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Full length article

The circular economy of packaging waste in Austria: An evaluation based on statistical entropy and material flow analysis

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A B S T R A C T

Transitioning to a circular economy needs robust data, feasible indicators, and practicable evaluation methods. This paper analyses Austria's 2020 packaging waste flows, assesses capture, recycling, incineration, and landfilling, and tests statistical entropy analysis (SEA) as an alternative evaluation tool. Results indicate that Austria will attain EU recycling targets for total packaging (68 %), aluminium (61 %), ferrous metals (96 %), glass (82 %) and paper (80 %); plastics (25 %) and aluminium oxidation present substantial challenges. SEA effectively highlights the material concentration of aluminium, ferrous metals and glass in recycling streams but reveals that Austria's waste management system disperses plastics and paper due to incineration. Further research should improve analyses of combustibles and integrations of energy recovery and material substitution. The study highlights that recycling alone is inadequate for achieving a circular economy. Essential components include design for recycling, reuse, and reduction. Although difficult to quantify and frequently overlooked, they are vital for sustainable resource management.

1. Introduction

The concept of circular economy aims at increasing resource efficiency through waste reduction, reuse, recycling, and sustainable consumption and production (Corona et al., 2019; Ellen MacArthur Foundation 2019; Hartley et al., 2024). Hence, the EU's Circular Economy Package includes ambitious recycling rates, particularly for packaging waste (PW), which is mainly contained in municipal solid waste (MSW). By 2025, 65 % of all PW and 50 % of plastic, 25 % of wood, 70 % of ferrous metals (Fe), 50 % of aluminium (Al), 70 % of glass, and 75 % of paper and cardboard PW must be recycled (EU, 2018a). In this context, collecting and reporting waste and recycling data to Eurostat is crucial (EC, 2019a; EEA, 2016), and thorough comprehension of material movements within the economy is essential for precise calculations of recycling rates, deriving insights into environmental performances, and measuring goals (Amadei et al., 2023; Van Eygen et al., 2018).

Currently, the quality and accuracy of the reported Eurostat data present considerable insufficiencies. Lederer and Schuch (2024) state that Eurostat (2023) reveals reported quantities for Fe PW retrieved from incineration bottom ash (IBA) in Austria for 2020, but corresponding data for Al are absent despite recovery being practised (Warrings and Fellner, 2018). The authors attribute the problem to data availability issues, which are also prevalent in other EU countries (Bruno et al., 2021; Fletcher and Dunk, 2023). Furthermore, metal PW recovered from mixed MSW was not considered within calculated recycling rates (Lederer and Schuch, 2024). Similarly, deficient information on the fate and alternative routes of other PWs, such as glass and paper, might lead to neglecting recycling potentials (Mühl et al., 2024) or misguided reporting practices (Van Caneghem et al., 2019). This aspect also holds importance for member states that employ mixed MSW sorting and recovery from IBA, such as Cyprus, Germany, Greece, the Netherlands, Norway and Spain (Blasenbauer et al., 2024; Cimpan et al., 2015; Edo et al., 2020; Lederer et al., 2022; Lederer and Schuch, 2024; Picuno et al., 2021a; (Thanos) Bourtsalas and Themelis, 2022; Thoden van Velzen et al., 2021).

Due to the ability to map and display such flows, material flow analysis (MFA) is pivotal for evaluating PW management systems and targets (Allesch and Brunner, 2015). Many MFA studies looked at plastic

Abbreviations: Al, Aluminium; Fe, Ferrous Metals; IBA, Incineration Bottom Ash; MBT, Mechanical Biological Treatment; MFA, Material Flow Analysis; MRF, Material Recovery Facility; MSW, Municipal Solid Waste; MSWI, Municipal Solid Waste Incineration; PW, Packaging Waste; RDF, Refuse Derived Fuel; SEA, Statistical Entropy Analysis; TC, Transfer Coefficient.

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PW with different scopes, e.g. EU level (Amadei and Ardente, 2022; Amadei et al., 2023; Antonopoulos et al., 2021; Eriksen et al., 2020; Kawecki et al., 2018) or countries and economic units (Brouwer et al., 2018, 2019; Callewaert et al., 2023; Gonçalves et al., 2024; Jang et al., 2020; Lombardi et al., 2021; Lopez-Aguilar et al., 2022; Madden and Florin, 2024; Picuno et al., 2021b; Pimentel Pincelli et al., 2021; Schneider et al., 2022; Schmidt and Laner, 2021; Thomassen et al., 2022; Van Eygen et al., 2018). Regarding metal, glass and paper PW, research has been done in Austria (Gritsch and Lederer, 2023; Lederer and Schuch, 2024; Warrings and Fellner, 2018), the Netherlands (Thoden van Velzen et al., 2020), Flanders (Van Caneghem et al., 2019), Norway (Mattson et al., 2024) and the EU (Dworak et al., 2022; Passarini et al., 2018; Tallentire and Steubing, 2020). Hitherto, MFAs fall short of distinguishing between PW and other metals (Dworak et al., 2022; Passarini et al., 2018; Warrings and Fellner, 2018), differentiating between Fe and Al (Tallentire and Steubing, 2020; Van Caneghem et al., 2019), and considering mixed waste sorting or IBA recovery (Tallentire and Steubing, 2020). Other studies are based on city level only (Gritsch and Lederer, 2023) or are built upon lower data resolution and do not include glass, paper or plastic PW (Lederer and Schuch, 2024). Considering that data qualities substantially affect the results and comparability of MFAs on PW in the EU and its member states, a consistent approach to data collection and MFA modelling for all major PW materials is required, as performed by Mattson et al. (2024) for Norway.

However, the information content of recycling rates is restricted to aspects of material efficiency whilst failing to consider system efficiencies and resource effectiveness (Schmidt and Laner, 2023). For this reason, some authors calculate not only recycling rates but also other MFA-based indicators like the separate collection rate, which forms an important base for the recycling rate, as well as the waste incineration rate and the landfilling rate. The Austrian Circular Economy Strategy hereby mentions applying statistical entropy analysis (SEA) as an alternative indicator to measure the circular economy of wastes (BMK 2022, 2024). SEA is a method to assess and quantify a system's ability to concentrate or dilute substances; it integrates MFAs with statistical entropy functions to evaluate substance distribution within a given system (Rechberger and Brunner, 2002).

Thus far, SEA has been conducted on copper, phosphorous, and nitrogen flows to evaluate their resource use efficiency on a macro-systems level (Laner et al., 2017; Rechberger and Graedel, 2002; Tanzer and Rechberger, 2020; Yue et al., 2009), as well as on smartphones and buildings, where SEA was used to reveal and assess the interplay between design and recyclability (Roithner et al., 2022a, 2022b), and further reuse and recyclability (Parchomenko et al., 2023). Additionally, studies have been carried out on the resource effectiveness of the European automotive sector, evaluating resource utilisation and functionality losses of materials and products over time (Parchomenko et al., 2021). In the case of plastic waste, SEA has been employed as a tool to measure the complexity of plastics, aiming to identify the products that most significantly impact the separation complexity of mixed plastic waste (Nimmegeers and Billen, 2021), in order to predict (Nimmegeers et al., 2021) and assess (Moyaert et al., 2022) its recyclability. Moreover, to compare mechanical and chemical recycling on different material categorisation levels (Skelton et al., 2022), and with regards to determining substance losses and enhancing plastic recycling processes (Compart and Gräbner, 2024). Prevalent SEA studies consider waste management typically as one or two stages within the resource consumption of economic entities (Meylan et al., 2017; Parchomenko et al., 2020; Rechberger and Graedel, 2002; Tanzer and Rechberger, 2020). Other contributions focus on isolated waste management practices: Velázquez Martínez et al. (Velázquez Martínez et al., 2019a,b) studied sieving processes and pre-processing stages to optimise lithium-ion battery recycling, and Rechberger (2001) employed SEA on the distribution of cadmium, lead and copper in the case of IBA utilisation for cement production.

