



The environmental benefits of improving packaging waste collection in Europe

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ABSTRACT

Collecting more waste via source separation waste collection systems is an essential part of increasing resource efficiency, achieving European recycling targets and closing the loop in a circular economy. Huge variation in the capture rates of packaging waste (paper, plastic, metal, composite material and glass) exists in Europe, even between municipalities with similar characteristics, which suggests there is great potential to increase the amount of these materials that can be recovered. In order to assess the environmental impacts linked to higher collection rates, a Life Cycle Assessment model was built that considers the reduced need for virgin materials as the system's loops are closed. An extra 18 million tonnes of waste could be collected annually in Europe if best practice collection strategies were to be deployed, leading to a 13% reduction in greenhouse gas production associated with the packaging and packaging waste. Although high collection performance is crucial for efficient resource use, improving source separation waste collection systems alone will not be enough to achieve recycling targets; material losses must be reduced throughout the value chain, i.e. at the sorting and recycling stages. By evaluating the circularity and environmental implications of current waste management, it can be shown at which points in the system the most improvement needs to be made for each material in order to facilitate the transition towards a circular economy.

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1. Introduction

The transition towards a circular economy, with its aim of extending the useful lifetime of materials by promoting recycling whilst lowering resource use, has become a priority in the European Union's (EU) vision of sustainable economic growth and global competitiveness (Milios, 2018; Tisserant et al., 2017). As such, the EU intends to increase the amount of packaging and packaging waste (PPW) that is recovered and recycled, 73 million tonnes of which is currently generated in Europe annually (Eurostat, 2016). This includes both packaging and non-packaging paper, cardboard (henceforth also categorised as "paper"), plastic, metal, composite material and glass. Thus, common recycling targets for PPW have been imposed which must be achieved by 2030 (European Parliament and Council, 2018), these are recycling rates of: 85% for paper, 55% for plastic, 60% for aluminium, 80% for ferrous metal and 75% for glass.

Improving the performance of waste collection systems (WCS), thus diverting more recyclable material towards the appropriate material recovery facility (MRF) and away from sending it for

disposal is an important sustainability issue and the obvious first step towards achieving the recycling targets (European Parliament and Council, 2018). In recent years, member states have made major investments in their WCS in response to the EU's circular economy ambitions, such as in collection schemes, sorting and reprocessing equipment and infrastructure (Hahladakis et al., 2018). However, various operational strategies have been developed, even between municipalities within the same member state, due in part to different morphologies and available resources. This has led to much variation in PPW material capture rates (i.e. the proportion of material collected separately from that which is included in the residual waste) among WCS; this suggests that there is much room for improvement in the management of waste in Europe.

Within the COLLECTORS project, data on source separation WCS throughout Europe have been gathered (COLLECTORS, 2019), i.e. WCS whereby PPW is collected separately from residual waste. Different collection methods for PPW materials are used, such as: single source separation, in which specific PPW materials are separated from each other before collection, and; various commingling strategies, in which different PPW materials are collected together. Commingling methods were introduced with the intention of improving capture rates, whilst at the same time reducing collection costs (Miranda et al., 2013). However, employing

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commingling methods may compromise recovery efficiencies and material quality downstream (Eriksen et al., 2019), e.g. due to higher levels of contamination. In addition to capture rates, an increased closing of material loops depends on the quality of the waste materials after collection and sorting.

The aim of this study was to identify the best performing waste collection methods for PPW in terms of capture rates, and to quantify the resource recovery potential and associated environmental benefits if these best practice methods were to be applied throughout Europe. We therefore developed a Life Cycle Assessment (LCA) based model that builds upon the COLLECTORS database and takes factors such as capture rate and material quality into account. Performing an LCA of waste management systems is not a simple task. This is due to the inherent complexity of the system in question; there are a considerable amount of variables and extensive information requirements for such a study (Ferreira et al., 2014). However, it has been shown to be an effective tool to assess the impact of whole waste management systems in a few studies (e.g. Eriksson et al., 2005; Giugliano et al., 2011). Here, it has been applied to a novel material flow model to analyse the potential of improved source separation WCS for closing the loops in a circular economy and reducing the environmental impacts associated with PPW. This, for the first time, has been based on the performance of best practice WCS in Europe.

This paper follows the fundamental stages of an LCA according to ISO 14,040 (ISO, 2009) and is thus organised as follows: the goal and scope are defined from the outset (Section 2.1) and the material flow model has been described (Section 2.2). These steps are followed by an inventory analysis (Section 2.3). The impact assessment methodology is then described and the scenarios which have been assessed are defined. The European capture rate data which informed the scenarios, the potential material resource recovery that results and the associated environmental impacts are presented in Section 3. Finally, a sensitivity analysis has been presented.

2. Methodology

2.1. Goal and scope

The goal of this study was to determine the potential improvements that can be made to resource recovery and the reductions in environmental impacts that could be achieved via improving source separation WCS. To do this, three scenarios were considered: “current collection”, “best practice” and “2030 recycling targets”, explained in detail in Section 2.5. Decisions at the collection stage do not only affect the management of waste, they affect the entire life cycle of paper and packaging material (PPM), as increased collection and recycling should lead to an increased substitution of virgin materials in the production of these materials (Fig. 1). Thus, a material flow analysis and LCA of the entire system were performed based on these scenarios.

The scope of the LCA was to assess the environmental implications of increasing the capture rate (Eq. S1, Supplementary material) of each PPW material in Europe, i.e. paper (non-packaging and packaging), plastic, metal, composite material and glass. The environmental impacts associated with the production of the PPM, the collection and sorting of the waste generated, and the fate of the materials were all considered within the model boundary. The PPM were produced from both virgin and recycled material, the proportions of which were dependent upon the capture losses of each material at the collection stage, the material losses at the sorting and the recycling stages, and the demand for each recycled material. The material flows through the system that were considered in this study are shown in Fig. 1; the amount of material that

