of ferrous metal packaging from 65 % to 93 %. The recycling rate for the sum of metal packaging thus increases from 53 % to 80 %. Data-improvement is required for future calculation of recycling rates of metal packaging.

*List of abbreviations*

| | |
|---|---|
| Al | Aluminum |
| APC | Air Pollution Control |
| AUT | Austria |
| BMK | Bundesministerium für Klimaschutz (Federal Ministry for Climate Protection) |
| CE | Circular Economy |
| EC | European Commission |
| EEA | European Environment Agency |
| Eq | Equation |
| EU | European Union |
| F | Flow |
| FBC | Fluidized Bed Combustor |
| Fe | Ferrous metal |
| FLA | Flanders |
| GI | Grate Incinerator |
| IBA | Incineration Bottom Ash |
| MA | Magistratsabteilung (Municipal Department) |
| MFA | Material Flow Analysis |
| MSW | Municipal Solid Waste |
| MSWI | Municipal Solid Waste Incineration |
| P | Process |
| PW | Packaging Waste |
| RDF | Refuse Derived Fuel |
| RR | Recycling Rate |
| TC | Transfer Coefficient |
| VIE | Vienna |
| wt | weight |

## 1. Introduction

In a world of multiple crisis, strategies that enhance the resilience and sustainable development of societies gain increasing relevance. One of these strategies is the circular economy (CE) that aims to reduce resource consumption, provide secondary raw materials by waste recycling, encourage the economy and protect the environment (Gaustad et al., 2018; Gavrilescu et al., 2023). In the EU, an important measure within the CE are recycling rate (RR) targets for municipal solid waste (MSW) streams like metal packaging waste (PW), were in the years 2025 and 2030, 50 % and 60 % of Al, and 70 % and 80 % of Fe, should be recycled (EU, 2018). Although emission factors for primary

* Corresponding author.
*E-mail address:* jakob.lederer@tuwien.ac.at (J. Lederer).
[1] Both authors equally contributed to this article.

https://doi.org/10.1016/j.resconrec.2024.107461
Received 21 October 2023; Received in revised form 6 January 2024; Accepted 29 January 2024
Available online 7 February 2024

production of aluminum has been reduced in the last years (European Aluminium, 2018), benefits of greenhouse gas by using recycled metals as secondary resources will continue (Turner et al., 2015). According to the European Environment Agency EEA (2022), nine EU member countries likely achieve the 2025 RR targets for MSW and PW. Among these countries is also Austria, which in 2019 failed to achieve the targets only for plastic PW (EEA, 2022). However, in 2020 the RRs of metal PW in Austria dropped to 66.1 wt-% from 84.7 wt-% in 2019 (BMK, 2023). A deeper look into Eurostat statistics indicate that the partial non-reporting of metal packaging recovered from incineration bottom ash (IBA) of MSW incineration (MSWI) is the main reason for this drop. While amounts recovered from IBA were reported for Fe, they were not displayed for Al packaging (Eurostat, 2023), even though Al is recovered from IBA in Austria (Warrings and Fellner, 2019). A probable reason for this is that the data required to calculate these amounts is not completely available. The rules to calculate recovered Al and Fe packaging from IBA foresee that the calculation distinguishes between Al and Fe packaging (Van Caneghem et al., 2019). To apply the calculation rules, first, the amounts of metals recovered from IBA, second, the concentration of Fe and Al in these metals, and third, the shares of packaging and non-packaging metal for Fe and Al, taken from waste analysis of the input into MSWI, are required. In Austria, the first variable can be taken from the federal waste management plan (BMK, 2023). The second variable, however, is not available at all. The third variable is only available for mixed MSW in Vienna, but not for the other provinces where Al and Fe PW was summarized as metal PW (Beigl, 2020; BMK, 2023; Gritsch and Lederer, 2023). Furthermore, waste sorting analysis for mixed commercial MSW, bulky MSW, and construction waste, which are the other main sources of metal PW, distinguished between Al and Fe packaging metals, but not considered Al and Fe non-packaging metals. As a consequence, a direct determination of Fe and Al PW recovered from IBA according to the calculation rules of the EC (2019) was not possible in Austria. Besides, since decades, substantial amounts of Al and Fe PW are recovered in Austria from mixed MSW sorting plants (Neubauer and Öhlinger, 2007; Warrings and Fellner, 2018). These amounts were also not reported to Eurostat, probably because this recovery route is not explicitly mentioned in the RR calculation rules of the EC (2019). However, they are recycled and their contribution to the RRs can be calculated in analogy to these amounts recovered from IBA. This is important also for other countries in the EU that apply mixed MSW sorting like Cyprus, Greece, the Netherlands, or Spain (Bourtsalas and Themelis, 2022; Cimpan et al., 2015; Lederer et al., 2022; Thoden van Velzen et al., 2020). To overcome these data gaps for Austria and providing an example also for other countries, this article asks the following research questions:

1) Which amounts of Al and Fe PW are present in Austrian MSW streams?
2) Which amounts of Al and Fe PWs are recovered by mixed and bulky MSW sorting and IBA treatment in Austria?
3) How do these amounts contribute to the RRs of Al and Fe PWs in Austria in the sense of the calculation rules of the EC (2019)?

To answer such questions, many studies used material flow analysis (MFA). Most of the recent MFA studies on PW were on plastic (eg. Amadei et al., 2023; Schuch et al., 2023). Of the few studies on metal PW, Warrings and Fellner (2018) modelled Al in Austria in 2013, but not Fe. Van Caneghem et al. (2019) and Tallentire and Steubing (2020) modelled metal packaging flows in Flanders (FLA) and the EU, not explicitly distinguishing between Al and Fe. The latter was carried out by Thoden van Velzen et al. (2020) for the Netherlands. As a consequence, there are only few studies that modelled the material flows of Al and Fe PW to calculate the RRs of these materials. Furthermore, in the past, MFA was also used to model the unknown quantities and concentrations of materials in input waste streams of waste treatment plants by measuring or collecting data on the output flows from these plants (Morf et al., 2000; Riber and Christensen, 2007). The present study will apply this indirect determination of material concentrations by reverse MFA modelling at country level.

## 2. Method and materials

### 2.1. Material flow analysis (MFA)

#### 2.1.1. System description and general information

MFA and the software STAN 2.6 were used for the calculation, distinguishing between goods like mixed MSW and substances like Al and Fe (Brunner and Rechberger, 2016; Cencic and Rechberger, 2008). Minor alloy elements where not considered, even though they may affect the recycling quality and potential applications (Cooper et al., 2020; Dworak et al., 2022; Løvik and Müller, 2014). Al and Fe as part of multi-material packaging like beverage cartons was also not considered, due to their negligible amounts and thus non-inclusion in metal packaging waste statistics (Warrings and Fellner, 2018). The system under investigation considers thus all relevant Al and Fe PW containing flows generated in Austria in the year 2020 according to BMK (2023), i.e. mixed household MSW, mixed commercial MSW, the combination of both (mixed MSW), bulky MSW, construction waste, and separately collected PW. Other sources of metal packaging of not specified origin are modelled as an additional MSW flow (other MSW, e.g. paint cans). The MFA model is shown in Fig. 1, a list of all abbreviations of the material flows is given in table S1 (supplementary file).

In the MFA model, processes are abbreviated as *P* and flows as *F*. Like in Lederer et al. (2020), processes are numbered consecutively starting with P0. The numbering of flows indicates the processes of origin and destination. If more than one flows have the same process of origin and destination, they are numbered (e.g. F_1-3.2 is the second flow from process P1 to process P3).

The colors of material flows in Fig. 1 indicate these for which: the mass of goods and the concentration of substances was given (green); the mass of goods was known (red); the concentration or the mass of substances was available (blue); or no information was found (black). For processes in grey color, input or output substance flows for Al and Fe packaging were calculated by mass balancing (Eq. (1)). Processes in yellow indicate these for which transfer coefficients (TCs) (Eq. (2)) were used in addition to mass balances for the calculation. Processes in white color are outside the system boundary and not considered for calculation.

