The impact of collection portfolio expansion on key performance indicators of the Dutch recycling system for Post-Consumer Plastic Packaging Waste, a comparison between 2014 and 2017

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Abstract

The recycling network of post-consumer plastic packaging waste (PCPPW) was studied for the Netherlands in 2017 with material flow analysis (MFA) and data reconciliation techniques. In comparison to the previous MFA of the PCPPW recycling network in 2014, the predominant change is the expansion of the collection portfolio from only plastic packages to plastic packages, beverage cartons and metal objects. The analysis shows that the amounts of recycled plastics products (as main washed milled goods) increased from 75 to 103 Gg net and the average polymeric purity of the recycled products remained nearly constant. Furthermore, the rise in the amounts of recycled products was accompanied with a rise in the total amount of rejected materials at cross docking facilities and sorting residues at the sorting facilities. This total amount grew from 19 Gg in 2014 to 70 Gg gross in 2017 and is over-proportional to the rise in recycled products. Hence, there is a clear trade-off between the growth in recycled plastics produced and the growth in rejects and residues. Additionally, since the polymeric purity of the recycled plastics did not significantly improve during the last years, most of the recycled plastics from PCPPW are still only suited for open-loop recycling. Although this recycling system for PCPPW is relatively advanced in Europe, it cannot be considered circular, since the net recycling yield is only 26 ± 2% and the average polymeric purity of the recycled plastics is 90 ± 7%.

1. Introduction

The EU strives towards a circular economy for all packaging materials to minimise the use of resources [European commission, 2015]. Whereas the circular economy has already developed largely for most packaging materials (paper & board, glass, metal), the collection & recycling system for post-consumer plastic packaging waste (PCPPW) is still least developed [European commission, 2015; Fellner et al., 2017; Afvalfonds, 2018]. Prior to 2009, only large PET bottles for water and soda beverages were collected from Dutch households via three deposit refund systems. An additional separate collection system for PCPPW was established in the Netherlands in 2009 [Bergsma et al., 2011]. The Dutch extended producer responsibility organisation for packaging waste, Nedvang, contracted cross-docking stations, sorting facilities, paid municipalities fixed fees for collecting PCPPW, organised waste transports and assured that sorted products were traded to certified recycling facilities. The collection portfolio was defined by Nedvang and included all post-consumer plastic packages. Packages were defined as objects that are sold with products inside and that are discarded without products they used to contain. All other materials and non-packaging plastics were excluded from the collection portfolio [Thoden van Velzen et al., 2013]. Several municipalities decided to retrieve the PCPPW via mechanical recovery from mixed municipal solid waste (MSW). Nedvang also facilitated and registered the sorting and recycling of the mechanically recovered PCPPW [Thoden van Velzen et al., 2013]. A detailed material flow analysis of this Dutch PCPPW recycling network in 2014 has previously been reported [Brouwer et al., 2018].

From January 2015 on, Dutch municipalities became responsible for managing the recycling chain of PCPPW as part of the national packaging agreement [Afvalfonds, 2019]. Groups of municipalities contracted waste service providers which dealt with transports, sorting facilities, trading, permits, notifications, etc. Nedvang became a monitoring organisation, registering notifications of recycling facilities and assuring the quality of sorted products by monthly quality inspections of sorting facilities. Municipalities received a fixed fee for every tonne of sorted product that has been traded to a certified recycling facility from the extended producer responsibility scheme [UMP, 2019]. With the responsibility for the PCPPW recycling chain also came the autonomy to define the width of the collection portfolio. Already in 2017 most of the municipalities had expanded the collection portfolio to plastic packages, beverage cartons and metal packages (locally known as PMD) and some chose plastic packages and beverage cartons as collection portfolio (named PD) [Thoden van Velzen et al., 2018a]. Such combined co-collection systems for packaging materials are common in Europe and many variations exist [Cimpan et al., 2015; Xevgenos et al., 2015; Seyring et al., 2016; Gallardo et al., 2018; Hahladakis et al., 2018]. These mixes of post-consumer packaging materials are often referred to as lightweight packages (LWP). The prime benefit of LWP co-collection systems over PCPPW mono-collection systems are the higher participation rates that are achieved [Woodard et al., 2006; Thoden van Velzen et al., 2019]. This translates in higher gross collection yields, which can lower the specific collection costs [Groot et al., 2014]. Therefore, expansion of the collection portfolio appears to be an attractive intervention to limit collection losses and to lower the collection costs.