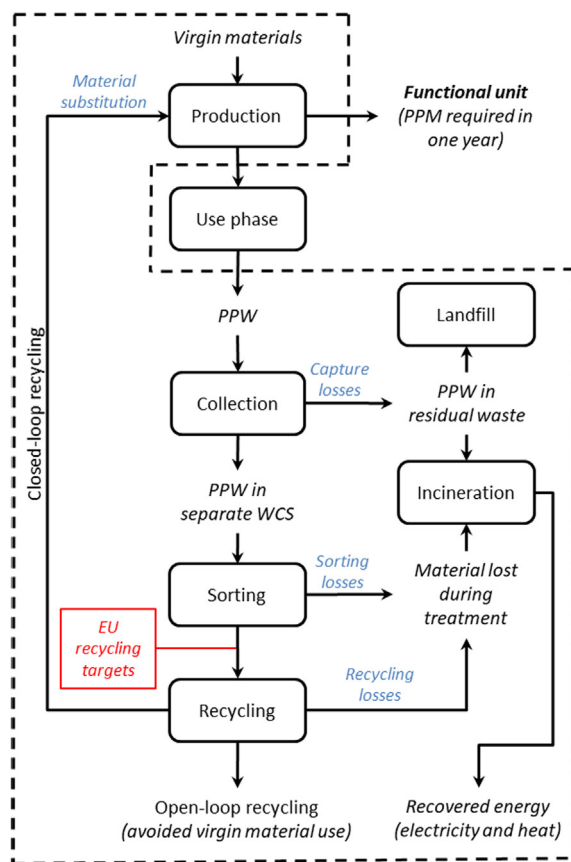


Fig. 1. Flow diagram of the system modelled. Material losses are indicated at various points in the life cycle. The material flow from which recycling rates are calculated in accordance with the EU recycling targets has also been indicated (European Parliament and Council, 2018).

reached each stage of the system in each scenario has been calculated following the general model described in Section 2.2. The use stage was not included within the scope of the analysis, as it can be assumed that it would not be affected by improvements in waste management. The source separation WCS included all collected waste that was separated from residual waste by the inhabitants; no resource recovery of PPW from the residual waste after collection was assumed, thus any material that entered mechanical treatment plants was assumed to be either put in landfill or incinerated. The functional unit of the LCA was the total PPM required in one year in Europe (Eurostat, 2016).

2.2. General model

Within the model presented in this study, PPM was produced from two systems defined by different material flows (F): primary production, which used only virgin materials to produce PPM, and production with both virgin materials and recycled materials, i.e. via closed-loop recycling (Eq. (1)). The PPM produced via closed-loop recycling incorporated recycled material at the substitution rate, i.e. the maximum amount of virgin material that can be replaced by recycled material, discussed later in this section. The total amount of each PPM that was produced was determined based on steady-state analysis, i.e. the production of the PPM was equal to the amount of PPW generated and equal in all three scenarios. The proportion of PPM produced from primary production and with closed-loop recycling was determined based on the material flow through the WCS; this, in turn, determined the total

share of recycled material in PPM and ultimately the environmental impacts associated with production.

$$F_{PPM} = F_{\text{primary production}} + F_{\text{closed-loop recycling}} \quad (1)$$

Following the production and the use phases the PPM becomes PPW and, before PPW can be recycled ($F_{\text{recycled material}}$), material losses (l) occur at various stages of the system. PPW is collected as part of a source separation WCS or in the residual waste. Materials that enter the residual waste are classed as material losses (l_{capture}). PPW materials that enter the source separation WCS are collected via single source separation or via commingling methods. For the purpose of this study, collection methods that incorporated commingling were broken up into two categories which represent the most widespread collection methods: plastic and metal and/or composite material commingling (PMC) and; plastic and metal and/or composite material commingled with paper (PMC + Fibres).

After collection, PPW materials are transported to MRFs before being subjected to two stages of treatment, defined in this study as sorting and recycling. At the sorting stage, material is lost (l_{sorting}) due to sorting inefficiencies and contamination (Table S1, Supplementary material). The level of contamination differs between the collection methods defined above (Eriksen et al., 2019). In addition, losses occur at the recycling stage due to recycling inefficiencies ($l_{\text{recycling}}$). The recycling losses are important in determining how much material is ultimately recycled (Eq. (2)), however they are not considered in the calculation of recycling rates (Eq. S2, Supplementary material) (European Parliament and Council, 2018).

$$F_{\text{recycled material}} = ((1 - l_{\text{capture}}) * (1 - l_{\text{sorting}}) * (1 - l_{\text{recycling}})) * F_{PPW} \quad (2)$$

Recycled materials cannot completely replace the virgin materials due to various factors, such as its quality and economic value. The quality of each recycled material is dependent on the contamination of the waste stream, as well as the inherent deterioration in the properties of the materials undergoing the recycling process (paper fibre shortening, plastic polymer chain scission and cross-linking etc.). The amount of virgin material that can be replaced via closed-loop recycling also depends on the economic competitiveness of the recycled material within the free market (Gala et al., 2015). The proportion of recycled material that can substitute virgin material is defined by the substitution rate ($r_{\text{substitution}}$) of each material. These substitution rates reflect both the technical and economic considerations of each material and have been taken from various literature sources (Section 2.3). Since the demand for PPM was based on steady-state analysis, the maximum amount of recycled material that could substitute virgin materials in the production of a specific PPM ($F_{\text{substitution, max}}$) was determined based on the total PPW generated and the substitution rate (Eq. (3)). Thus, within the model, $F_{\text{substitution, max}}$ equates to 100% production of a PPM with $F_{\text{closed-loop recycling}}$.

$$F_{\text{substitution, max}} = F_{PPW} * r_{\text{substitution}} \quad (3)$$

Within the model, closed-loop recycling of a specific PPW material occurs when that material is of sufficient quality and where there is demand for it in packaging production, i.e. $F_{\text{substitution, max}}$ has not been achieved (Eq. (4)). Where the amount of a recycled material exceeds the demand for it in packaging production, or the recycled material is not of sufficient quality to be used in that market, it is assumed to displace virgin materials for the use in non-packaging applications. This has been referred to here as open-loop recycling (Eq. (5)). The list of potential products that can be produced from each recycled material, and the materials these products would be conventionally made from, is extensive. In the analysis presented here, open-loop recycling was assumed to avoid the production of the same material, of equal quality, from

virgin materials. Hence, the difference between the impacts associated with producing the material from virgin materials and the impacts associated with the recycling processes was accredited to the system. In some cases, entirely different raw materials may be replaced by a recycled material. Thus, the avoided impacts associated with closed-loop recycling must be regarded as only indicative of the potential avoided impacts associated with the collected material (Suter et al., 2017).

$$F_{\text{closed-loop recycling}} = \text{MIN}(F_{\text{substitution, max}}, F_{\text{recycled material}}) \quad (4)$$

$$F_{\text{open-loop recycling}} = \begin{cases} F_{\text{recycled material}} - F_{\text{substitution, max}}, & \text{if } F_{\text{recycled material}} > F_{\text{substitution, max}} \\ 0, & \text{otherwise} \end{cases} \quad (5)$$

The capture losses, which entered the residual waste, were disposed of either in landfill or via incineration. It was assumed that 60% of the residual waste was sent directly to landfill and the remaining 40% was incinerated with energy recovery, based on current European averages (Eurostat, 2019). The entirety of the material that was lost during the sorting and recycling stages was assumed to be sent for incineration with energy recovery.