Since the data availability is higher for the province of Vienna than for the eight other provinces, Vienna is calculated separately. Flows and processes with a suffix VIE denominate these in Vienna. AUT indicates all nine provinces. AUT–VIE indicate all flows and processes in provinces outside of Vienna.

#### 2.1.2. Material flows of Al and Fe PW containing goods

First, the material flows of goods were calculated, mainly by mass balance as in Eq. (1)

$$\sum_{k,in=1}^{k,in=n,in} \dot{m}_{k,in} = \sum_{l,out=1}^{l,out=m,out} \dot{m}_{l,out} \tag{1}$$

where $\sum_{k,in=1}^{k,in=n,in} \dot{m}_{k,in}$ is the sum of input-material flows and $\sum_{l,out=1}^{l,out=m,out} \dot{m}_{l,out}$ is the sum of output-material flows. The data on the Al and Fe packaging containing wastes generated and treated in MSW sorting plants, as well as Al and Fe packaging containing outputs from these plants come from the Austrian federal waste management plan (BMK, 2023). For Vienna, more detailed data on these flows from the municipal waste management company MA 48 as cited in Gritsch and Lederer (2023) were used. Amounts of wastes treated by MSWI and the amounts of air pollution control (APC) residues and IBA generated from MSWI came from Austria's Environment Agency (Kellner et al., 2022).

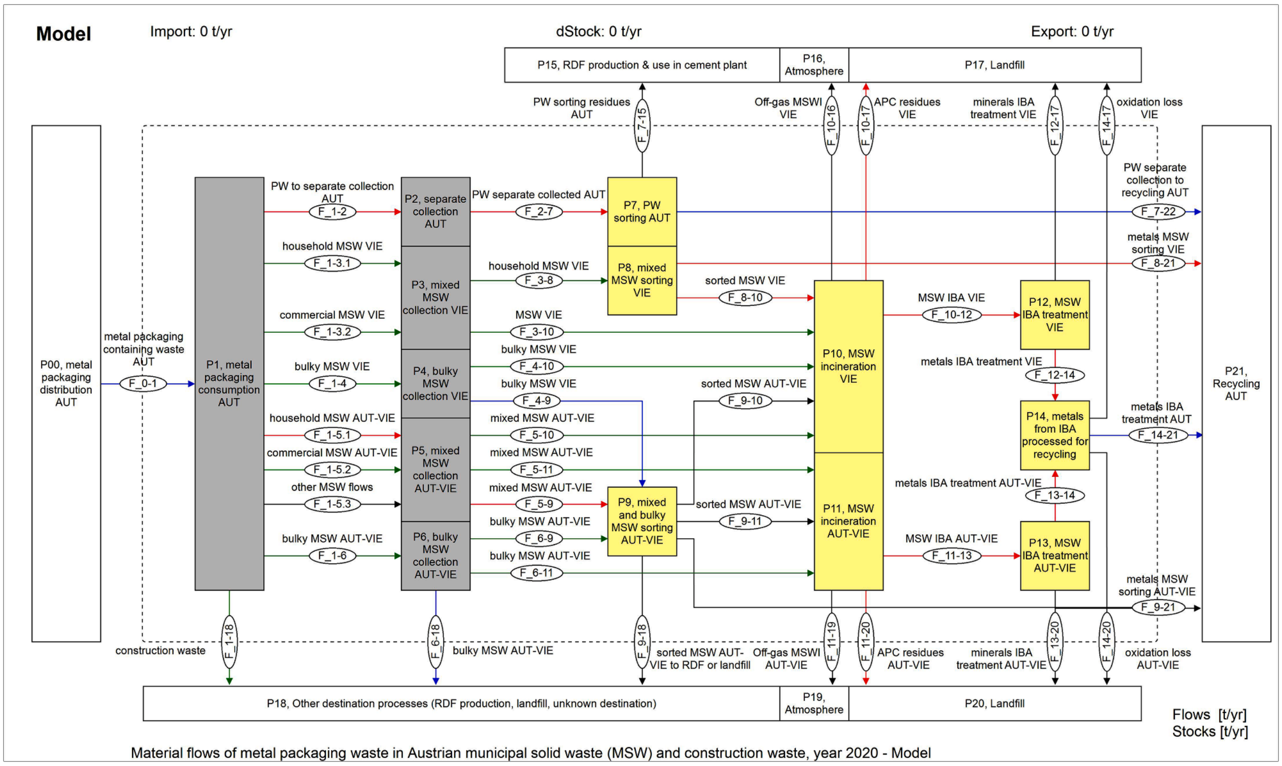


**Fig. 1.** Material flow analysis model for metal PW of Al and Fe in Austria for the year 2020.

Details on the data sources of the material flows of goods are shown in the supplementary file (Table S1).

Process P7, which is the sorting of PW contains metal packaging, was the only process for which material flows were calculated by the transfer function as in Eq. (2)

$$TC_{l,out} = \frac{\dot{m}_{l,out}}{\sum_{k,in=1}^{k,in=n,in} \dot{m}_{k,in}} \quad (2)$$

where $TC_{l,out}$ is the TC for an output material flow $l, out$, while $\dot{m}_{l,out,j}$ is the mass of the output-substance flow $l, out$. $\sum_{k,in=1}^{k,in=n,in} \dot{m}_{k,in}$ is the mass sum of input-material flows. While the TC in Eq. (2) shows the relation between one output flow and the sum of all input flows, TCs can also be related to one single input flow. This is shown in Eq. (3).

$$TC_{l,out} = \frac{\dot{m}_{l,out}}{\dot{m}_{k,in}} \quad (3)$$

TCs are generally given in $t_{output}$ / $t_{input}$. For a better illustration, the input and output flows to calculate the TCs are indicated as $TC_{input\text{-}flow \rightarrow output\text{-}flow}$. At the level of goods, only one TC was used for process P7. The $TC_{F\_2\text{-}7 \rightarrow F\_7\text{-}21}$ is 0.400 [$t_{output}$ / $t_{input}$] towards recycling for PW sorting plants sorting packaging of plastics and metals (Neubauer et al., 2019).

For two material flows of goods, the relations function in STAN as shown in Eq. (4) was used. Relations can be defined for flows or flow parameters. In this case, the material flows of Al oxidation losses by incineration (flows F_14-17 and F_14-20) were calculated at the substance level (Section 2.1.3). Then, it was defined that the mass of goods $m_i$ of the material flow $i$ equals 1.89 times the mass $m_{j,i}$ of substance $j$ (Al) in the same flow $i$. 1.89 is the conversion factor between Al and $Al_2O_3$ in [t/t].

$$\dot{m}_i = 1.89 \times \dot{m}_{(j,i)} \quad (4)$$

### 2.1.3. Substance flows of Al and Fe PW

To calculate the substance flows of Al and Fe metal packaging, mainly Eq. (5) was used:

$$\dot{m}_{ji} = \dot{m}_i \times c_{ji} \quad (5)$$

$\dot{m}_i$ is the material flow of a good $i$ and $c_{ji}$ is the concentration of substance $j$ in good $i$. $\dot{m}_{ji}$ is the material flow of a substance $j$ in a good $i$. Substance flows of Al and Fe PW generated came from BMK (2023), while these recycled from separate collection and sorting of PW waste were taken from Eurostat (2023). Fe PW recovered from IBA treatment in Austria (flow F_14-21) also came from Eurostat (2023). Since this amount of 8, 505 [t/yr] was reported, it was assumed that it consistet of 100 wt-% of Fe PW. Gritsch and Lederer (2023) provided concentrations of Al and Fe packaging in household MSW for Vienna, and Merstallinger and Fritz (2022) these in bulky MSW, commercial MSW, and construction waste for all nine Austrian provinces. The data in Gritsch and Lederer (2023) and Merstallinger and Fritz (2022) origins from waste sampling campaigns that also show the uncertainty levels of the concentration of Al and Fe packaging in waste flows, based on a confidence interval of 95 % (Beigl, 2020). These uncertainties were considered in the calculation (indicated as ±). Since both data sets are on a gross-basis, meaning that moisture and dirt are contained in the concentrations $c_{ji,gross}$, the net concentrations $c_{ji,net}$ without moisture and dirt for Al and Fe packaging had to be calculated. This was done after Beigl (2020) by applying a net-factor $NF_{ji}$ of 0.883 [t/t] (Eq. (6) and supplementary file Table S2).