The second motivation for portfolio expansion stems from the Dutch circular economy policy [VANG, 2014]. As a practical and tangible performance indicator the specific capture rate for mixed MSW was chosen which should ideally be reduced. This policy led to a myriad of changes in the collection of both mixed MSW and recyclable materials within the 388 municipalities, in terms of collection methods, carriers and frequencies. One of the most common changes was the expansion of the collection portfolio from PCPPW to PMD.

The Dutch expansion in collection portfolio from PCPPW mono-collection to PMD co-collection resulted in more residual waste in the collected material [Thoden van Velzen et al., 2019; Leenaars & Boer, 2017]]. This more heterogeneous feedstock poses a larger challenge for the sorting and recycling facilities. Multiple incumbents were worried that the sorted products and recycled products might contain more contaminants, cross-contamination of product residues might occur and agglomerates of various packages might be formed. This is the so-called quantity-quality trade-off assumption. In the scientific literature to date, the impact of portfolio expansion on PCPPW recycling systems has not been described. Hitherto several collection systems for PCPPW and LWP have been described in conjunction with their sorting and recycling infrastructure [Hahladakis et al., 2018; Dahlbo et al., 2018; Eygen van et al., 2018; Kranzinger et al., 2017]. However, no systematic technical analysis is available that describes a mono-collection system for PCPPW in recycling performance indicators before and after the collection portfolio has been expanded to LWP collection and hence there is no scientific evidence for this quantity-quality trade-off.

The objective of this paper is to study the quantity-quality-trade-off, by comparing a MFA of 2017 with the previous published MFA of 2014 [Brouwer et al., 2018], when there was a mono-collection system for PCPPW. This comparison will clarify the impacts of the recent portfolio expansion on several performance indicators of the PCPPW recycling chain. Three circular performance indicators will be discerned: the net recycling yield, the average polymeric purity of the main recycling products and the average polymeric purity of the valuable recycling products, see 2.2. Additionally several technical performance indicators will be compared: net collection yields, sorting division in terms of recovered masses, sorting fates, composition of sorted products, composition of washed milled goods, etc.

2. Materials and methods

2.1 Method and scope of MFA

This paper compares two material flow analyses (MFA) of PCPPW in the Netherlands from the households to the produced washed milled goods: for 2014 [Brouwer et al., 2018] and 2017 (this article). The MFA itself is performed on the object-level (packages, non-packaging articles and residual waste components) and describes the recycling network from the civilians to the sorted products. Since there are two simultaneous retrieval methods in the Netherlands (separate co-collection and mechanical recovery from mixed MSW), these are described in separate sub-models. Stan-software is used to reconcile both sub-models [Cencic, 2016; Stan, 2012]. The subsequent mechanical recycling steps are described with transfer coefficients on the material level [Brouwer et al., 2018].

Since most of the PCPPW is co-collected with beverage cartons and metal packages in 2017, the MFA-based model for 2017 is more than a simple update. Beverage cartons, ferrous metals, non-ferrous metals and various types of non-packaging plastics are included in the datasets. Simultaneously the MFA-based model is improved to allow for the calculation of errors in the final results. The description of the PCPPW recycling network of 2014 [Brouwer et al., 2018] has also been improved and extended with an error calculation and this improved version will be the reference point for 2014 [Picuno, 2017; Thoden van Velzen et al., 2018a]. A more detailed description of the origin of the data and the MFA modelling, is given in Appendix A.

Beverage cartons and metal articles are only modelled as co-collected materials up to sorted products, as the further recycling of these materials is beyond the scope of this paper. In the Netherlands also three deposit refund schemes for large PET bottles for water and beverages are operated, with roughly 28 Gg of bottles collected annually. This is registered as post-industrial packaging waste and hence is out of scope for this paper [Bergsma et al., 2011].