Previously, SEA was not tested as a method for PW or MSW management evaluation or used towards Al, Fe, glass or paper PW. This study fills this gap by employing SEA on Austria's PW flows - marking its first application on a comprehensive national waste management system - building on the method's recognition in the Austrian Circular Economy Strategy as a metric for assessing circularity through the quantification of material dissipation (BMK, 2022, 2024). Thus, the study not only showcases the applicability of SEA at the systemic level but also aligns with national strategic priorities and provides a benchmark for assessing waste management system circularity. By integrating SEA with MFA, the study quantifies statistical entropy within material flows across all PW. This approach further reveals systemic inefficiencies and identifies opportunities to enhance material recovery and recycling processes. Therefore, this study investigates the following research questions: (1) What are the material flows of PW in Austria, and how does Austria's PW management perform based on MFA indicators? (2) What is the statistical entropy of PW management in Austria?

2. Materials and methods

The overarching methodological framework is designed to provide a comprehensive and systematic approach to analysing Austria's PW management system. Integrating MFA and SEA enables both the quantitative mapping of material flows and the evaluation of system performance with respect to resource concentration and dispersion. As a first step, an MFA model was developed and populated with data to quantify the material flows within the system. To evaluate the fate and treatment pathways of different PW materials - Al, Fe, glass, paper and plastics - key MFA-based indicators were calculated, including metrics for separate collection, recycling, incineration, and landfilling. Lastly, the concentration or dispersion of PW within the system was assessed using SEA, providing a detailed analysis of the system's ability to manage and recover PW effectively.

2.1. Material flow analysis of PW

2.1.1. Methodological basis

The method of MFA examines the status and change of material flows and stocks within a spatially and temporally defined system, guided by the principle of mass conservation (Brunner and Rechberger, 2016). In this study, the software STAN 2.6.801 (subSTance flow ANalysis), which employs error propagation and data reconciliation to quantify uncertainties (Cencic, 2016; Laner et al., 2014), was utilised according to the standard ÖNORM S 2096 (Cencic and Rechberger, 2008). Thereby, a process is defined as material dispersion, concentration, transformation, alteration or storage. The respective in- and outputs of processes are characterised by their mass flows (Rechberger and Brunner, 2002). Generally, flows are denoted by their process of origin and destination and are defined as goods (e.g. mixed MSW) comprising different subgoods (Brunner and Rechberger, 2016) (e.g. plastics or paper). Flows ($\dot{m}_{i,k}$) of subgoods (k) are calculated via their concentration ($c_{i,k}$) within a flow ($\dot{m}_i$) of goods:

$$\dot{m}_{i,k} = c_{i,k} \cdot \dot{m}_i \tag{1}$$

Additionally, each process is characterised by a set of transfer coefficients (TCs), the fractioning of specific goods or subgoods within the process, either for individual inputs or the total input (Brunner and Rechberger, 2016). The TC of a good is determined by dividing the output flow of that good ($\dot{m}_{i,out}$) by the total input of the same good ($\sum_i \dot{m}_{i,\ in}$). Likewise, TCs can be computed for subgoods.

$$TC_i = \frac{\dot{m}_{i,out}}{\sum_i \dot{m}_{i,\ in}} \tag{2}$$

$$\sum_i TC_i = 1 \tag{3}$$

2.1.2. Scope, model and system boundaries

Considering the MSW management in Austria, the following processes were modelled: mechanical-biological treatment (MBT), material recovery facilities (MRF) for separately collected (single-stream/commingled) and mixed waste, MSW incineration (MSWI), IBA sorting plants with inputs distinguished by firing technologies - grate and fluidised bed incineration - landfilling and recycling processes. Production and use as refuse-derived fuels (RDFs), e.g. in cement kilns, fall beyond system boundaries as they are defined as products rather than waste. The resulting MFA system is displayed in Fig. 1.

The following flows containing metal, glass, paper or plastic PW in Austria were included in the calculations: single-stream (glass and metal PW) and commingled separately collected waste (lightweight (LW)PW comprising metal together with plastic PW, paper PW along with other non-packaging paper waste), mixed MSW (household and commercial), bulky and construction waste. For these flows of goods, several omissions had to be made: Excluded waste flows contain organic waste due to small concentrations of PW (BMK, 2023), metal scrap containers at recycling yards and informal PW collection activities owning to low reported occurrence and relevance in Austria (Ramusch et al., 2015), deposit-return systems which exist mainly for glass, and littering due to low overall amounts of PW and data accuracy (Stoifl and Oliva, 2020). Public waste bins and street sweepings were not modelled explicitly, as these are treated alongside mixed MSW (Kladnik et al., 2024). All separately collected waste enters a single-stream/commingled MRF and is subsequently recycled or used as RDF, equalling system exports. Respective sorting rejects are incinerated or fed into a mixed waste MRF. In Austria, no untreated MSW is landfilled; mixed MSW, bulky, and construction waste are either directly incinerated, directed to a mixed waste MRF or an MBT facility. Outputs from the respective mixed waste MRFs/MBTs are recycled, exported as RDF, incinerated or landfilled. Occasionally, mixed waste MRF outputs are dried and treated in MBTs. After incineration, all IBA from grate and fluidised bed incineration are sent to stationary IBA treatment plants or mobile set-ups before final disposal in landfills.

Concerning subgoods, Fe, Al, glass, paper and plastic PW were encompassed in the MFA model. Residual and dirt contents were subtracted, only net contents were considered. Multilayer PW, such as beverage cartons, metallised and laminated plastic films, and minor alloy elements, were neglected due to system scope and complexity.

2.1.3. Input data curation

In general, data were selected based on criteria of completeness, temporal and geographical correlation, which means that representative data, data with <3 years of difference from the year of study and data from the exact area under study were chosen (Weidema and Wesnæs, 1996). With regard to the material flow of goods, information about