$$c_{ji,net} = c_{ji,gross} \times NF_{ji} \quad (6)$$

In addition to Eq. (5), the transfer function (Eqs. (2) and (3)) using TCs was used. The $TC_{F\_2\text{-}7 \rightarrow F\_7\text{-}21}$ for Al and Fe PW separately collected and sorted for recycling in PW sorting plants (process P7) are 0.90 ± 0.02 for Al and 0.97 ± 0.02 for Fe (supplementary file, Table S3). The TCs come from PW sorting plants in Austria, Belgium and the UK (Cimpan et al., 2015; Gritsch and Lederer, 2023; Kleinhans et al., 2021) and the data and calculation is shown in Table S3 in the supplementary file. The $TC_{F\_3\text{-}8 \rightarrow F\_8\text{-}21}$ of mixed MSW sorting in Vienna (process P8) of

0.00 ± 0.00 for Al and 0.26 ± 0.00 for Fe was taken from Gritsch and Lederer (2023). For the $TC_{F\_4\text{-}9 \rightarrow F\_9\text{-}21}$, $TC_{F\_5\text{-}9 \rightarrow F\_9\text{-}21}$, and $TC_{F\_6\text{-}9 \rightarrow F\_9\text{-}21}$ in mixed and bulky MSW sorting in the other eight Austrian provinces (process P9), TCs of 0.635 ± 0.115 for Fe and 0.295 ± 0.155 for Al were used (Skutan and Brunner, 2006).

For by IBA treatment in Vienna, the $TC_{F\_10\text{-}12 \rightarrow T\_12\text{-}14}$ towards recycling is 0.900 ± 0.000 for Fe and 0.530 ± 0.000 for Al, respectively (Gritsch and Lederer, 2023). In the other eight provinces, the $TC_{F\_11\text{-}13 \rightarrow F\_13\text{-}14}$ of 0.34±0.15 for Al towards recycling was used. These TCs are the mean values ± standard deviations from eight IBA treatment plants in the EU (Allegrini et al., 2014; Biganzoli and Grosso, 2013; Gritsch and Lederer, 2023; Holm and Simon, 2017; Huber, 2020). Fe from IBA treatment was not necessary to be calculated by TCs since data on the total amount recovered in Austria (flow F_14-21) was available from Eurostat (2023). The data and calculation is shown in the supplementary file Table S4. MSWI oxidation losses for Al were assumed at 5–15 wt-% of the aluminum input into MSWI and IBA treatment (Hu et al., 2011), realized in the calculation in a subsequent modelled process P14. The based on this range determined $TC_{F\_12\text{-}14 \rightarrow F\_14\text{-}17}$ and $TC_{F\_12\text{-}14 \rightarrow F\_14\text{-}20}$ is 0.10 ± 0.05 towards landfilling.

Finally, the relation function was used to calculate the substance flows of material flow F_1-5.1 *household MSW AUT–VIE to collection* and F_1-5.3 *other MSW flows*. In the relations function in Eq. (7)

$$c_{jx} = c_{jy} \quad (7)$$

$c_{jx}$ is the concentration of a substance $j$ in the good $x$ and $c_j$ is the concentration of a substance $j$ in the good $y$. For both flows, the concentrations were assumed to be equal ($c_{j,F\_1\text{-}5.1} = c_{j,F\_1\text{-}5.3}$). The same was assumed also for other flows of mixed MSW ($c_{j,F\_5\text{-}9} = c_{j,F\_5\text{-}10} = c_{j,F\_5\text{-}11}$) and sorted MSW ($c_{j,F\_9\text{-}10} = c_{j,F\_9\text{-}11} = c_{j,F\_9\text{-}18}$).

According to the definition of the calculation point for recycling (Van Caneghem et al., 2019) further losses during the recycling process in metal smelters or furnace are not considered.

### 2.2. Contribution of metal packaging from MSW and IBA to the RR

For the RR including bulky and mixed MSW sorting and IBA treatment Eq. (8) was used

$$RR_j = \left(\left(\dot{m}_{j,sep\_coll} + \dot{m}_{j,rec\_MSW\&IBA}\right)/\dot{m}_{j,gen}\right) \times 100[\%] = \left(\left(\dot{m}_{j,F\_7-21} + \dot{m}_{j,F\_8-21} + \dot{m}_{j,F\_9-21} + \dot{m}_{j,F\_14-21}\right)/\dot{m}_{j,F\_0-21}\right) \times 100[\%] \quad (8)$$

where $RR_j$ is the RR for substance $j$, $\dot{m}_{j,sep\_coll}$ is the mass of the material flow of substance $j$ separately collected and sent to recycling, $\dot{m}_{j,rec\_MSW\&IBA}$ is the mass of the material flows of substance $j$ recovered from mixed MSW and IBA and sent to recycling, and $\dot{m}_{j,gen}$ is the mass of the material flow of substance $j$ generated. The second part of Eq. (7) only shows the material flows used in the particular case of this study. Since sorting losses in bulky and mixed MSW sorting, PW sorting, and IBA treatment, were considered, and also moisture and dirt contents as well as aluminum oxidation were subtracted, the calculation rules by EC (2019) should be fulfilled, as the losses are higher than the 14 wt.% suggested by EEA (2022).

## 3. Results and discussion

### 3.1. Material flows of goods, aluminum and ferrous metal packaging in Austria

#### 3.1.1. Material flows of goods

Fig. 2 shows the material flows of goods containing Al and Fe PW. With respect to the destination of the waste flows considered, construction waste (flow F_1-18) could not have been clearly traced to its destination. Thus, this waste flow and the therein contained Al and Fe PW were not considered. In reality, however, the portion of construction waste which contains Al and Fe packaging, is likely to be treated in mixed and bulky MSW sorting plants or directly incinerated (BMK, 2023). In this case, the therein contained Al and Fe PW was partially recovered for recycling. For bulky MSW, the destination of 120,000 [t/yr] was unclear (flow F_6-18). Former studies assumed that sorting