2.3. Inventory modelling

2.3.1. Paper

Paper represented 49% of the PPW by mass considered in this study (Eurostat, 2016) including the carton board (2.3%) recovered from composite material. This was further divided into newsprint (5.1%) and other non-packaging paper (2.8%), graphic paper (19%), cardboard (15%) and other packaging paper (4.7%) (FAO, 2018; Pivnenko et al., 2016). The process inputs and outputs during the production of each type of paper were based on European averages, as modelled in the ecoinvent database (Wernet et al., 2016) (see supplementary material for a summary of ecoinvent processes used). The primary production of paper included wood handling, mechanical pulping, energy and chemical inputs and internal wastewater treatment. Paper recycling included wood handling for the incorporation of virgin material, mechanical pulping, bleaching and deinking of paper waste, energy and chemical inputs and internal wastewater treatment (Merrild et al., 2008).

The average percentage of unrecoverable fibres due to contamination found at the MRF was set at 1.0% when paper was collected via a single source separation method and 12% when collected via a commingling method (Miranda et al., 2013, 2011). Paper collected by the source separation WCS was assumed to be sorted into four different grades at the sorting stage (Gala et al., 2015). The contribution of each grade to the production of recycled fibres in paper production was based on transfer coefficients for the production processes (Table S2, Supplementary material) reported in the literature (Pivnenko et al., 2016). The recycled fibres were added to the different material types at the substitution rates: 83% for newsprint, 29% for other non-packaging paper, 84% for the packaging paper and cardboard, and 43% for carton board (Gala et al., 2015; Rigamonti et al., 2009; Seigné-Ittoiz et al., 2015). The sorting and recycling losses were assumed to be incinerated with energy recovery.

2.3.2. Plastic

Plastic represented 22% of the PPW by mass (Eurostat, 2016). Plastic packaging can be divided into several polymer types: polyethylene terephthalate (PET); high density polyethylene (HDPE); low density polyethylene (LDPE); polypropylene (PP) and; polystyrene (PS). These polymers constituted 12%, 19%, 28%, 30% and 11% of the total plastic packaging waste mass respectively (Plastics Europe, 2017). Data for the production and recycling of

these polymers were derived from the European plastics industry eco-profiles (Plastics Europe, 2019).

Considerable sorting and recycling losses of each type of polymer were calculated based on the study presented by Eriksen et al. (2018), where the average contamination of the waste stream was dependent on the collection method employed in each case (Tables S1 and S3, Supplementary material). These losses were consistent with other reports of plastic packaging chain analysis (Brouwer et al., 2018; European Plastics Converters, 2016). For each type of polymer, the amount of high, medium and low quality materials able to be produced from mechanical recycling processes were calculated based on the collection method used to collect plastic (Eriksen et al., 2019). As with paper fibres, a reduction in the quality of the plastic polymers occurs during the recycling process (Gala et al., 2015; Hahladakis and Iacovidou, 2018; van der Harst and Potting, 2014). The substitution rates of each type of polymer were estimated based on quality losses of recycled polymers compared to polymers produced via primary production (Van Eygen et al., 2018); for PET, HDPE, LDPE, PP and PS these were 93%, 73%, 61%, 75% and 67% respectively. Sorting and recycling losses were incinerated with energy recovery.

2.3.3. Metal

Metal represented 6% of the PPW by mass, of which 75% was aluminium and 25% was steel (tinplate) (Eurostat, 2016). The production of metal packaging was based on European averages inecoinvent. Metal is smelted then tapped into a holding furnace where it is heated using natural gas and alloying elements are added. It is at this point during closed-loop recycling that recovered metal is also added. Pre-processing yields were considered in the recycling rate of metal (Brimacombe et al., 2005; Løvik and Müller, 2014; Niero and Olsen, 2016). The substitution rate of aluminium and steel packaging was 75% and 50% respectively (Gala et al., 2015). Unlike for paper and plastic, there was no difference in the sorting losses assumed between collection methods (Cimpan et al., 2015).

2.3.4. Composite material

Packaging made out of composite material represented 3% of the PPW by mass (Eurostat, 2016). This material was composed of carton board (75%), discussed in Section 2.3.1, polyethylene (21%) and aluminium (4%). The production of composite material, as well as its other life cycle stages, were thus attributed to paper, plastic and metal in these proportions respectively. After sorting, the pulp was concentrated, and the fibre was used to produce new paper products. The polyethylene and aluminium were expected to be used in the cement industry: for this, the plastic was incinerated with energy recovery, whilst the aluminium was used as a bauxite substitute (Pretz, 2010).

2.3.5. Glass

Glass represented 23% of the PPW by mass (Eurostat, 2016). The primary production of glass included virgin material and energy inputs, water consumption, emissions to air and water and waste production based on European averages, available inecoinvent. Recycled glass packaging included these same processes with the addition of the resource and energy requirements associated with crushing the waste glass and mixing the cullet with raw materials before melting it in a furnace.

Glass retains its colour after recycling; thus, a distinction was made in this study between waste glass of different colours i.e. white, green and brown. The quality of recovered glass and the recycling rate were determined by the colour purity of the sorted cullet and the level of contamination, such as from metal parts, ceramics and organic material e.g. paper (Villanueva and Eder, 2011). The substitution rate was limited by the maximum colour

contamination limits for packaging glass and the market demand for each colour. These were 0.61%, 0.84% and 0.55% for white, green and brown glass respectively. The cullet that was not of suitable quality to produce packaging entered open-loop recycling. All glass losses that occurred after the collection stage were assumed to happen at the MRF, with all sorted material ultimately being recycled (Giugliano et al., 2011).

2.4. Impact assessment

The ReCiPe methodology was applied in the LCA presented in this study (Goedkoop et al., 2008). The inventory results were assigned to 10 different impact categories addressed at the mid-point level, based on the expected types of impact on the environment that result from PPW (Arena et al., 2004; Lopes et al., 2002; Skals et al., 2008). These categories were: the global warming potential (GWP), fossil resource depletion potential (FDP), fresh water eutrophication potential (FEP), marine eutrophication potential (MEP), terrestrial acidification potential (TAP), photochemical oxidant formation potential (POFP), particulate matter formation potential (PMFP), human toxicity potential (HTP), freshwater ecotoxicity potential (FETP) and natural land transformation potential (NLTP).