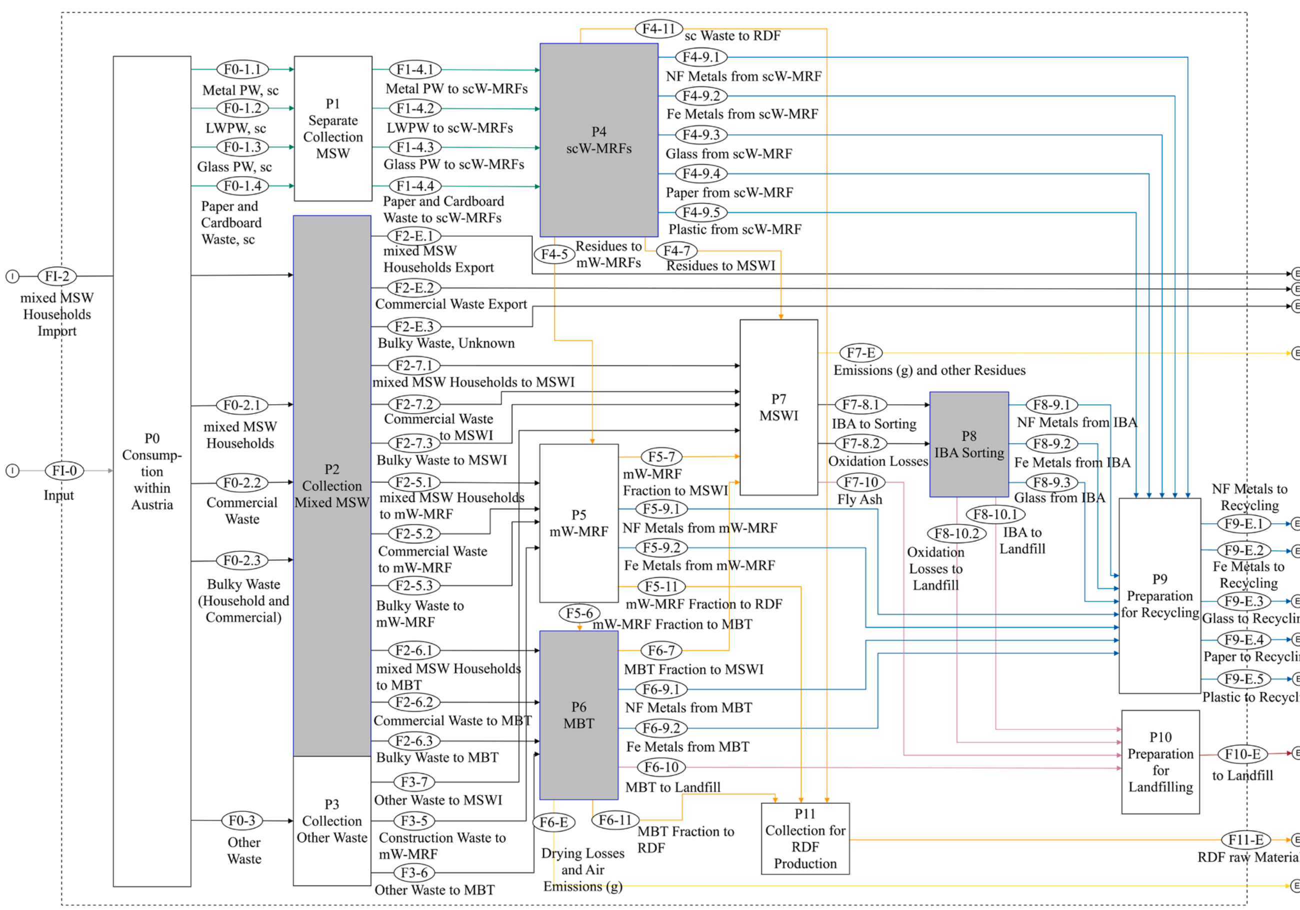

Fig. 1. Material flow analysis (MFA) model of the Austrian municipal solid waste (MSW) management system 2020 for the calculation of Al, Fe, glass, paper and plastic packaging waste (PW) flows. Rectangles represent processes (P), with grey ones indicating sub-systems detailed in Supplementary Material 1, Figures S-1 to S-4. Flows (F) are colour-coded: green for separately collected streams, blue for recyclables, black for mixed and other wastes, orange for mechanical treatment fractions, yellow for gaseous emissions, and pink for landfill flows. Dotted lines mark system boundaries. Horizontal flows are labelled above/below, vertical flows on the sides. Intersecting flows do not interact; changes occur only within processes. Abbreviations: ferrous (Fe), incineration bottom ash (IBA), mechanical biological treatment (MBT), municipal solid waste (MSW), municipal solid waste incineration (MSWI), material recovery facility (MRF), mixed waste (mW), non-ferrous (NF), refuse derived fuel (RDF), separately collected (sc), separately collected waste (scW).

waste generation, collection and MBTs was acquired from the Austrian Federal Waste Management Plan (BMK, 2023). Data concerning MSWI were extracted from the status report of waste incineration (Kellner et al., 2022). Another status report relating to the sorting and recycling of plastic waste in Austria delivered TCs for lightweight PW MRFs (Neubauer et al., 2021); TCs for glass MRFs were determined according to input specifications for glass sorting plants and smelter operators (BV Glas, BDE, bvse, 2014; bvse, BDE, 2013), whereas TCs for IBA treatment originate from Mühl et al. (2024) and Gritsch and Lederer (2023). Paper MRFs were characterised by the TCs calculated via the Austrian Federal Waste Management Plan data (BMK, 2023). Due to insufficient and unavailable data on mixed waste MRFs, primary data collection was performed. Therefore, mixed waste MRF operators were contacted to disclose information about waste inputs, outputs, origins and destinations. For all facilities that did not provide data, waste was allocated corresponding to their geographical location according to the Austrian principle of waste disposal self-sufficiency of the federal states. For MBTs and MSWI, the inputs were split in the same manner, respectively.

Concerning subgoods, Eurostat (2024) data were employed for waste input, recovery from IBA and recycling, if available. TCs for MRFs for separately collected waste were calculated from the Austrian Federal Waste Management Plan (BMK, 2023). Merstallinger and Fritz (2022) delivered data on metal, glass, paper and plastic PW contents for commercial, bulky and construction waste, whereas net factors were extracted from Beigl (2020). The metal PW contents of metal outputs from mixed waste MRFs/MBTs were determined by sampling campaigns in two Austrian plants. Thus, TCs for metals in mixed waste MRFs were also assessed. The sampling was conducted using a one-dimensional approach following Gy's theory of sampling (Gy, 1992); for in-detail methodology and results, see Supplementary Material 1, Chapter S2.1.3.2.3. Considering all other subgoods and outputs, information for mixed waste MRFs/MBTs was obtained from Blasenbauer et al. (2024), with all data normalised to the input of mixed MSW. Literature values (Biganzoli et al., 2014; Hu et al., 2011) were used for Al oxidation rates, and the mass flow of goods was determined using a conversion factor of 1.89 to the subgood flow (Lederer and Schuch, 2024). TCs for IBA sorting were taken from Gritsch and Lederer (2023) and Mühl et al. (2024). All input data are listed in Chapter S2.1.3 in the Supplementary Material 1.

2.1.4. MFA-based indicators

For all subgoods, the separate collection rates - also known as capture rates (Tallentire and Steubing, 2020) - recycling rates, incineration rates and landfilling rates were calculated based on the MFA model (see Table 1). Due to different collection schemes, two streams regarding metal PW must be considered for the separate collection rate. In the case of plastic, glass and paper PW, only one flow needs to be counted. The respective recycling flows were considered for the recycling rate.