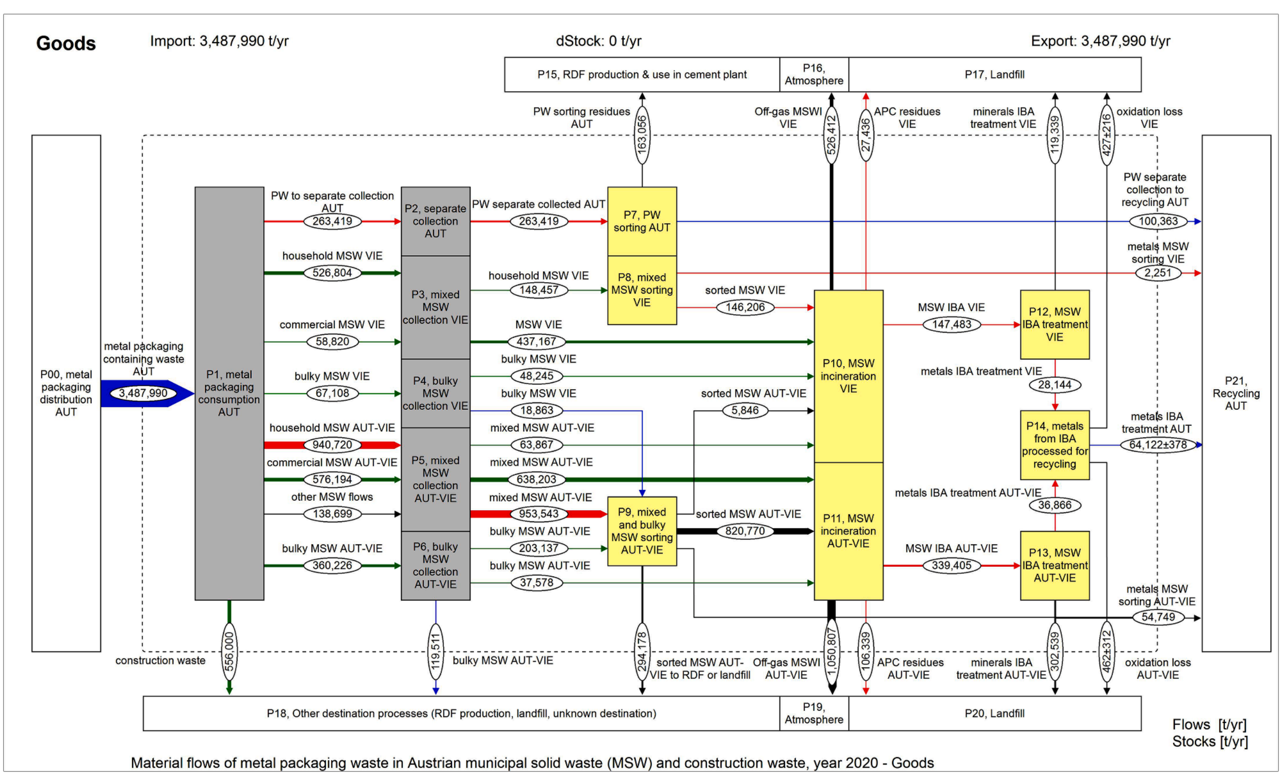


**Fig. 2.** Material flows of goods containing metal PW of Al and Fe in Austria for the year 2020. A comma (,) marks a thousand-separator.

residues from MSW sorting were either landfilled, or incinerated as RDF in cement kilns (Warrings and Fellner, 2018). The therein contained Al and Fe PW would then be lost for recycling. In fact, these residues are mainly incinerated in MSWI (flows F_8-10, F_9-10, F_9-11), allowing a recovery of the therein contained Al and Fe PW by IBA treatment (Kellner et al., 2022). Furthermore, residues from PW or mixed and bulky MSW sorting (flows F_7-15 and F_9-18) which are used to prepare RDF for the cement industry, undergo further treatment in RDF production plants and it is likely that Al and Fe PW is partially recovered in these plants and recycled.

### 3.1.2. Material flows of Al PW

Fig. 3 shows the material flows of Al PW. The bulk of Al PW is concentrated in three material flows, namely source-segregated LPW (flow F_1-2, ~8,840 [t/yr]), household MSW VIE (flow F_1-3.1, ~3,720 [t/yr]), and household MSW AUT–VIE (flow F_1-5.1, ~8,370 [t/yr]). These are altogether about ~20,930 [t/yr] or 87 % of all Al PW generated in Austria. Furthermore, the total amount of Al PW generated in Austria is almost equally divided among the three main treatment routes, which are sorting of separately collected PW (~8,840 [t/yr]), sorting of mixed and bulky MSW (~7,220 [t/yr]), and incineration without pre-treatment (~7,190 [t/yr]).

Mixed and bulky MSW sorting is a MSW treatment route that Austria has in common with countries like Cyprus, Greece, the Netherlands, and Spain, and like in these countries, considerable amounts of Al metal PW is recovered by this treatment technology (Bourtsalas and Themelis, 2022; Thoden van Velzen et al., 2020). A crucial question is the sorting efficiency, or, as termed in MFA, the TC of Al metal recovery. In this article, a TC of 0.295 ± 155 to recycling was used, based on the sophistic and rich in detail study of Skutan and Brunner (2006). The range of comparable figures in literature is, however, huge, and lies in between 0.12 in Madrid (Istrate et al., 2021), 0.29 in Barcelona (Cimpan et al., 2015), 0.33–0.95 in eight plants in Castilla y León (Montejo et al., 2013), and 0.78–0.83 in six plants in Cyprus, Greece, and Spain (Bourtsalas and Themelis, 2022). Even though most of these figures did likely not include moisture and dirt contents (which the study of Skutan and Brunner did), they indicate that higher TCs than the ones assumed in this study, are realistic in modern MSW sorting plants with better technology than the ones investigated by Skutan and Brunner (2006) 20 years ago. This, however, can only be judged when having better explanations for the large differences reported, but these explanations can hardly be found in literature. For this reason, the TCs for Fe and particularly Al PW in mixed and bulky MSW sorting should be further investigated, and traced back to different MSW inputs, plant designs and operation practices (Cimpan et al., 2016).

While mixed and bulky MSW sorting is present in Austria and other countries, a big difference to Cyprus and Greece for instance is that in Austria, most of the sorting residues from these plants are incinerated and not landfilled. The first allows a further recovery of metals from IBA. Furthermore, and in contrast to Spain and the Netherlands, the incineration of this fraction takes mainly place in fluidized bed combustion (FBC) rather than in grate incineration (GI) MSWI plants (Fellner et al., 2015; Leckner and Lind, 2020). This has considerable consequences with respect to the recovery of Al PW by IBA treatment. Al metals in the IBA from FBC are present in larger particle size fractions if compared to IBA from GI, and also less coated with minerals. The reasons for this are twofold: first a lower temperature of 630 °C in the fluidized bed if compared to over 1,000 °C on the grate; and second a dry IBA discharge in the FBC compared to a wet discharge in GI. As a consequence, Al metals in FBC IBA have a higher quality and can easier be recovered if compared to these from GI IBA (Blasenbauer et al., 2023). This is also shown by Mühl et al. (2023) who analysed mineral residues from IBA treatment, finding much less residual metal particles in IBAs from FBCs if compared to these from GIs. Since GIs operate under higher temperatures (Blasenbauer et al., 2023; Leckner and Lind, 2020), the oxidation rate of Al metals is likely higher (Hu et al., 2011). However, for FBC, hitherto no Al metal oxidation study exists. Since the same counts for Al recovery from FBC IBA treatment, future research on the behavior of Al

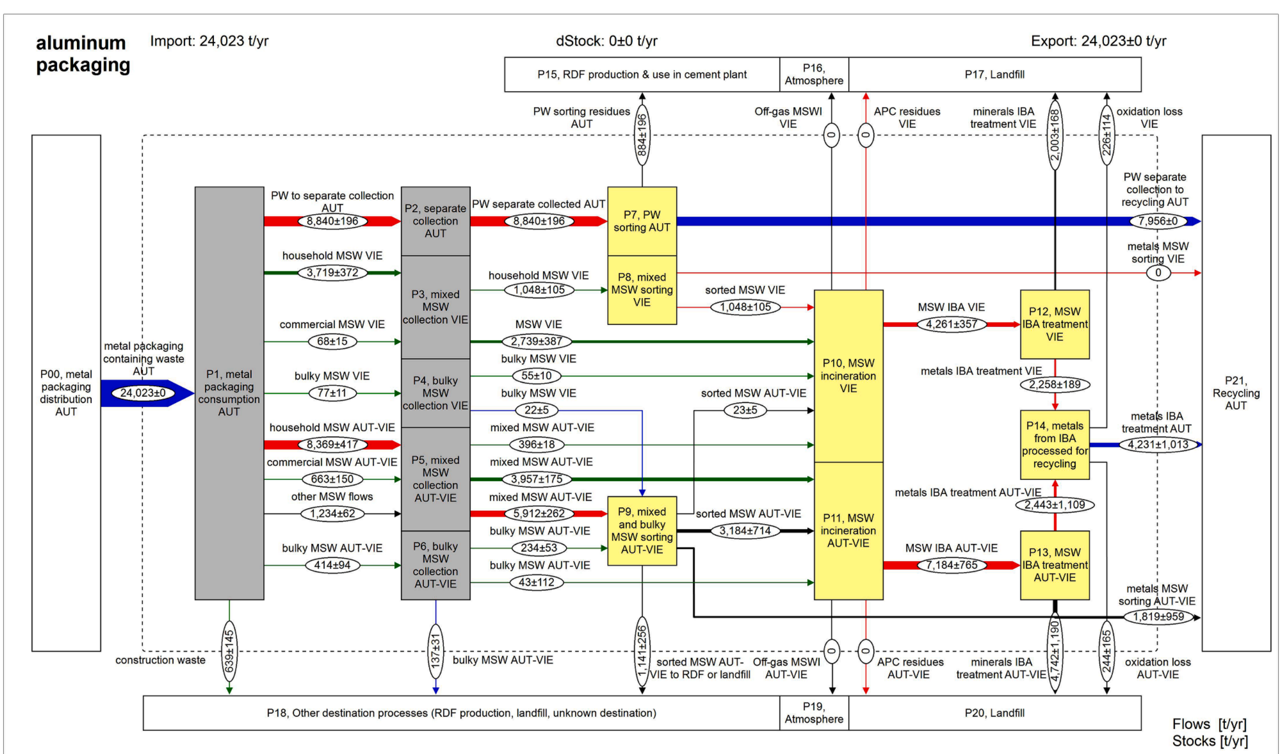


**Fig. 3.** Material flows of metal PW of Al in Austria for the year 2020. A comma (,) marks a thousand-separator.

packaging and other metals in MSW incineration and IBA treatment should also consider FBC. Finally, the IBA treatment plant whose improvement was studied in the experiment by Mühl et al. (2023), implemented the experimental design in industrial scale in the year 2021. Thus, higher amounts of metals are likely being recovered from the year 2022, and these amounts should be considered in reporting of Al PW recycled.