For each of the environmental impact categories assessed, the total impact value associated with each stage of the system was calculated for each material. The stages were: primary production, collection and sorting, closed-loop recycling, open-loop recycling, disposal and energy recovery. The primary production included the impacts associated with the production of PPM from virgin materials only (Table S4, Supplementary material). The collection and sorting included the impacts associated with the transportation of the waste and sorting it at the MRFs. Closed-loop recycling included the impacts associated with the production of PPM from both virgin materials and recycled materials included at the substitution rate, including the impacts associated with recycling processes (Table S5, Supplementary material). The difference between the impacts associated with producing a material from virgin materials and the impacts associated with producing the same material from recycled materials, avoided when material entered open-loop recycling, was reported separately and subtracted from the total impacts when calculating the net environmental impacts (Table S6, Supplementary material).

Disposal included all the associated impacts of incinerating and landfilling the waste, including the capture, sorting and recycling losses. The electricity and heat production that can be achieved via the incineration process for each PPW material was based on information found within theecoinvent database (Wernet et al., 2016). The energy captured during the incineration process displaced the equivalent electricity (31% renewables, 30% nuclear, 27% coal, 11% natural gas and 1% oil) and heat (natural gas) produced in the EU (Table S7, supplementary material). The environmental impacts of incinerating each material were added to the total impacts of the system, whereas the total impacts associated with the avoided energy production were subtracted from the total impacts when calculating the net environmental impacts.

2.5. Scenarios

Three scenarios were considered in this study: 1) current collection, based on the weighted average of current collection methods; 2) best practice, whereby the current best performing municipalities were used as a case in point for how collection can be improved by following good example, and; 3) 2030 recycling targets, i.e. the capture rates required based on achieving the EU 2030 targets for the recycling rates of each of the PPW materials, using source separation WCS with current sorting and recycling

losses (European Parliament and Council, 2018). The amounts of each PPW material generated were equal in each scenario assuming demand would be unaffected. Therefore, a greater proportion of the PPM could be produced from recycled material if more PPW was collected, sorted and recycled.

The capture rates applied in the model for the current collection scenario were based on the average capture rates reported in the COLLECTORS database, in which information has been compiled on the PPW WCS of 135 municipalities from 24 EU member states (COLLECTORS, 2019). Data collection took the form of consultation with stakeholders and an extensive literature review of national reports and isolated case studies. The characteristics of the municipalities included in this study varied in terms of area size, population density, level of tourism, GDP and total waste generated. The proportions of the different collection methods (single source separation, PMC commingling and PMC + Fibres commingling) employed by Europe as a whole was assumed to match the proportions of these methods employed by the municipalities reported in the COLLECTORS database. Hence, in the current collection scenario, PPW was collected using all three waste collection methods based on these proportions.

No municipality assessed in this study achieved the capture rates for every waste material that would be required to meet the EU 2030 recycling targets. Thus, a posteriori criterion was established to identify the best performing WCS and calculate the capture rates that were applied to the model for the best practice scenario. The data suggests that the capture rate of plastic can be increased by switching to the PMC commingling method, whilst the capture rates of the other waste materials would be increased or insignificantly affected depending on the previous collection method (Section 3.1). Thus, plastic was used to determine the capture rates of the best practice scenario as follows: municipalities with a capture rate for plastic greater than that needed to meet the EU recycling target of 55% were identified ($n = 9$). The average capture rates of the PPW materials in these cases were then calculated. Each of these cases employed the PMC commingling method. Thus, the PMC commingling method was assumed to replace all the other collection methods in the best practice scenario. Single source separation was assumed for both paper and glass.

The capture rates required to achieve the EU 2030 recycling targets were based on the amount of each material that would have to be collected by source separation WCS in order to achieve the 2030 EU recycling targets. The calculation of recycling rates should be based on the weight of packaging waste which enters the recycling operation (European Parliament and Council, 2018), thus the capture rates in this scenario took into consideration the sorting losses. As no EU recycling target has been proposed for composite material, it was assumed to have the same capture rate in the 2030 recycling targets scenario as it does in the best practice scenario. Again, the PMC commingling method was assumed to replace all the other collection methods and single source separation was assumed for both paper and glass.

2.6. Sensitivity analysis

A sensitivity analysis was performed to identify the parameters most influential on the environmental impacts of the WCS. The changes in the net environmental impact values associated with the current collection scenario, when the parameters were adjusted individually, were determined. These parameters were: the capture rate, sorting efficiency and recycling efficiency (i.e. the capture losses, sorting losses and recycling losses were reduced), and; the incineration rate of the material that ends up in the residual waste. The environmental impact implications of switching from the current proportion of different collection methods (as was assumed for the current collection scenario) to only the

PMC commingling collection method (as was assumed for the best practice and 2030 recycling targets scenarios) was also tested.

3. Results and discussion

3.1. Collection methods and capture rates

A range of collection methods for the PPW were represented in the COLLECTORS database (Table S8, Supplementary material). In many cases, materials were collected via single source separation: paper ($n = 100$), plastic ($n = 22$), metal ($n = 29$) and glass ($n = 129$). However, where collected, composite material was always commingled with at least one other material. As discussed in Section 2.2, collection methods that incorporated commingling were divided into two categories: PMC ($n = 86$) and PMC + Fibres ($n = 26$). Glass was commingled with at least one other material in only a small number of cases ($n = 6$), with each of these examples commingling glass with different materials. Thus, there was insufficient information collected to determine the effects of commingling glass with the other materials and these cases were not included within the analysis. There was no significant difference between the capture rate of metal when it was collected via single source separation and when it was commingled with plastic and composite material (Table 1). On the other hand, the capture rate of plastic was significantly greater when commingled with metal and/or composite material, when compared to single source separated plastic. When paper was also commingled (PMC + Fibres) the capture rate of every waste material was reduced significantly.

There was much variation in the PPW capture rates between municipalities with similar characteristics (Tables S9 and S10, Supplementary material). When analysed using a general linear model with country as a fixed factor, only GDP and population density were significant factors for the capture rates of plastic and glass respectively. The length of time a WCS has operated and the waste management policies in place at each municipality were not considered in this study but may be important for determining capture rates. The analysis presented here suggests great opportunity for member states to learn from one another and increase primary resource substitution in Europe.