2.2. Statistical entropy analysis

Following Shannon's statistical entropy function (Shannon, 1948) from information theory, Rechberger and Brunner (2002) first applied the concept to MFA processes with sets of mass flows and concentrations of substances. In doing so, each process either concentrates, dilutes or maintains the flow of substances; the statistical entropy (H_i) regarding a flow (i) is defined by this substance alteration (cf. Rechberger and Brunner (2002) and Rechberger and Graedel (2002)):

$$H_i = \dot{m}_i^{norm} \cdot c_{i,k} \cdot ld(c_{i,k}) \tag{8}$$

Thereby, the extent of the change in substance flows is dependent on the concentration ($c_{i,k}$) of the respective substance (k) and the normalised mass flow of goods ($\dot{m}_i^{norm}$). Mass flows of goods ($\dot{m}_i$) are normalised by dividing them by the total substance turnover ($\dot{X}_{i,k}$), allowing the application of SEA on systems containing several processes:

$$m_i^{norm} = \frac{\dot{m}_i}{\sum_i \dot{X}_{i,k}} \tag{9}$$

$$\dot{X}_{i,k} = \dot{m}_i \cdot c_{i,k} \tag{10}$$

Referring to the divisor term, $c_{i,k}$ denotes the concentration of k in $\dot{m}_i$. For all flows in the solid aggregate state as well as contained gases or liquids, $c_{i,k}$ alone describes the dilution or concentration of substances. Considering gaseous or aqueous emissions into the environment, $c_{i,k}$ is dependent on the media's background concentrations. Regarding Austria's MSW management, mainly gaseous emissions occur; aqueous emissions are neglectable for the scope of the investigation. Contrary to the original definition of $c_{i,k}$ (Rechberger and Brunner, 2002), this study uses the approach of Laner et al. (2017) in the case of emissions to the atmosphere, which posits the assumption that the system's emissions do not increase the background concentration by 1 % but rather that the substances are diluted to the background concentration. In the case of PW management in Austria, the primary emissions consist of gaseous CO_2, which are minor compared to the background CO_2 concentration in the atmosphere. Therefore, this approach is considered appropriate for the conditions observed. Consequently, $c_{i,k}$ and m_i^{norm} are described as follows:

$$c_{i,k} = \begin{matrix} c_{i,k} \\ \\ c_{k,\text{atmo},g} \end{matrix} \text{ for} \left\{ \begin{matrix} i = 1 \cdots \eta \\ \\ i = \eta + 1 \cdots \eta_g \end{matrix} \right. \text{ for} \left\{ \begin{matrix} \text{solid} \\ \\ \text{gaseous} \end{matrix} \right\} \text{emissions} \tag{11}$$

$$m_i^{norm} = \begin{matrix} m_i^{norm} \\ \\ \dfrac{\dot{X}_{i,k}}{c_{k,\text{atmo},g} \cdot \sum_i \dot{X}_{i,k}} \end{matrix} \text{ for} \left\{ \begin{matrix} i = 1 \cdots \eta \\ \\ i = \eta + 1 \cdots \eta_g \end{matrix} \right. \text{ for} \left\{ \begin{matrix} \text{solid} \\ \\ \text{gaseous} \end{matrix} \right\} \text{emissions} \tag{12}$$

In these instances, $c_{i,\text{atmo},g}$ denotes the atmospheric background concentration of the respective substance for all gaseous emissions (η_g) conversely to all solid emissions (η). To determine the suitability of SEA for circular economy evaluation of multi-material systems, this study used subgoods instead of substances. Thereby, subgood emissions into the air are observed primarily from paper and plastic PW during MSWI. While other subgoods may also emit particulates or gases, these emissions are deemed negligible. In the context of paper or plastic incineration, the subgoods are diluted to $c_{i,\text{ atmo},g} = 180$ g/t, corresponding to the air CO_2-C content of 425 ppm CO_2 (NOAA, 2024); CO-C amounts are minor. Due to C being the dominant element in both paper (Lohmann and Blosen, 2010) and plastic PW (Roosen et al., 2020), it was chosen as the sole determinant for $c_{i,\text{atmo},g}$.

The relative statistical entropy (RSE) is used to investigate the system's ability to concentrate subgoods, whereas its value is calculated with the help of the maximum statistical entropy (H^{max}):

Table 1
Calculation of different MFA-based indicators applied.

Indicator	Calculation	Equation Number	Reference
Separate Collection Rate	$SCR = \frac{\text{Separately Collected Waste}}{\text{Total Waste Input}}$	(4)	Haupt et al. (2018)
Recycling Rate	$RR = \frac{\text{Waste to Recycling}}{\text{Total Waste Input}}$	(5)	EC (EC, 2019b)
Incineration Rate	$IR = \frac{\text{Waste to MSWI}}{\text{Total Waste Input}}$	(6)	Pomberger et al. (2017)
Landfilling Rate	$LR = \frac{\text{Waste to Landfill}}{\text{Total Waste Input}}$	(7)	EU (EU, 2018b)

$$RSE = \frac{H}{H^{max}} \tag{13}$$

H^{max} is dependent on whether a subgood is emitted to an environmental compartment or contained within the investigated system (Parchomenko et al., 2020): In the emission case H^{max} is calculated via Eq. (14), whereby $c_{i,\text{atmo},g}$ denotes the minimal background concentration of the emitted subgood in the environment. Contrarily, H^{max} in closed systems is configured by a set of flows with identical concentrations, as calculated with Eq. (15) (Parchomenko et al., 2020).

$$H^{max} = ld\left(\frac{1}{c_{i,\ \text{atmo},g}}\right) \tag{14}$$

$$H^{max} = ld\left(\sum_i m_i^{norm}\right) \tag{15}$$

Analogous to Laner et al. (2017), the MFA system is divided into stages, whereby it is construed to represent a chain of processes. The number of stages $(n_s = n_p + 1)$ corresponds to the amount of chain processes (n_p) (Rechberger and Graedel, 2002), ensuring mass conservation for each stage (s). A stage is characterised by flows from the former stage $(s - 1)$ and flows to the next stage $(s + 1)$, that can either be inputs or outputs from/to previous or subsequent processes, recycling flows between two processes from preceding or following stages, inputs in or exports out of the system (Laner et al., 2017). The stage statistical entropy (H_s) for a subgood is determined by the sum of subgood flow entropies within the same stage:

$$H_s = -\sum_i H_i \tag{16}$$

In this study, the stages are defined as products before consumption, waste generation, collection, sorting, incineration, IBA sorting, recycling and landfilling $(s = 0,\ 1\ldots7)$. The step-by-step calculation for each stage is displayed in Supplementary Material 2. Subsequently, the subgood concentration efficiencies (SCEs) can be calculated with Eq. (17) by comparing the overall differences between stages; hence, the subgood concentration efficiency denotes how much of a subgood is transferred into a single stream (Rechberger and Brunner, 2002). It was determined from the waste generation stage $(s = 1)$ to assess the PW system's concentration ability, neglecting the product phase.

$$SCE = \frac{\Delta H}{H} \tag{17}$$

3. Results and discussion

3.1. Material flows of PW in Austria 2020

The material movement and fate of PW in Austria 2020 could be determined with the MFA, and in Fig. 2, the material flows of all PW materials are depicted. To generate comparability, flow values are divided by the total input of PW, respectively. In the following, results for each PW material are discussed separately, whereas Fig. 3 summarises the MFA-based circular economy indicators. All values are characterised as mean values ± standard deviation of a normal distribution. Results for the MFA on goods level can be found in the Supplementary Material 1, Chapter S.3.1.ff.