#### 3.1.3. Material flows of Fe metal PW

Fig. 4 shows the material flows of Fe PW. Most Fe PW is in PW to separate collection (flow *F_1-2*, ~27,260 [t/yr]), followed by household MSW AUT–VIE (flow *F_1-5.1*, ~7,130 [t/yr]) and household MSW VIE (flow *F_1-3.1*, ~4,190 [t/yr]). Since these flows altogether make ~38,580 [t/yr] or ~94 % of all Fe PW generated in Austria, the other material flows are of minor relevance. Furthermore, these figures already indicate that separate collection is of higher relevance for Al than for Fe PW, since Fe can easier be recovered from MSW and IBA (Bourtsalas and Themelis, 2022; Šyc et al., 2020). In detail, and if compared to literature, for mixed and bulky MSW sorting plants in Austria, the TC used for Austria without Vienna was 0.635 ± 0.115, which is higher than in Madrid (Istrate et al., 2021), in the same range as in Castillia y León (Montejo et al., 2013) and Barcelona (Cimpan et al., 2015), but lower than these reported by Bourtsalas and Themelis (2022). Again, the reasons for these differences, even though they are smaller than for Al metals, are unclear, may have to do with plant operation and outline, and should be investigated. A good example for this is the mixed MSW sorting plant in Vienna that only recovers 26 wt-% of the input. The main reason is that the mixed MSW is classified into three particle-size fractions (0–125 mm, 125–250 mm, ≥250 mm), of which only one fraction (125–250 mm) undergoes Fe separation. If the other particle-size fractions would also undergo Fe separation, a higher TC towards recycling would be realize at this stage. To proof this assumption, however, further analysis as well as experiments have to be carried out.

### 3.2. Concentrations of Al and Fe packaging waste in mixed MSW from households

A major challenge for calculating the amounts and RR of Al and Fe PW in Austria is the non-existence of the concentrations of these metals in the mixed MSW from households in eight of the nine Austrian provinces. In this study, these concentrations, all on a net-basis, were determined as 0.0089 ± 0.0004 [t/t] for Al PW and 0.0076 ± 0.0007 [t/t] for Fe PW for these eight provinces (see Fig. 5). Since the concentrations in Vienna were 0.0071 ± 0.0007 [t/t] for Al PW and 0.0080 ± 0.0008 [t/t] for Fe PW, the weighted average for Austria is 0.0083 ± 0.0007 [t/t] for Al PW and 0.0077 ± 0.0005 [t/t] for Fe PW. For both metals combined, the modelled weighted concentration is 0.0160 ± 0.0012 [t/t] in Austria. In comparison, the weighted concentration in Austria as determined by the household mixed MSW sorting analysis from all nine Austrian provinces as compiled by Beigl (2020) was 0.0150 [t/t], with a range from 0.0130 to 0.0190 [t/t]. This means that the modelled value of this study is slightly higher than in Beigl (2020), but well in the range of the minimum and maximum values. Fig. 5 shows the calculated concentrations in comparison to these measured by sampling, taken from Beigl (2020).

### 3.3. Modelled recycling rates of metal packaging in Austria

As shown in Fig. 6, Austria achieves the RR targets of the year 2025 for both Fe and Al PW. In case of Al PW, slight further improvements are needed to reach the recycling target for 2030.

The modelling results also show that IBA treatment contributes more to the RR than mixed MSW sorting, which is clear since more bulky and mixed MSW and thus also therein contained metal PWs are incinerated in MSWI plants than treated in MSW sorting plants. For Al packaging, the RR increased from 33 % to 58 % by including Al packaging from bulky and mixed MSW sorting and IBA treatment. The RR of Fe packaging increased from 65 % to 93 %. By maintaining this state, the RR

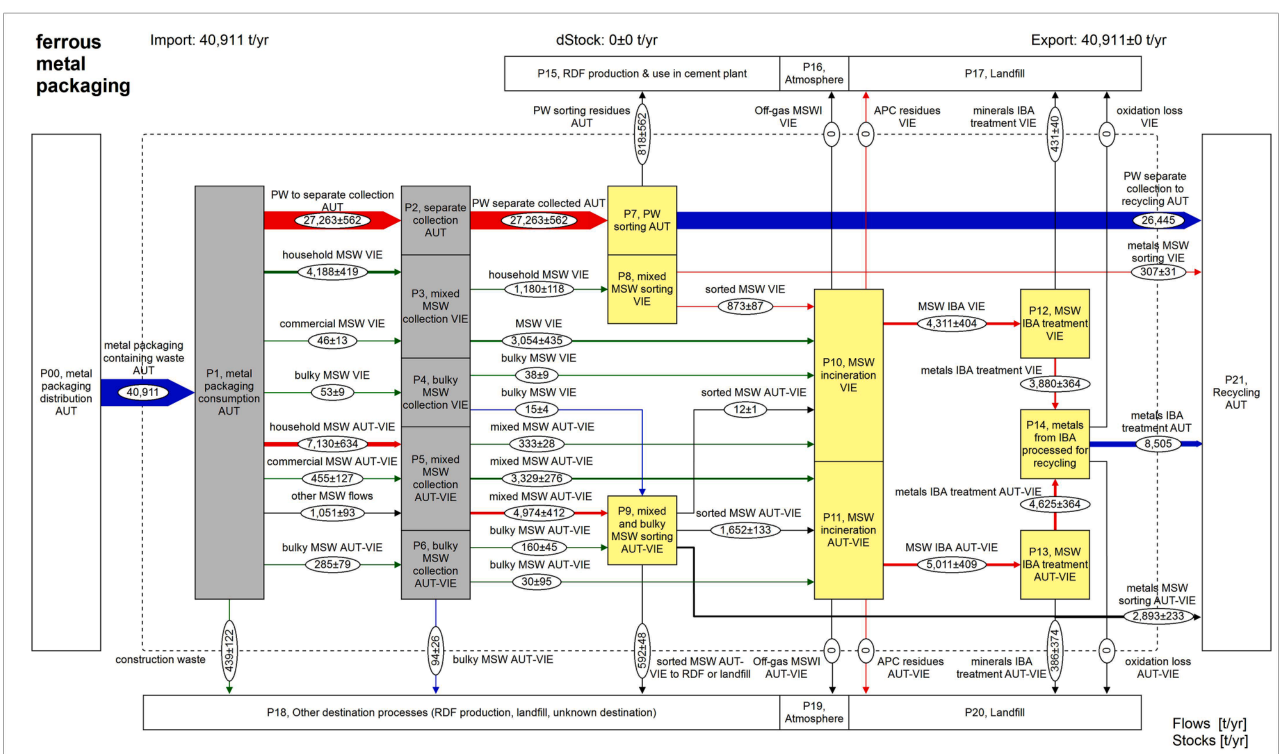


**Fig. 4.** Material flows of metal PW of Fe in Austria for the year 2020. A comma (,) marks a thousand-separator.

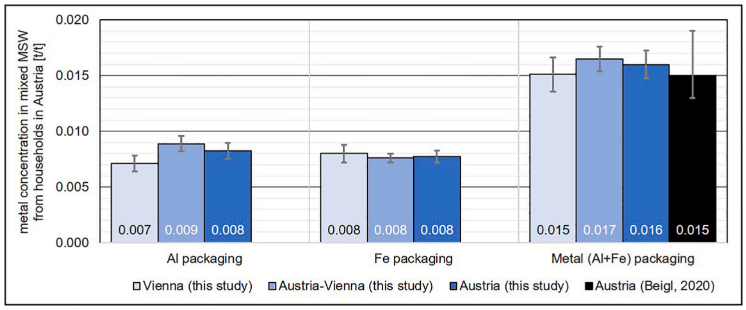


**Fig. 5.** Modelled concentrations of packaging metals in [t/t] in mixed MSW from households in this study in comparison to the values from Austrian waste sampling campaigns as compiled by Beigl (2020).