Commingling collection methods are generally considered to be less costly and more convenient than the single source separation method, which frequently leads to increased capture rates (Cimpan et al., 2015; Miranda et al., 2013). Municipalities that employed the PMC commingling method had a significantly higher capture rate for plastic than those that collect plastic separately (Table 1). However, single source separation methods include PET bottle deposit schemes for instance, which are often operated separately from local waste management. Information on the PPW collected by these systems may be only partially available for some municipalities; the issue of waste data quality and comparability between municipalities is a fundamental challenge when conducting waste management studies (Wuppertal Institute, 2017).

Municipalities which employed PMC + Fibres commingling had lower capture rates for every PPW material. This might simply reflect national collection trends, as a significant country effect was observed; the majority of the municipalities which employed PMC + Fibres commingling within the COLLECTORS database were in the UK and France (Table S8, Supplementary material). A possible explanation for this may lie in the behaviour of citizens; more research is needed in order to understand these social factors, but it is clear that when there is a strong conviction both about the environmental benefits of recycling and the responsibility of cooperating, people are more willing to participate (Vicente and Reis, 2008).

Table 1

The average capture rates (%) of waste materials in Europe. The first row shows average capture rates for each material when collected via single source separation. The second and third rows show the average capture rates of each material when collected in conjunction with specific collection methods. The final three rows show the capture rate of each waste material for each scenario. All European data used to calculate capture rates were extracted from COLLECTORS (2019).

	Paper	Plastic	Metal	Composite material	Glass
Collection methods¹					
Single source separation of each material	60 ^a	24 ^a	46 ^a	n/a	65 ^a
Single source separation of paper, PMC commingling and single source separation of glass	62 ^a	38 ^b	48 ^a	31 ^a	68 ^a
PMC + Fibres commingling and single source separation of glass	50 ^b	17 ^c	22 ^b	12 ^b	56 ^b
Scenarios					
Current collection	58	29	37	22	65
Best practice	75	69	61	48	89
2030 recycling targets	86	76	64 ²	48	94

¹ Like-for-like superscript letters indicate no significant difference in capture rate for each waste material between collection methods ($p < 0.05$).

² Capture rate for aluminium; the capture rate for ferrous metals would have to be 90% to meet recycling targets.

3.2. Potential resource recovery

Based on the scope of the WCS considered in this study, the 2030 recycling targets are ambitious. An extra 18 million tonnes of waste could be collected annually in Europe if best practice collection strategies were deployed based on the best case studies and steady-state analysis (Fig. 2). This is 6.0 million tonnes short of the amount of material that would have to be collected in order to achieve the EU 2030 recycling targets (Figure S1, Supplementary material).

For paper, only the demand for recycled fibres made by carton board (exerted by the production of composite material) was met by current collection. In the best practice and 2030 recycling targets scenarios, the demand for paper fibres in the production of newsprint and cardboard was also met (Table S6, supplementary material). There was assumed to be no demand for medium quality recycled plastic polymers for the production of packaging and thus these polymers entered open-loop recycling (Eriksen et al., 2019). The demand for aluminium was not met in any scenario, thus only aluminium recovered from composite material entered open-loop

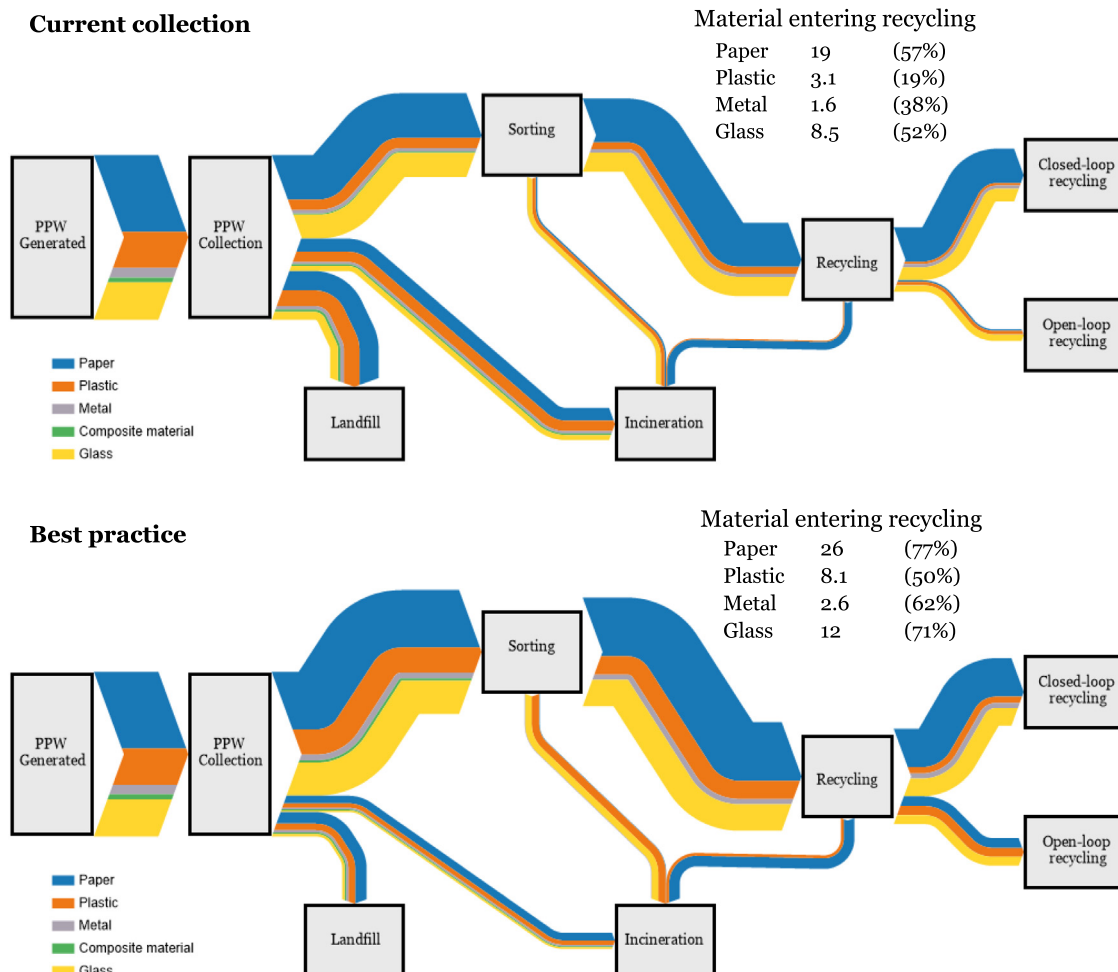


Fig. 2. Sankey diagrams showing material flows in the current collection and best practice scenarios. The amount of each material that enters the recycling stage (million tonnes) and its (recycling rate) is displayed for each scenario. All European data used to calculate capture rates were extracted from COLLECTORS (2019).

recycling. The demand for steel was not met by current collection, but was met by the best practice and 2030 recycling targets scenarios and thus entered open-loop recycling. The demand for each type of glass was not met by any scenario; the collected glass that entered open-loop recycling was assumed to be of insufficient quality for the production of packaging glass, based on average recovery information (Villanueva and Eder, 2011).