3.1.1. Aluminium

The material flows of Al PW in 2020 show that Austria is on track with the proposed EU recycling targets for 2025 and reveal that further investigations are necessary to accurately calculate the quotas for 2030. With a production of ~24,000 t/yr, ~8,900±200 t/yr Al PW are collected separately; the corresponding mass in household mixed MSW equals ~13,900±200 t/yr. A total of ~14,700±1,950 t/yr Al PW are recycled, of which ~7,950 t/yr stem from separate collection, ~2,450 t/yr from mixed waste MRFs/MBTs and ~4,300±1,950 t/yr from IBA. This amounts to a recycling rate for Al PW of 61±8 %. Here, around 33 % originate from separate collection, 10 % from mixed waste MRFs and 18±8 % from IBA, underscoring the importance of adopting a multifaceted approach to address recovery and recycling within a waste management system. The high uncertainty can be attributed to the Al oxidation rates in MSWI, which vary considerably in the literature (Biganzoli et al., 2012, 2014; Biganzoli and Grosso, 2013; Gökelma et al., 2021; Hu et al., 2011). Such Al losses during incineration have been implemented by Lederer and Schuch (2024) with 5–15 % and Thoden van Velzen et al. (2020) with 10–20 %, although this study has utilised values of 36.5 ± 26 % (Biganzoli et al., 2014; Hu et al., 2011). Visibly, further investigations into oxidation rates are necessary, which

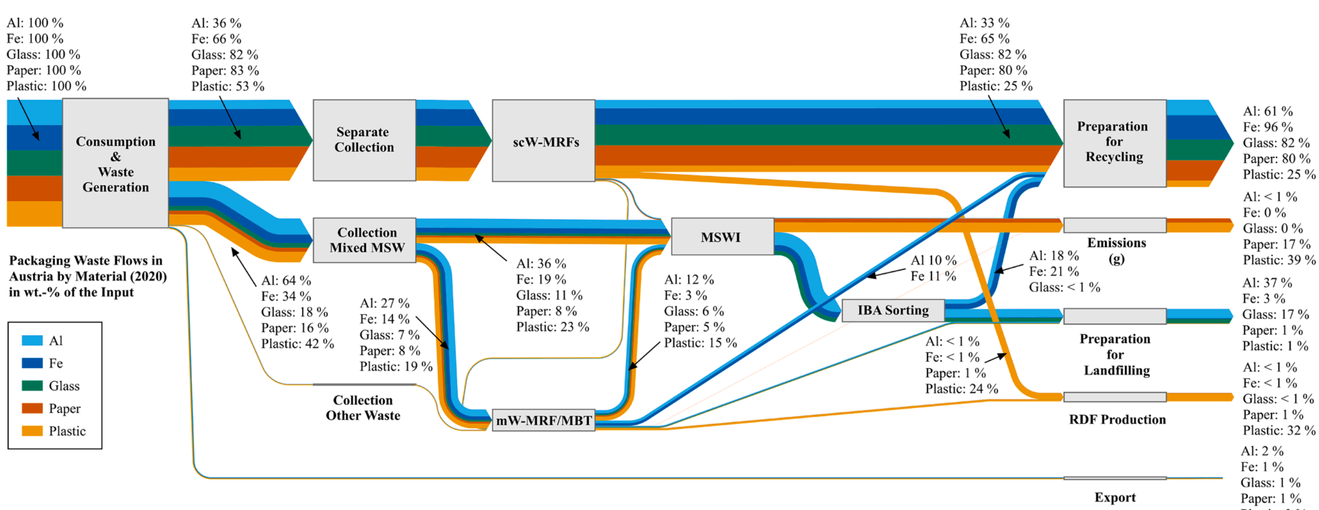

Fig. 2. Packaging waste (PW) material flows in the Austrian municipal solid waste (MSW) management system in 2020. Mean values are depicted in wt.-%, rounded to whole numbers. Where feasible, flow values have been positioned adjacent to the flows. The remaining flow values are detailed in the Supplementary Material 1, Figure S-11. The depicted flows were divided by the respective mass put on the market for each PW material (Al, Fe, Glass, Paper, Plastic) to generate comparability between the materials. Thereby, 100 % of each packaging material enters the system and is divided into different treatment pathways. This 100 % equals around 24,000 t for Al, 41,000 t for Fe, 311,500 t for Glass, 615,400 t for Paper and 299,000 t for Plastic PW. Abbreviations: incineration bottom ash (IBA), mechanical biological treatment (MBT), municipal solid waste incineration (MSWI), material recovery facility for mixed waste (mW-MRF), refuse derived fuel (RDF), material recovery facility for separately collected waste (scW-MRF).

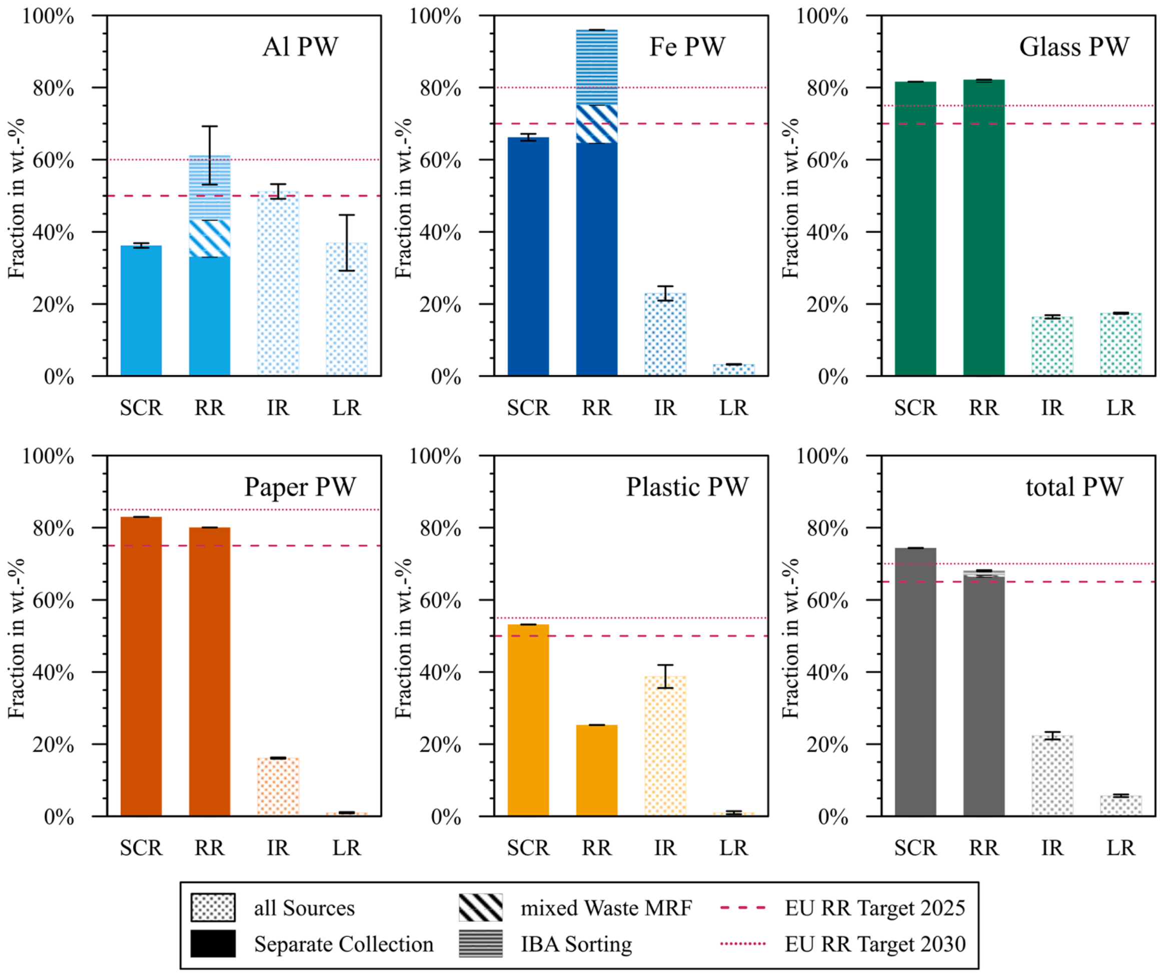

Fig. 3. Circular economy indicators (mean values) - separate collection rate (SCR), recycling rate (RR), incineration rate (IR) and landfilling rate (LR) - for Al, Fe, Glass, Paper, Plastic and all (total) packaging waste (PW) in Austria for 2020. Indicators were split according to the contribution of waste management processes as a fraction of PW input per material or of all PW mass: SCRs solely consider separately collected waste (only fully filled bars), in the RRs further mixed waste material recovery facilities (MRFs, hatched bars) and incineration bottom ash (IBA) sorting (lined bars) are included. For the IR and LR, limited distinctions of the wastes' origin can be made; therefore, all sources are considered (dotted bars). Error bars indicate standard deviations. EU targets for each material (dashed lines) only relate to RRs.

are decisive in determining whether the 2030 recycling target is achieved (see Fig. 3). Regarding the landfilling rate, an amount of ~8,900 ±1,900 t/yr Al PW, or 37±8 %, is landfilled, half of which is oxidised.