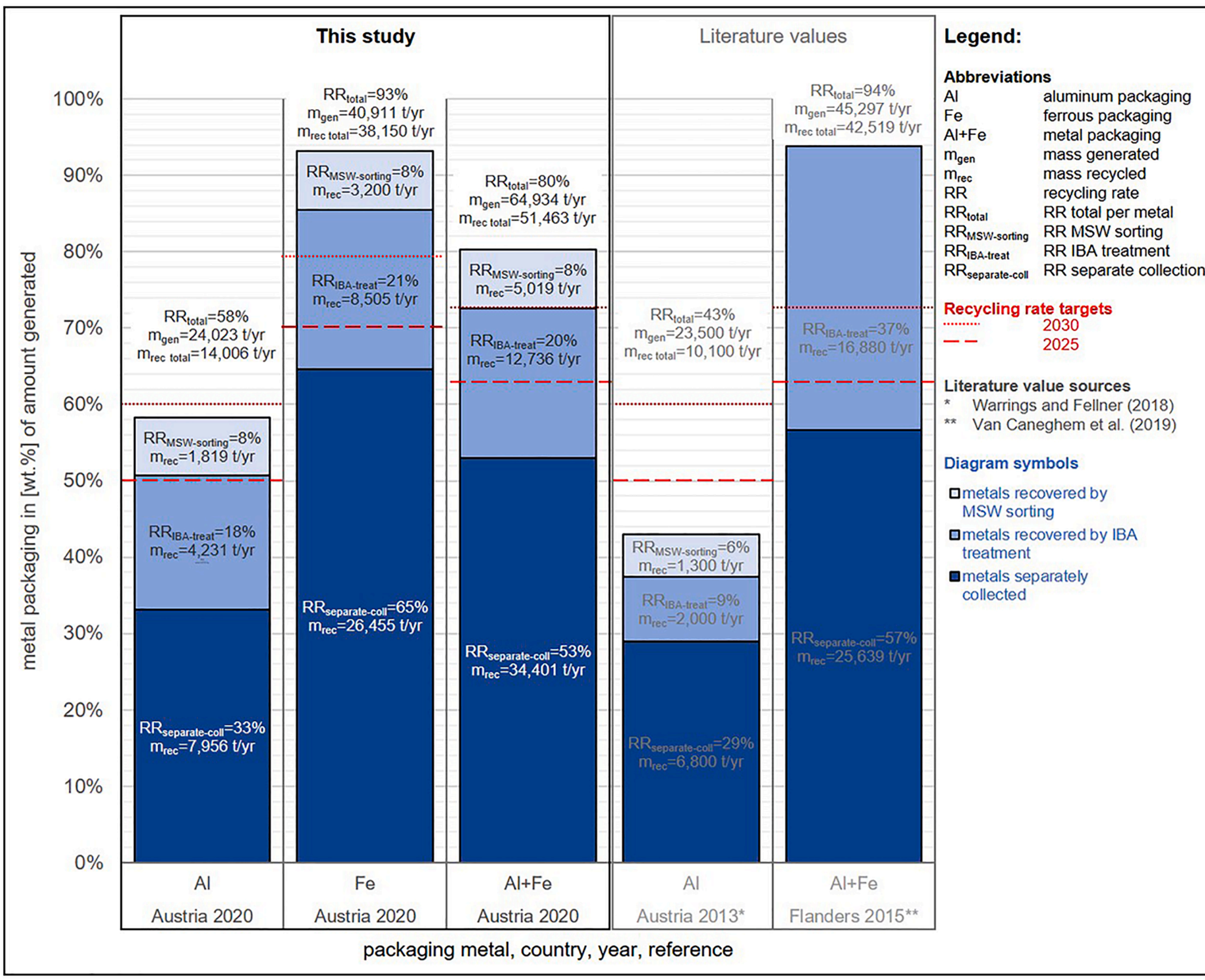


**Fig. 6.** Recycling rate (RR) for metals recycled total ($RR_{total}$), from mixed and bulky MSW sorting ($RR_{MSW\text{-}sorting}$), from IBA treatment ($RR_{IBA\text{-}treat}$) and from separate collection ($RR_{separate\text{-}coll}$), as well as generated mass ($m_{gen}$) and recycled mass ($m_{rec}$) of metal packaging in Austria (AUT) for the year 2020 based on the MFA in this study in comparison to the study of Warrings and Fellner (2018) for the year 2013, and data from the Belgian region of Flanders (FLA) in the year 2015, based on Van Caneghem et al. (2019).

target for Al packaging can be met for the year 2025, but not for the year 2030. For Fe packaging, the RR target for both, 2025 and 2030 can be met with current performance of the MSW management system in Austria. The modelled RR in this study for both metal PWs combined is then 80 %, which is 14 % higher than the 66 % reported by BMK (2023).

In comparison to other studies, the values for Al PW are higher in this study than in Warrings and Fellner (2018), mainly as these authors consider residues from bulky and mixed MSW sorting to be incinerated in cement kilns, and not in MSW incineration plants. As shown by Van Caneghem et al. (2019), RR values for metal packaging in this study are lower than these reported in Flanders in the year 2015. However, Van Caneghem et al. (2019) tried to crosscheck these reported data by sampling of metals from IBA treatment plants. They did not find as many packaging metals as reported and it is not clear whether the missing amounts were either not identifiable as packaging metals, or if they were simply not there. In comparison to the Netherlands, the results of this study show lower RRs for Al (58 % vs. 70-77 %) and Fe PW (93 % vs. 94-97 %). The difference can be explained by different assumptions on recovery rates of metals from IBA (Thoden van Velzen et al., 2020).

### *3.4. Options to increase metal packaging recycling in Austria*

To achieve higher RRs for Al packaging, some further improvements in separate collection and sorting technologies are required. While a harmonization of separate collection and a deposit-refund system, both implemented in 2025, target the first aspect (BMK, 2023), the second aspect has partly already been set in place. For instance, the modernization of one large IBA treatment plant as tested at pilot scale in Mühl et al. (2023) was implemented in the end of the year 2021. HTP (2020) also showed the potential of upgrading smaller and older waste sorting plants by additional Al removal in the fine and mid-range material stream, leading to an increase of the recovery of Al metals from 53 % to 82 %. Since also bulky and mixed MSW sorting plants likely operate with higher recovery efficiencies than as assumed in this study, PW sorting residues undergo metal recovery before being used as RDF, and construction wastes likely also be treated in MSW sorting plants with Al and Fe removal, it is very possible that also here the amounts sent to recycling were higher in reality than the modelling results of this study suggest. Thus, it is not only about improvements of technologies, but also about improving the data base to achieve higher RR for packaging metals in Austria and other countries.

This means that next to improved technology, more data is needed, particularly on the distribution of metals in mixed MSW and other waste streams, distinguishing between Al and Fe, and packaging and non-packaging; quantities and composition of Fe and Al recovered from mixed and bulky MSW sorting and IBA treatment; and disposal routes for construction waste. Based on this data, both, RRs for reporting, as well as TCs for material flow modelling, can be calculated.