Most mechanical biological treatment plants recover metal from residual waste post collection, with others also recovering paper, certain types of plastic and composite material (Cimpan et al., 2015). It has been shown that there is no reduction in the quality of plastic when recovered from the residual waste, although less plastic may be recovered overall when compared to source separation WCS (Jansen et al., 2012; Luijsterburg and Goossens, 2014). As the focus of this study was to assess the potential of source separation WCS in closing the loop in a circular economy, it was beyond the scope to assess the potential of waste recovery from residual waste as either a complementary method to source separation WCS or a substitute for separate collection of certain materials. However, considering the recycling rates in the best practice scenario, a combination of these methods will likely be required to achieve the 2030 recycling targets for all PPW materials. For instance, Brouwer et al. (2018) showed that 22% of generated plastic was recovered ready for recycling in the Netherlands via a PMC commingling method. This is consistent with the recycling rates in Europe using PMC commingling in the model. In addition, Brouwer et al. (2018) showed that 7.7% of the plastic that entered the residual waste could be recovered ready for recycling; thus, in total, the Netherlands had a recycling rate for plastic of 28%. In the best practice scenario, where 69% of the plastic was collected via a PMC commingling method, the plastic recycling rate was 50%; this means a further 5% of the total generated plastic waste would have to be recovered from residual waste post collection to achieve the 2030 recycling targets (European Parliament and Council, 2018).

3.3. Potential environmental implications

The percentage change in each impact category for each PPW material between the current and best practice scenarios have

been presented in Fig. 3. The additional change that would be expected in each impact category should the 2030 recycling targets be achieved has also been shown. Of these impact categories, the impact value of each stage of the system for GWP, FEP, MEP and FETP have been reported here (Fig. 4). The total net reduction in these impact categories in the best practice scenario compared to current collection were 13%, 17%, 15% and 18% respectively. Detailed figures for FDP, TAP, POFP, PMFP, HTP and NLTP have been reported in the supplementary material (Figs. S2–S7). The total reduction in these impact categories in the best practice scenario compared to current collection were 16%, 13%, 16%, 17%, 14% and 30% respectively.

Whilst the percentage change in the impact values between the scenarios were similar for some impact categories, the impacts were often concentrated at different stages of the system or in different materials. The specific PPM that was the largest contributor to environmental impact varied between impact categories. In terms of net impact reduction between current collection and the best practice scenario, plastic showed the greatest potential in the GWP, FDP, MEP and POFP impact categories. The largest potential net impact reductions in NLTP was associated with improved paper collection. Metal showed the most potential reduction in FEP, TAP, PMFP, HTP and FETP, despite the low capture rate assumed for metal relative to paper, plastic and glass in the best practice scenario and the relatively low metal waste generation. This was due to the differences in environmental impacts between closed-loop recycling and primary production of metals (Tables S4–S5, Supplementary material). However, since both the post collection recovery of metal from residual waste and the recovery of metals from incineration bottom ash were not considered in this study, the potential reduction in the impacts between the two scenarios for metal, in particular, may be overestimated.

For plastic, some environmental impacts were greater for production with recycling than for the primary production, resulting in an increase in the FEP and NLTP in the best practice scenario compared to current collection (Fig. 3). Contribution analysis showed that this results from the increased electricity requirements incurred at the resource recovery and recycling stages and is thus dependent upon the electricity market mix assumed by the model (Simões et al., 2011). For the purpose of this study, the

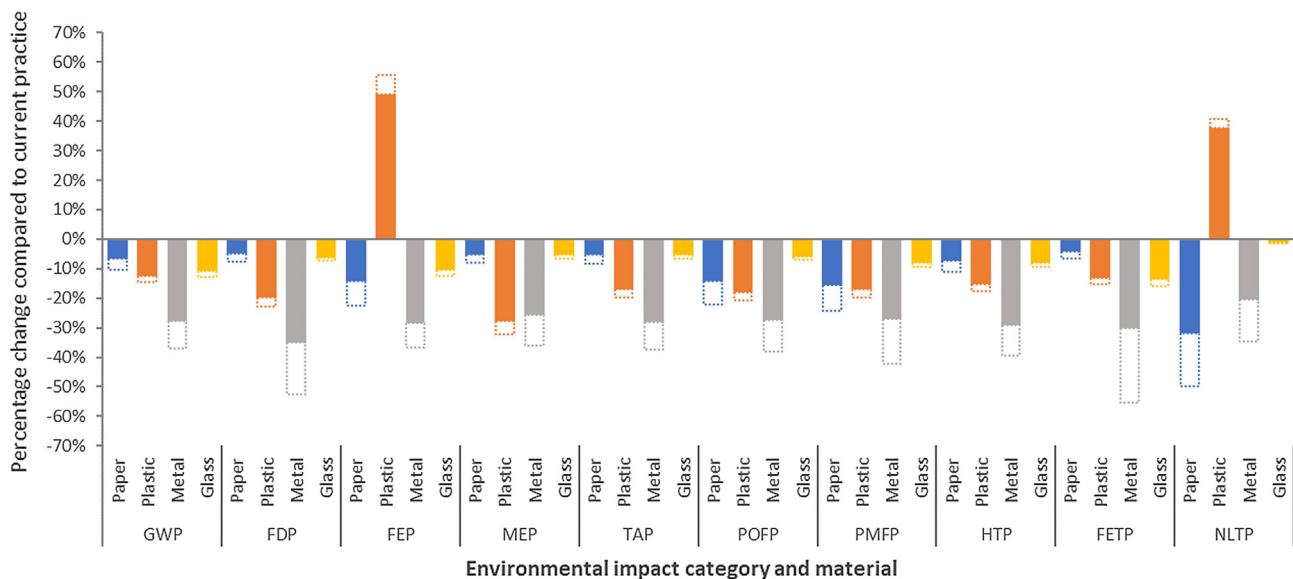


Fig. 3. The percentage change in each environmental impact category between current waste collection practices and best practice for each material (solid bars). The potential impact change should capture rates increase to levels at which EU 2030 recycling targets are met are also shown (dashed bars). All European data used to calculate capture rates were extracted from COLLECTORS (2019).