To support Al PW recovery and recycling, and thereby keep Al in the loop, improved separate collection and sorting of mixed MSW should be facilitated, as the demand and greenhouse gas emission savings for low-quality Al scrap are limited (Allegrini et al., 2015), as well as the recovery efficiencies for Al from IBA. Here, the Austrian deposit-return system for single-use beverage cans commencing 2025 will increase separate collection. For all Al PW not separately collected, the recovery with other non-ferrous metals in mixed waste MRFs is already prevalent, although sorting efficiencies are improvable. Currently, the mean TC for net Al recovery from mixed wastes equals 0.41±0.01, with best-performing mixed waste MRFs around 0.46. These values are notably higher than the 0.295±0.155 used by Lederer and Schuch (2024), attributable to the data update. Nevertheless, even Austria's best practice examples exhibit lower TCs than some reported by other authors (Montejo et al., 2013; (Thanos) Bourtsalas and Themelis, 2022), although it is unclear if those TCs are based on net or gross levels. This can be ascribed to high throughputs in mixed waste MRFs with heavy belt loads and the fact that not all mixed waste MRFs in Austria are equipped with eddy current separators. These aspects indicate opportunities for technical advancements to further enable a circular economy for Al PW.

3.1.2. *Ferrous metals*

Regarding Fe PW 2020, ~39,300 t/yr were recycled, corresponding to a recycling rate of 96 %, revealing suitable waste management. Thereby, ~26,400 t/yr originate from separate collection, ~4,300 t/yr from mixed waste and ~8,500 t/yr from IBA. Up to a quarter of Fe PW is incinerated, only 3 % is landfilled. Fe oxidation losses were not included in the calculations, albeit reported in low amounts (López-Delgado et al., 2003; Tayibi et al., 2007). However, oxidation during MSWI might notably affect Fe recycling and should be considered in further research.

The net TCs for Fe separation from mixed MSW equal 0.80±0.10 for Austria, with the most efficient plants achieving up to 0.93. Here, an increase in TCs compared to Lederer and Schuch (2024) is shown; the TCs for Fe align with those documented by (Thanos) Bourtsalas and Themelis (2022). Due to the superior separation efficiencies of Fe, replacing Al packaging materials with Fe might be beneficial from a waste management perspective; further research is necessary to assess the higher weight and connected effect on transportation, energy, food safety, and life cycle impacts (Fellner et al., 2018; Geueke et al., 2018; Passarini et al., 2018; Turner et al., 2015).

3.1.3. Glass

For glass PW, 82 % of the total ~311,500 t/yr were collected separately, nearly all recyclable. This surpasses the best practice of Tallentire and Steubing (2020) and shows Austria's well-performing PW management system for glass. To increase the circularity of glass PW, measures promoting separate collection and reuse should be preferred, such as the recently agreed-upon proposal of the EU Packaging Directive with a 10 % reusable quota for beverage packaging (EC, 2022). In Austria, initiatives include introducing 0.33 l reusable beer bottles (Vetropack, 2024) in addition to 0.5 l, and developing a reusable system for wine bottles (Österreichisches Ökologie-Institut, 2023).

An alternative approach could involve the recovery of glass originating from mixed wastes. In Austria, mixed MSW from households typically contains ~4 % glass PW, as shown in this study as well as in sorting analyses (Beigl, 2020). Glass within this mixed MSW is immediately crushed by the shredder upon entering a mixed waste MRF or MBT, owing to its brittle nature, and it subsequently accumulates in the sieve underflow. The recovery of such mixed waste origin glass is practised in Spain, Cyprus and Greece, with operating TCs ranging from 0.03 to 0.49 (Cimpan et al., 2015; Montejo et al., 2013; (Thanos) Bourtsalas and Themelis, 2022). It should be noted, however, that the processing required to recover glass from this material stream is assumed to be energy- and cost-intensive.

Additionally, the expansion of glass recovery from IBA appears promising (Bruno et al., 2021; Mühl et al., 2023). However, in 2020, only one Austrian IBA sorting plant conducted glass separation from fluidised bed IBA with ~1700 t/yr glass PW recovered, mirrored in a 16 % incineration rate and 17 % landfilling rate of glass PW. In recent years, glass separation from fluidised bed IBA has been implemented in multiple sorting plants, and a rise in such recovered glass fractions can be observed. In Austria, 30 % of the total waste mass incinerated is directed to fluidised bed incineration plants, resulting in a 17 % share of fluidised bed IBA in total IBA. Even if all glass from the fluidised bed IBA is diverted, the glass in the IBA from grate incineration will still be landfilled, as its recovery is not feasible (Bayuseno and Schmahl, 2010; Blasenbauer et al., 2023; Mühl et al., 2023). Therefore, implementing separation pre-incineration should be considered. Currently, issues towards closed-loop recycling of IBA-glass persist with extraneous materials, impurities, and contaminants like heavy metals (Mühl et al., 2023). Post-sorting may offer improvements, but further research is required. Open-loop processes are available options but lack uniform contaminant limit values and consistent quality standards.

3.1.4. Paper

In 2020, ~510,500 t/yr were separately collected, corresponding to a separate collection rate of 83 %. In Austria, only paper PW from separate collection is recycled; paper in mixed wastes is almost exclusively incinerated (incineration rate = 16 %) or directed to RDFs. After the sorting of separately collected paper, this amounts to a recycling rate of 80 %, sufficient for the EU 2025 target but beneath the 2030 target. A future increase in paper PW put on the market due to the substitution of plastics and a rise in e-commerce is likely (Cayé et al., 2023; Escursell et al., 2021). To comply with EU targets and increase the circularity of paper PW, the separate collection should be improved. However, challenges associated with the established collection and recycling processes of composite and multi-material paper PW prevail and are expected to intensify in the future due to this substitution effect (Schmidt and Laner, 2021; ZSVR and UBA, 2023).

Beyond this, paper recycling from alternative sources such as lightweight PW or mixed MSW can be explored, which is already practised in Southern Europe (Cimpan et al., 2015; (Thanos) Bourtsalas and Themelis, 2022) and currently investigated in Germany (Spies et al., 2024). Hereby, the main challenges represent moisture and dirt contents (Miranda et al., 2013) and recognising that recycling may not always be favourable due to the incorporation of contaminants is crucial. This aspect is critical when recycling rates surpass 50 % and cascading recycling with potential contaminant accumulation occurs (Pivnenko et al., 2016). Moreover, the inevitable degradation of fibre quality with successive recycling limits material use cycles, underscoring the importance of prioritising waste reduction and reuse. Besides, further research into contaminants and the feasibility of producing paper from mixed wastes remains essential.