### *3.5. The circular economy of metal packaging beyond recycling rates*

The CE is currently one of the most important strategies for resource conservation and sustainable development, and despite the claim for additional indicators, the RRs are an important metric to measure CE policies and implementation (Fellner and Lederer, 2020). However, RRs also have limitations to represent a circular economy in terms of quality restrictions or restrictions on use in certain applications. This is of particular relevance for metals recovered from IBA which contain more extraneous materials than their counterparts from separate collection, limiting their use to rebar or cast aluminum, but not high-quality products like packaging (Dworak and Fellner, 2021; Haupt et al., 2017; Løvik and Müller, 2014). As long as there is enough demand for such low-quality scrap, this might not be a big problem. However, this might change in the future, as shown by Buchner et al. (2017) for the case of aluminum: when the future car fleet consists of electric vehicles only, the demand for cast aluminum will be too low to absorb low-quality aluminum scrap. Considering these aspects might be the difference between a recycling rate based and a product based circular economy.

## 4. Conclusions

To assess the progress towards achieving the RR, a good knowledge base is required. This knowledge base, however, consists not only of data, but also of scientific methods how to establish these data. The EC calculation rules for RRs determination is one such method that brings clarification to reporting countries (Van Caneghem et al., 2019). These rules again require data established by science-guided methods, in particular waste sampling and the determination of the contents of materials such as Al and Fe PW in waste streams. If, as in the present case, these data are partly not direct available, other scientific methods can help to gain these data, and material flow analysis is one of them. However, the present case also highlighted some additional data gaps, particularly with respect to the transfer coefficients of mixed and bulky MSW sorting and IBA treatment plants. In order to improve the assessment of the achievement of the RR targets in general and the contribution to these by state-of-the-art waste treatment in particular, data gaps of both, waste composition and waste treatment, should be closed.

## CRediT authorship contribution statement

**Jakob Lederer:** Formal analysis, Investigation, Methodology, Project administration, Resources, Writing – original draft. **Dieter Schuch:** Conceptualization, Data curation, Investigation, Methodology, Supervision, Writing – review & editing.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

Data will be made available on request.

## Acknowledgements

The financial support by the Austrian Federal Ministry for Labour and Economy, the National Foundation for Research, Technology and Development and the Christian Doppler Research Association is gratefully acknowledged. Furthermore, we acknowledge the support of our company partners involved in this work, which were Abfallbehandlung Ahrental GmbH, Altstoff Recycling Austria AG, Brantner Österreich GmbH, Linz Service GmbH, and Wien Energie GmbH. In addition, we thank Magistratsabteilung MA48, Vienna’s public Waste Management provider, for its support. The authors also acknowledge TU Wien University Library for financial support through its Open Access Funding Program.

## Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2024.107461.

## References

Allegrini, E., Maresca, A., Olsson, M.E., Holtze, M.S., Boldrin, A., Astrup, T.F., 2014. Quantification of the resource recovery potential of municipal solid waste incineration bottom ashes. Waste Manag. 34 (9), 1627–1636. https://doi.org/10.1016/j.wasman.2014.05.003.

Amadei, A.M., Rigamonti, L., Sala, S., 2023. Exploring the EU plastic value chain: a material flow analysis. Resour., Conserv. Recycl. 197, 107105 https://doi.org/10.1016/j.resconrec.2023.107105.

Beigl, P., 2020. Auswertung der Restmüllzusammensetzung in Österreich 2018/2019 (Analysis of the mixed municipal solid waste composition in Austria 2018/2019). Universität für Bodenkultur, Institut für Abfallwirtschaft, Vienna.

Biganzoli, L., Grosso, M., 2013. Aluminium recovery from waste incineration bottom ash, andits oxidation level. Waste Manag. Res. 31 (9), 954–959. https://doi.org/10.1177/0734242x13493956.

Blasenbauer, D., Huber, F., Mühl, J., Fellner, J., Lederer, J., 2023. Comparing the quantity and quality of glass, metals, and minerals present in waste incineration bottom ashes from a fluidized bed and a grate incinerator. Waste Manag. 161, 142–155. https://doi.org/10.1016/j.wasman.2023.02.021.

BMK, 2023. Bundes-Abfallwirtschaftsplan (BAWP) 2023 Teil 1 (Federal waste management plan 2023 Part 1). Bundesministerin für Klimaschutz, Umwelt, Energie, Mobilität, Innovation und Technologie (BMK), Vienna.

Bourtsalas, A.C., Themelis, N.J., 2022. Materials and energy recovery at six European MBT plants. Waste Manag. 141, 79–91. https://doi.org/10.1016/j.wasman.2022.01.024.

Brunner, P.H., Rechberger, H., 2016. Handbook of Material Flow Analysis: For Environmental, Resource, and Waste Engineers. CRC press, Boca Raton.