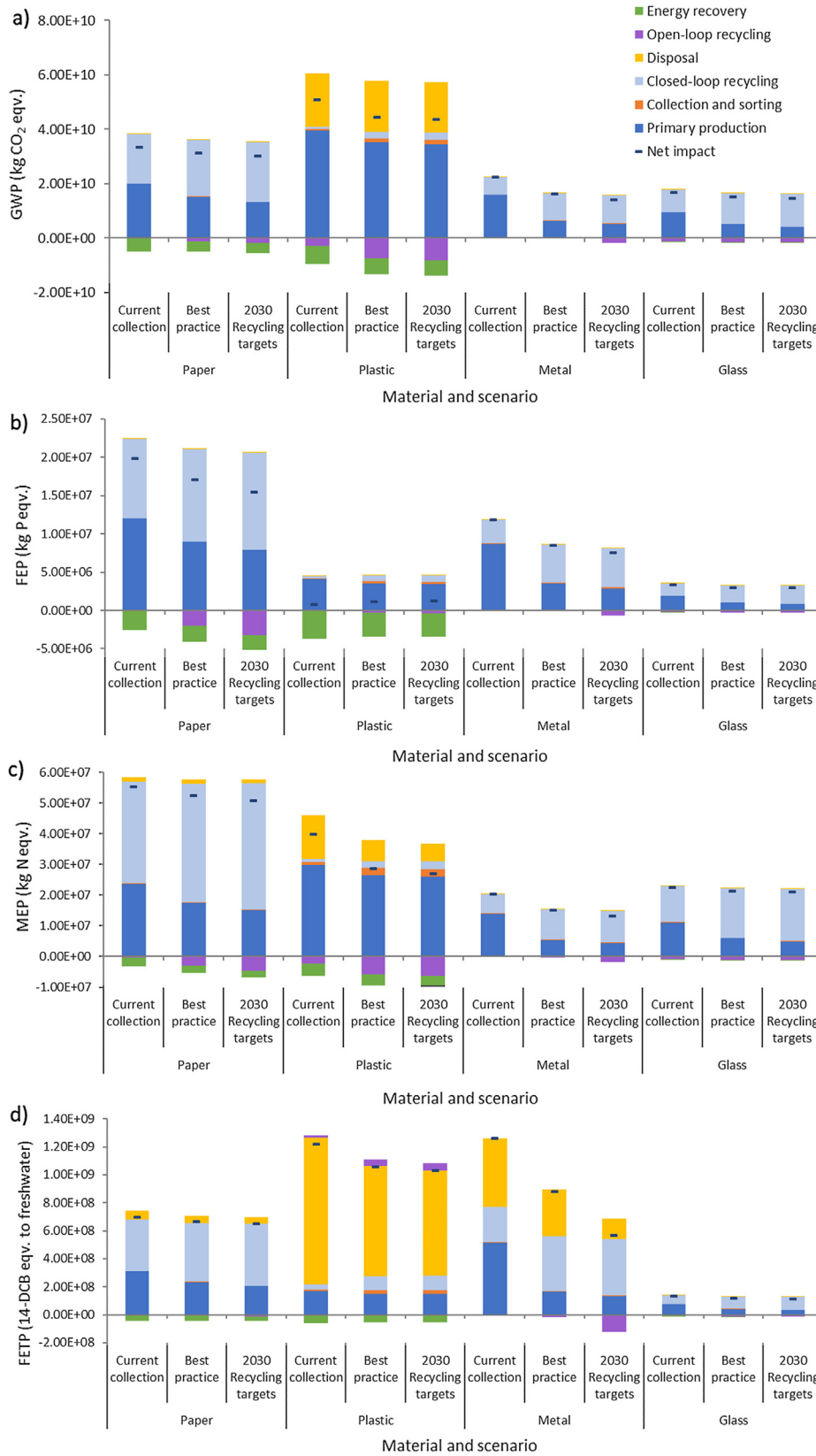


Fig. 4. Environmental impacts associated with PPW materials discerned by life cycle stage, in current waste collection practice, the best practice scenario and the 2030 recycling targets scenario (i.e. should capture rates increase to levels that EU 2030 recycling targets are met). The impact categories shown are a) Global warming potential (GWP), b) freshwater eutrophication potential (FEP), c) marine eutrophication potential (MEP) and d) freshwater ecotoxicity potential (FETP). All other impact categories assessed are reported in the Supplementary material. All European data used to calculate capture rates were extracted from COLLECTORS (2019).

impacts related to electricity production were unaltered between scenarios; most of the impacts associated with future electricity production should be reduced as the proportion of energy derived from renewable sources increases in Europe.

There was an axiomatic increase in the collection and sorting associated with the best practice scenario, and thus the environmental impacts attributed to these activities, compared with current collection. However, the environmental impacts associated with the collection and sorting of the PPW accounted for only a small portion of the overall impact for each impact category. For instance, the highest contribution from the waste collection and sorting stage was observed in the MEP impact category; accounting for only 1.0% and 2.5% of the net impact for the current collection and best practice scenarios respectively.

The PPW accounts for 2.8% of the greenhouse gas produced in Europe (European Environment Agency, 2005). Thus, improving recycling rates achieved by current collection to recycling rates achieved by the best practice scenario would result in a 0.37% reduction in greenhouse gas production in Europe, or 0.47% if the targets outlined by the recent PPW directive were to be met. However, Europe must reduce greenhouse gas emissions by a further 23% in order to meet targets outlined in the Paris climate agreement (United Nations, 2016). Reducing greenhouse gas produced by energy production, industry and transport remain the priorities in the threat of runaway climate change. Closing the material loops is still an important goal however, as incinerating the material with energy recovery will become a relatively less environmentally beneficial alternative as Europe continues to develop its renewable energy industry.

3.4. Sensitivity analysis

The sensitivity analysis shows at which points of the WCS targeted improvements can have the most impact (Table 2). The effects of adjusting each parameter on the net environmental impact values for current collection were calculated. The gross total change in the environmental impact values associated with adjusting all parameters simultaneously (including capture rate, sorting and recycling efficiencies, incineration rate and using only the PMC commingling method) has also been reported; the change is synergistic, meaning that the gross total change in the value for each environmental impact category is greater than the sum of the change related to each parameter.