3.1.5. Plastic

Austria is not an exception within the EU, as challenges with plastics recycling are also present in the country: Of ~299,000 t/yr plastic PW put on the market, 53 % are separately collected. After sorting in single-stream/commingled MRFs, a recycling rate of 25 % can be achieved, which must still double to reach the EU 2025 target. Due to the high incineration rate of 52±5 %, a vast recycling potential is neglected. Furthermore, an additional ~97,000±9,500 t/yr plastic PW is utilised as RDF. Minor amounts (~3000±1,300 t/yr) are landfilled after MBT. Compared with the findings of Van Eygen et al. (2018), the situation has remained relatively unchanged since 2013. The incineration, RDF and landfilling quantities are almost identical to 2020, with 120,000 t/yr, 96,000 t/yr and 3,400 t/yr for 2013 (Van Eygen et al., 2018). Similarly, recycling rates and the amount of packaging introduced to the market have not experienced considerable shifts, attributable to the fact that between 2013 and 2020, no legislative changes or alterations in the collection and recovery systems occurred (Picuno et al., 2021a). With the 2021 amendment to the Packaging Ordinance (BGBl. II Nr. 597/2021), measures were introduced for the first time, standardising and expanding the collection system for plastic PW. Also, it mandated a minimum recycled content of 25 % in PET beverage bottles from 2025, and stipulated that only recyclable or reusable plastic packaging may be placed on the market commencing 2030. Subsequently, the 2023 amendment (BGBl. II Nr. 284/2023) introduced the establishment of a deposit-return system for single-use beverage bottles starting in 2025.

Despite these efforts, plastics will continue to pose the biggest challenge for recycling within the EU (Dahlbo et al., 2018; Roosen et al., 2022; Thomassen et al., 2022); design guidelines (Gritsch et al., 2024) and other recycling pathways, such as chemical recycling (Ragaert et al., 2017) or recovery from mixed MSW, must be considered (Cimpan et al., 2015; Esguerra et al., 2024; Picuno et al., 2021a). The latter is already in place in some EU countries, and in Austria, there is a theoretical potential of ~14,000 to 53,000 t/yr (Blasenbauer et al., 2024). It has already been shown, albeit with more effort and lower yields, that similar recyclate qualities can be generated from mixed wastes (Luijsterburg and Goossens, 2014; Thoden van Velzen et al., 2021) compared to separate collection. Nevertheless, research into the effects on RDFs and calorific values for incineration must be conducted. Moreover, investigations considering contamination, greenhouse gas emissions, and energy balances are still pending.

3.2. Statistical entropy analysis

The course of the relative statistical entropies and subgood concentration efficiencies for all materials through the individual stages are shown in Fig. 4. Stage 0 represents the product phase, wherein all packaging is consolidated into single product flows, characterised by an entropy minimum. Hereby, to generate comparability to recycling rates, the statistical entropy difference within packaging materials is not accounted for. Since the efficiency of the waste management system is to be assessed, the products entering the system are defined by zero entropy.

3.2.1. Non-combustibles: aluminium, ferrous metals and glass

Overall, the SEA results indicate that Austria's PW management system performs effectively for Fe, Al, and glass. Through efficient collection and sorting processes, a high concentration of these materials within recycling streams can be achieved. After consumption and discarding, the maximum entropy is reached at the waste generation stage

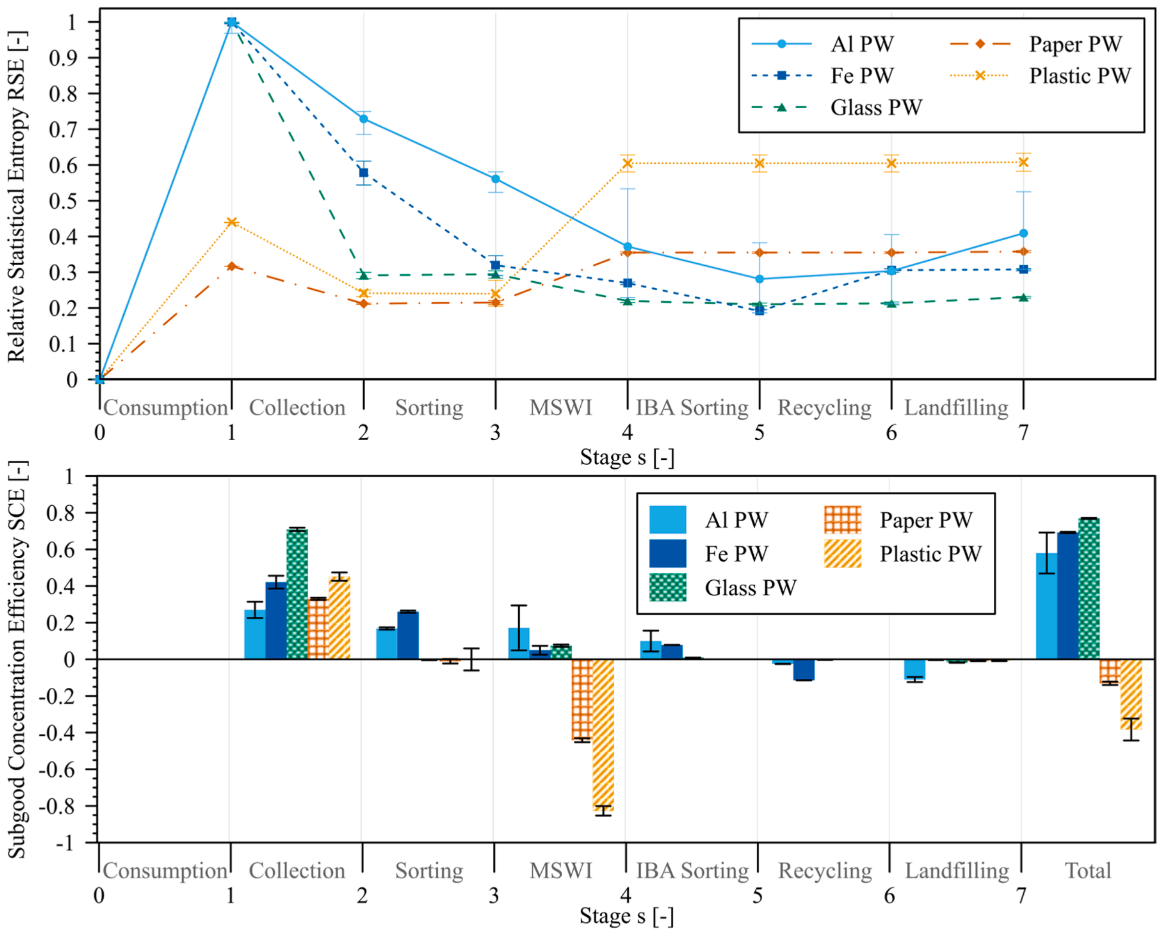

Fig. 4. Evolution of the mean relative statistical entropies (RSEs) and mean subgood concentration efficiencies (SCEs) for Al, Fe, Glass, Paper and Plastic packaging waste (PW) within the stages of the Austrian municipal solid waste (MSW) management system in 2020: consumption, collection, sorting, municipal solid waste incineration (MSWI), incineration bottom ash (IBA) sorting, and landfilling. The error bars show standard deviations.

for non-combustible PWs. This has already been reported by Rechberger (2001) for cadmium in an MSW system: the exact location of the subgoods within MSW and information about concentrations are unknown, and only one mean concentration over all waste fractions is available, resulting in a H^{max} at stage 1. Higher separate collection rates correspond to greater observed decreases in relative statistical entropies, associated with increased subgood concentration efficiencies (cf. Fig. 3 and 4).