Buchner, H., Laner, D., Rechberger, H., Fellner, J., 2017. Potential recycling constraints due to future supply and demand of wrought and cast Al scrap—A closed system perspective on Austria. Resour. Conserv. Recycl. 122, 135–142. https://doi.org/10.1016/j.resconrec.2017.01.014.
Cencic, O., Rechberger, H., 2008. Material flow analysis with software STAN. J. Environ. Eng. Manag. 18 (1), 440–447.
Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015. Central sorting and recovery of MSW recyclable materials: a review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manage. 156, 181–199. https://doi.org/10.1016/j.jenvman.2015.03.025.
Cimpan, C., Maul, A., Wenzel, H., Pretz, T., 2016. Techno-economic assessment of central sorting at material recovery facilities – the case of lightweight packaging waste. J. Clean. Prod. 112, 4387–4397. https://doi.org/10.1016/j.jclepro.2015.09.011.
Cooper, D.R., Ryan, N.A., Syndergaard, K., Zhu, Y., 2020. The potential for material circularity and independence in the US steel sector. J. Ind. Ecol. 24 (4), 748–762. https://doi.org/10.1111/jiec.12971.
Dworak, S., Fellner, J., 2021. Steel scrap generation in the EU-28 since 1946 – sources and composition. Resour. Conserv. Recycl. 173, 105692 https://doi.org/10.1016/j.resconrec.2021.105692.
Dworak, S., Rechberger, H., Fellner, J., 2022. How will tramp elements affect future steel recycling in Europe? – A dynamic material flow model for steel in the EU-28 for the period 1910 to 2050. Resour., Conserv. Recycl. 179, 106072 https://doi.org/10.1016/j.resconrec.2021.106072.
EC, 2019. Commission implementing decision (EU) 2019/665 of 17 April 2019 amending decision 2005/270/EC establishing the formats relating to the database system pursuant to European Parliament and Council Directive 94/62/EC on packaging and packaging waste, in: (EC), E.C. (Ed.). Brussels.
EEA, 2022. Many EU Member States not on track to meet recycling targets for municipal waste and packaging waste. European Environment Agency (EEA), Copenhagen.
EEA, 2022. Early warning assessment related to the 2025 targetsfor municipal waste and packaging waste. Country profile Austria. European Environment Agency (EEA), Copenhagen.
EU, 2018. Directive (EU) 2018/852 of the European Parliament and of the Council of 30 May 2018 amending directive 94/62/EC on packaging and packaging waste. European Union, Brussels.
European Aluminium, 2018. Environmental profile report: life-Cycle inventory data for aluminium production and transformation processes in Europe. Report. Brussels, Belgium: European Aluminium.
Eurostat, 2023. Packaging waste by waste management operations [ENV_WASPAC_custom_7026780], in: Eurostat (Ed.). Brussels.
Fellner, J., Lederer, J., 2020. Recycling rate – The only practical metric for a circular economy? Waste Manag. 113, 319–320. https://doi.org/10.1016/j.wasman.2020.06.013.
Fellner, J., Lederer, J., Purgar, A., Winterstetter, A., Rechberger, H., Winter, F., Laner, D., 2015. Evaluation of resource recovery from waste incineration residues – the case of zinc. Waste Manag. 37, 95–103. https://doi.org/10.1016/j.wasman.2014.10.010.
Gaustad, G., Krystofik, M., Bustamante, M., Badami, K., 2018. Circular economy strategies for mitigating critical material supply issues. Resour. Conserv. Recycl. 135, 24–33. https://doi.org/10.1016/j.resconrec.2017.08.002.
Gavrilescu, D., Seto, B.-C., Teodosiu, C., 2023. Sustainability analysis of packaging waste management systems: a case study in the Romanian context. J. Clean. Prod., 138578 https://doi.org/10.1016/j.jclepro.2023.138578.
Gritsch, L., Lederer, J., 2023. A historical-technical analysis of packaging waste flows in Vienna. Resour., Conserv. Recycl. 194, 106975 https://doi.org/10.1016/j.resconrec.2023.106975.
Haupt, M., Vadenbo, C., Hellweg, S., 2017. Do we have the right performance indicators for the circular economy?: insight into the Swiss waste management system. J. Ind. Ecol. 21 (3), 615–627. https://doi.org/10.1111/jiec.12506.
Holm, O., Simon, F.-G., 2017. Innovative treatment trains of bottom ash (BA) from municipal solid waste incineration (MSWI) in Germany. Waste Manag. 59, 229–236. https://doi.org/10.1016/j.wasman.2016.09.004.
HTP, 2020. The ideal aluminium packaging sorting model. Report. Aachen, Germany: HTP GmbH & Co. KG.
Hu, Y., Bakker, M.C.M., de Heij, P.G., 2011. Recovery and distribution of incinerated aluminum packaging waste. Waste Manag. 31 (12), 2422–2430. https://doi.org/10.1016/j.wasman.2011.07.021.
Huber, F., 2020. Modelling of material recovery from waste incineration bottom ash. Waste Manag. 105, 61–72. https://doi.org/10.1016/j.wasman.2020.01.034.
Istrate, I.-R., Galvez-Martos, J.-L., Dufour, J., 2021. The impact of incineration phase-out on municipal solid waste landfilling and life cycle environmental performance: case study of Madrid, Spain. Sci. Total Environ. 755, 142537 https://doi.org/10.1016/j.scitotenv.2020.142537.
Kellner, M., Schindler, I., Jany, A., 2022. Status report waste incineration. Umweltbundesamt (Federal Environment Management Agency), Report No. 0831. Vienna. Download from: https://www.umweltbundesamt.at/fileadmin/site/publikationen/rep0831.pdf.
Kleinhans, K., Hallemans, M., Huysveld, S., Thomassen, G., Ragaert, K., Van Geem, K.M., Roosen, M., Mys, N., Dewulf, J., De Meester, S., 2021. Development and application of a predictive modelling approach for household packaging waste flows in sorting facilities. Waste Manag. 120, 290–302. https://doi.org/10.1016/j.wasman.2020.11.056.
Leckner, B., Lind, F., 2020. Combustion of municipal solid waste in fluidized bed or on grate – a comparison. Waste Manag. 109, 94–108. https://doi.org/10.1016/j.wasman.2020.04.050.
Lederer, J., Bartl, A., Blasenbauer, D., Breslmayer, G., Gritsch, L., Hofer, S., Lipp, A.-M., Mühl, J., 2022. A review of recent trends to increase the share of post-consumer packaging waste to recycling in Europe. Detritus 19, 3–17. https://doi.org/10.31025/2611-4135/2022.15198.
Lederer, J., Gassner, A., Kleemann, F., Fellner, J., 2020. Potentials for a circular economy of mineral construction materials and demolition waste in urban areas: a case study from Vienna. Resour., Conserv. Recycl. 161, 104942 https://doi.org/10.1016/j.resconrec.2020.104942.
Løvik, A.N., Müller, D.B., 2014. A material flow model for impurity accumulation in beverage can recycling systems. In: Grandfield, J. (Ed.), Light Metals. Springer, Cham. https://doi.org/10.1007/978-3-319-48144-9_151, 2014.
Merstallinger, M., Fritz, G., 2022. Ermittlung der Verpackungsanteile und Verpackungsmassen im Sperrmüll und im Gewerbeabfall 2022 (Determination of the packaging contents and packaging masses in bulky waste and commercial waste 2022). Technisches Büro Merstallinger, FHAnalytik, Vienna, Korneuburg.
Montejo, C., Tonini, D., Márquez, M.d.C., Fruergaard Astrup, T., 2013. Mechanical–biological treatment: performance and potentials. An LCA of 8 MBT plants including waste characterization. J. Environ. Manag. 128, 661–673. https://doi.org/10.1016/j.jenvman.2013.05.063.
Morf, L.S., Brunner, P.H., Spaun, S., 2000. Effect of operating conditions and input variations on the partitioning of metals in a municipal solid waste incinerator. Waste Manag. Res. 18 (1), 4–15. https://doi.org/10.1034/j.1399-3070.2000.00085.x.
Mühl, J., Skutan, S., Stockinger, G., Blasenbauer, D., Lederer, J., 2023. Glass recovery and production of manufactured aggregate from MSWI bottom ashes from fluidized bed and grate incineration by means of enhanced treatment. Waste Manag. 168, 321–333. https://doi.org/10.1016/j.wasman.2023.05.048.
Neubauer, C., Öhlinger, A., 2007. Mechanische Abfallbehandlung (MA) von gemischten Siedlungs-und Gewerbeabfällen in Österreich (Mechanical waste treatment of mixed municipal solid waste and commercial waste in Austria). Environment Agency Austria, Vienna.
Neubauer, C., Stoifl, B., Tesar, M., Thaler, P., 2020. Sortierung und recycling von Kunststoffabfällen in Österreich: status 2019. Umweltbundesamt (Federal Environment Agency), Vienna.
Riber, C., Christensen, T.H., 2007. Indirect determination of chemical composition and fuel characteristics of solid waste: methodology and application. working paper. Environment & resources DTU. Technical University of Denmark., Lyngby. Download from https://orbit.dtu.dk/en/publications/indirect-determination-of-chemical-composition-and-fuel-character.
Schuch, D., Lederer, J., Fellner, J., Scharff, C., 2023. Separate collection rates for plastic packaging in Austria – a regional analysis taking collection systems and urbanization into account. Waste Manag. 155, 211–219. https://doi.org/10.1016/j.wasman.2022.09.023.
Skutan, S., Brunner, P.H., 2006. Stoffbilanzen Mechanisch-Biologischer Anlagen zur Behandlung von Restmüll SEMBA (Materialbalances of Mechanical-Biological Plants for the Treatment of Mixed Municipal Solid Waste SEMBA) Technische Universität Wien. Download from. https://www.bmk.gv.at/dam/jcr:493bbae5-18c2-4761-ab41-20ab0ea31431/Studie_SEMBA.pdf.
Šyc, M., Simon, F.G., Hykš, J., Braga, R., Biganzoli, L., Costa, G., Funari, V., Grosso, M., 2020. Metal recovery from incineration bottom ash: state-of-the-art and recent developments. J. Hazard. Mater. 393, 122433 https://doi.org/10.1016/j.jhazmat.2020.122433.
Tallentire, C.W., Steubing, B., 2020. The environmental benefits of improving packaging waste collection in Europe. Waste Manag. 103, 426–436. https://doi.org/10.1016/j.wasman.2019.12.045.
Thoden van Velzen, E.U., Brouwer, M.T., Smeding, I.W., 2020. Recyclingpercentage metaalverpakkingen in Nederland in 2017: Volgens de nieuwe EU rekenmethode (Metal packaging recycling rate in the Netherlands in 2017: according to the new EU calculation method). Rep. Wageningen Food Biobased Res. 2160 https://doi.org/10.18174/536135.
Turner, D.A., Williams, I.D., Kemp, S., 2015. Greenhouse gas emission factors for recycling of source-segregated waste materials. Resour. Conserv. Recycl. 105, 186–197. https://doi.org/10.1016/j.resconrec.2015.10.026.
Van Caneghem, J., De Coster, E., Vandenbergh, K., De Broyer, S., Lambrix, N., Weemaels, L., 2019. Closing the household metal packaging cycle through recovery of scrap from waste-to-energy bottom ash: the case study of Flanders. Resour. Conserv. Recycl. 144, 115–122. https://doi.org/10.1016/j.resconrec.2019.01.028.
Warrings, R., Fellner, J., 2018. Current status of circularity for aluminum from household waste in Austria. Waste Manag. 76, 217–224. https://doi.org/10.1016/j.wasman.2018.02.034.
Warrings, R., Fellner, J., 2019. Management of aluminium packaging waste in selected European countries. Waste Manag. Res. 37 (4), 357–364. https://doi.org/10.1177/0734242X19832368.