The greatest total reductions in the environmental impacts resulted from increasing the material capture rates. However,

some individual materials did not follow this trend. The sensitivity analysis emphasises the importance of the sorting and recycling efficiencies of plastic on the environmental impacts; increasing either of these efficiencies had a larger effect on GWP than increasing the capture rate of plastic. In order to make a major contribution towards a circular economy, it is necessary to reduce sorting and recycling losses by reducing contamination at the collection stage and improving recycling technology. The sensitivity analysis suggests that if all municipalities were to switch to the PMC commingling method, which assumed reduced material losses at the sorting and recycling stages but all other parameters (including capture rates) were unaffected, GWP would be reduced by the same amount as increasing the sorting efficiency of plastic at the MRF by 10% (Table 2).

The EU aims to reduce landfilling of PPW, thus the sensitivity of the impact categories to the assumed rate of residual waste incineration (with energy recovery) was tested. Increasing the incineration rate of residual waste reduced the impact value of most categories. Notably, by increasing the incineration rate of residual waste in the best practice scenario only, the FEP and NLTP of plastic were decreased in the best practice scenario compared to current collection. Whilst increased greenhouse gas production is an inevitable consequence of the increased combustion of waste (Table 2), the associated impact category value reductions were mainly related to the avoided conventional energy production due to energy recovery. Again, this results from the displacement of current European energy generation, the impacts of which are likely to be reduced in the future with the increased reliance on renewable energy. However, the increased FETP was due to metal not being removed from residual waste or from the incineration bottom ash in the model presented here; this highlights the environmental importance of post collection metal recovery.

4. Conclusion

In this study the improvements that can be made in European PPW collection by following the example of best practice, switching ubiquitously to the PMC commingling method, were assessed. The capture rates of paper, plastic, metal, composite material and glass could be increased by, on average, 29%, 138%, 65%, 118% and 37% respectively, by improving collection rates to observed best practice throughout Europe. However, the recycling targets outlined by the EU were shown to be difficult to achieve by improving the performance of source separation WCS alone. Even where the model was run with 100% capture rate for each PPW

Table 2
Percentage (%) change in the net environmental impact values associated with PPW in the current collection scenario when the model parameters were adjusted individually by 10%, or PPW were to be collected via a PMC commingling method. The effects of adjusting all of these parameters on the environmental impact values are also shown.

Parameter	Material	GWP	FDP	FEP	MEP	TAP	POFP	PMFP	HTP	FETP	NLTP
Capture rate (+10%)	Paper	-0.55	-0.36	-2.34	-0.56	-0.49	-1.07	-1.44	-0.96	-0.24	-8.56
	Plastic	-0.28	-0.67	0.02	-0.55	-0.25	-0.44	-0.23	-0.15	-0.29	-0.01
	Metal	-0.79	-0.59	-1.49	-0.60	-1.01	-0.74	-1.14	-1.25	-1.82	-0.03
	Composite material	0.00	-0.02	-0.06	-0.02	-0.02	-0.02	-0.02	-0.01	0.02	-0.17
	Glass	-0.38	-0.19	-0.25	-0.21	-0.33	-0.24	-0.40	-0.21	-0.12	-0.02
	Total	-1.99	-1.82	-4.12	-1.94	-2.09	-2.52	-3.25	-2.59	-2.45	-8.78
Sorting efficiency(+10%)	Paper	-0.04	0.02	-0.43	-0.12	-0.06	-0.26	-0.34	-0.24	-0.09	-2.31
	Plastic	-0.63	-0.43	0.47	-0.07	-0.07	-0.35	-0.10	-0.07	-0.45	0.09
	Metal	-0.55	-0.45	-1.02	-0.43	-0.70	-0.53	-0.89	-0.88	-1.58	-0.03
	Glass	-0.38	-0.19	-0.25	-0.21	-0.33	-0.24	-0.40	-0.21	-0.12	-0.02
	Total	-1.60	-1.05	-1.23	-0.84	-1.17	-1.38	-1.73	-1.40	-2.24	-2.26
Recycling efficiency(+10%)	Paper	-0.14	0.06	-1.58	-0.44	-0.21	-0.95	-1.26	-0.88	-0.32	-8.54
	Plastic	-0.90	-0.69	-0.03	-0.22	-0.28	-0.47	-0.26	-0.29	-0.54	-0.01
	Total	-1.04	-0.63	-1.60	-0.66	-0.48	-1.43	-1.51	-1.17	-0.86	-8.56
Incineration of residual waste (+10%)		0.38	-0.60	-1.15	-0.85	-0.43	-0.22	-0.32	-0.08	1.78	-0.25
Collection Method (100% PMC)		-0.63	-0.40	-0.01	-0.19	-0.13	-0.59	-0.42	-0.29	-0.52	-2.17
Gross Total		-5.25	-4.70	-7.51	-4.50	-4.33	-6.30	-7.24	-5.63	-4.95	-21.1

material, there was an insufficient quantity of plastic polymers, as well as certain grades of paper fibres and types of cullet, to fully meet the demand of paper and packaging production and to close the loops. Thus, in addition to increasing capture rates, losses along the resource recovery process chain need to be minimised.

The WCS employed by a municipality is only one factor affecting overall recycling rates; sorting and recycling limitations, competition between recycled and virgin materials, price volatility of recycled materials, as well as social and socioeconomic factors (e.g. waste management policies, education, willingness to participate and financial incentives such as pay-as-you-throw schemes) are also important considerations in closing the loop. The cost that improving recycling rates presents to citizens is also important when considering the sustainability of a WCS (Da Cruz et al., 2014, 2012). Increasing PPW collection must be combined with increased post collection recovery efficiency, which will be achieved via systemic waste management improvements, including better packaging design and technological advancements at the sorting and recycling stages (e.g. by replacing mechanical recycling with chemical recycling (Rahimi and Garcíá, 2017)). Whilst recycling is an important part of the circular economy, extending the lifetime of products or phasing some of them out is also imperative; the limitations of current waste management discussed here highlight the importance of also reducing the demand for PPM, such as by introducing more ambitious waste-combating policies and abatement campaigns (Willis et al., 2018). By evaluating the circularity and environmental implications of PPW holistically in this way, the total resource recovery potential can be evaluated.

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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Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.wasman.2019.12.045>.

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