Furthermore, for these PWs, the MSWI (stage 4) represents an entropy-decreasing process, as the subgood fraction in exhaust gasses and fly ash is minor. So, most metal and glass are concentrated in the IBA, which aligns with earlier studies (Rechberger, 2001). Oxidation levels of Al during incineration vastly influence the relative statistical entropy of Al, evidenced by the large error bars. This effect can be substantial enough to nearly nullify concentration effects. Hence again, further research is necessary to better understand the behaviour of Al during MSWI, as current findings show considerable variability (Biganzoli et al., 2012,2014; Biganzoli and Grosso, 2013; Gökelma et al., 2021; Hu et al., 2011). Nevertheless, it can be concluded that the maximum concentration of Al PW can be achieved through different waste treatment methods depending on the oxidation level. Separation prior to incineration is more effective at higher oxidation levels, whereas cleaner fractions can be recovered through recovery from IBA at lower oxidation levels. Analogously, Al oxidation levels severely influence entropy levels during landfilling (stage 7), with higher oxidation increasing entropy.

Stage 6, recycling, leads to a positive change in the relative statistical entropy for metals. This increase is solely based on the theoretical dilution with non-packaging metals. It is the same material in practice; therefore, no actual dilution occurs. When applying SEA to subgoods, defining the objects under investigation is paramount, and results vary depending on the demarcation level (e.g. plastics and metal or polyethylene and Al) (Skelton et al., 2022). PW is delineated not by physical-material properties but by legal criteria, which leads to the observed dilution effect during the recycling of Al and Fe. This indicates that with regard to further concentration efforts, closed-loop recycling of metal PW is preferable. Regarding glass PW, no prominent H-change is observed during recycling, owing to the low non-packaging glass content in mixed MSW (BMK, 2023; Beigl, 2020), and any such glass or glass from bulky waste is typically recovered during IBA sorting, wherein glass extraction is less common in Austria.

3.2.2. Combustibles: paper and plastic

Following their minimum relative statistical entropy within the waste sorting stage, the investigated combustibles, paper and plastic PW, reach their H^{max} after incineration, caused by maximum dissipation into the atmosphere. Aligning results with Kaufman et al. (2008) are observable, who investigated the fate of carbon during MWSI. Given that paper and plastic primarily consist of carbon, similar behaviour is anticipated. Because the incineration entropy is dominant, no further relative statistical entropy change can be identified in the later stages 5 to 7, although other authors have described a considerable H decrease through sorting (Skelton et al., 2022), if subsequent incineration is not accounted for.

As this study represents the first application of SEA on PW involving multi-material systems, focusing on subgoods rather than substances, the issue of H^{max} concerning combustibles and the background concentration $c_{k,\mathrm{atmo},g}$ must be addressed. Notably, problems were identified with materials like paper, which, despite having high recycling rates, still exhibit negative subgood concentration efficiencies due to incineration, as seen in Fig. 4. Adhering strictly to the SEA maxime of continuously favouring material concentration could lead to the erroneous assumption that landfilling non-recyclable paper or plastic PW should be preferred over waste-to-energy processes or the usage as RDF. Nonetheless, maximizing recycling of combustible PW substantially reduces the material losses through dissipation during incineration. However, the applicability of SEA to combustible materials warrants further scrutiny in this context. Since no other studies are available for comparison, this research serves as a foundational basis, highlighting the need for further investigation. The SEA method must be refined for broader applications, raising the question of whether concentration is always desirable in such contexts.

3.2.3. *Benefits and limitations*

SEA offers valuable insights into waste management and recycling practices, surpassing the knowledge provided by recycling rates alone. It enables tailored management decisions, such as addressing the optimal point for aluminium recovery in relation to the impact of oxidation levels in MSWI. Additionally, SEA also reveals information about preferred recycling pathways. It helps to evaluate the merits of different recycling approaches: concerning PW, closed-loop recycling is not only commonly preferable from a resource management perspective but also legally mandatory in some cases due to health implications regarding, e. g. food-grade material. This study demonstrates that higher recycling rates typically correspond to lower statistical entropies for non-combustible materials.

However, SEA has limitations. The method struggles to effectively address the waste management of combustible materials, whereby dissipation in MSWI dominates the statistical entropy. This limits its applicability in assessing the full lifecycle of these materials. Furthermore, while SEA emphasizes material concentration, it may inadvertently promote mono-landfilling as an optimal strategy, overlooking broader benefits such as energy recovery and neglecting the negative impacts associated with landfilling. SEA enfolds its strength in comparing specific scenarios, making its results highly context-dependent. The results do not have standalone meanings and require careful interpretation. The method's utility relies on addressing underlying questions about which materials to concentrate, why these materials should be concentrated, why concentration is the preferred approach at all, and where materials should be concentrated. These questions demand thorough considerations; connected assumptions and implications must be clearly communicated and argued, which may restrict the broader use of SEA as well as the final dissemination of the results.

4. Conclusion

The transition towards a circular economy needs assessment tools and the establishment of a data basis to accurately measure progress and identify areas for improvement. Current challenges include false reporting, data gaps, and data quality. Existing indicators measuring the circular economy within the EU member states, especially recycling rates, primarily focus on material efficiency, neglecting the broader systematic aspects of resource effectiveness (Schmidt and Laner, 2023). This study utilised MFA to evaluate the circular economy of PW in Austria for 2020. The resulting model lays a robust foundation for future comparisons and investigations, emphasising the importance of diverse waste treatment options for advancing circularity and showing potential for more efficient material recovery. The findings reveal that Austria can reach all the 2025 EU PW recycling targets for Al, Fe, glass and paper, with plastics remaining a considerable challenge. Additionally, the study marks a successful initial application of SEA to a PW system at the national level, highlighting the complexities of comparing combustibles with non-combustibles and indicating the need for further methodological refinement.

Yet, the MFA model exhibits several limitations, notably its technology-focused approach, which stresses recycling as a key process for the circular economy of PW while neglecting other crucial factors such as reduction, reuse, and design for recycling, albeit approaches to implement them have been explored in the literature (Laner et al., 2017). Future research should address the quantification challenges and data gaps associated with these factors, including TCs of mixed waste MRFs and compositions of incinerated waste. Feasible steps towards an improved data basis include regular sampling and sorting campaigns with revised sorting catalogues (such as distinguishing Fe and non-ferrous metals). If municipalities and federal states plan to conduct such waste analyses, involving the scientific community in consultation is advisable. Moreover, balancing MRFs appears to be a viable approach to gain further insights into the state of the art in separation of metals as well as other recycleables from mixed wastes.

Separate collection alone appears insufficient to achieve the required quotas, particularly for plastic PW. Whilst the recovery of recyclables from mixed MSW might provide additional quantities of PW to reach the EU targets and generally increase the circularity of PW, alternative recycling pathways, especially for paper and plastics, present unresolved questions regarding contaminants, extraneous material, life cycle assessments and economic implications.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used Grammarly, DeepL and ChatGPT in order to carry out spellchecks, conduct translations and improve readability. After using these tools, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

CRediT authorship contribution statement

Anna-Maria Lipp: Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jakob Lederer:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

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.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2025.108193.

Data availability

Data will be made available on request.

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