2.2 Performance indicators

Three overall circular performance indicators (CPI's) were used to describe the whole network and several performance indicators for parts of the recycling network, see below. The first CPI was the net recycling yield. This yield was calculated based on the total net amount of plastic flakes originating from plastic packages present in the main recycled products (washed milled goods), divided by the net potential of plastic packages on the Dutch market. As main recycled products are considered: the sinking fraction from PET bottles and the floating fractions from PE, PP, Film and Mixed plastics (MIX). The performance of the recycling network in terms of quality was expressed with two CPI's: as the average polymer purity of all the main recycling products and as the average polymer purity of the valuable recycling products. The polymer purity is determined on the level of washed milled goods and equals the weight share of the targeted polymer. For recycled PET, PE and PP these are obviously, PET, PE and PP, respectively. For recycled Film this was PE and for recycled MIX this was PE and PP. The main recycling products are the sinking fraction from PET and the floating fractions from PE, PP, Film and MIX. The valuable recycling products are the sinking fraction from the PET bottles and the floating fractions of PE and PP. The latter recycled products are relatively pure and possess a clear positive market value.

With regard to the side-products of recycling, only the floating recycled product of the sorted product PET is typically composed of only PP and PE and as such, often sold to polyolefin recycling companies and mechanically recycled. To our knowledge the sinking fractions of the sorted product PE, PP and Film are not mechanically recycled but rather incinerated, due to the relatively high risk of PVC contamination in these materials [Ragaert et al., 2017]. The sinking product of mixed plastics is occasionally mechanically recycled as a filler in intrusion processes.

Other key performance indicators used to describe parts of the recycling chain are: the net collection yield, the sorting division in terms of recovered masses, the sorting fates per packaging type, the composition of sorted products and the composition of washed milled goods.

Sorting fates describe the distribution of separately collected packages over the sorted products, for example 76% of the PET bottles end up in the PET sorted product, 18% in the other plastic sorted products and 6% in the sorting residue. For additional insights also the compliance of the sorted products to existing DKR-specifications is verified and the End-of-Life fates per packaging type, the recovered masses of mechanical recycling and the origin of polymeric contaminants in the recycling products were calculated. The end-of-life fates describe the distribution of packaging types over recycling products, residues and mixed MSW in relation to the potential present at the households.

3. Results

3.1 PCPPW Recycling network of 2017

The PCPPW recycling network for the Netherlands in 2017 and the most important net masses are schematically shown in figure 1. A simplified Sankey diagram based on only the net plastic packaging weights is added as figure 2. The quality of the underlying model is good, as is eminent from the high value for the data reconciliation parameter of 0.92 and the relatively small errors in most of the results. The model estimates the potential of plastic packages (amount of PCPPW at households) for Dutch households in 2017 to equal 350 ± 7 Gg net for 2017 (or 20.4 ± 0.4 kg net.cap^{-1}.a^{-1}). There are two retrieval methods of PCPPW in the Netherlands: separate (co)-collection and mechanical recovery from mixed MSW.

Plastic packages are separately collected via three separate collection schemes for LWP: Plastics (P), Plastic together with Beverage Cartons (PD) and Plastics together with Beverage Cartons and Metal packages (PMD). Of these separate collection schemes, the PMD co-collection scheme contributes the most PCPPW. Quality inspections have been introduced at either cross-docking stations or the entrance gate of sorting facilities and collected material that is too polluted is now rejected [Vereniging Afvalbedrijven, 2017]. In 2017 roughly 6% of the collected LWP was rejected for this reason, which amounts to 15 Gg gross. This rejected LWP has an average share of residual waste of 30% (based on gross weights).

These separately collected materials are fed as mixed input in 6 different sorting facilities to produce 9 sorted products and a sorting residue. The names of these sorted products and their corresponding sorting DKR-specification codes are: PET bottles DKR 328-1, PET-trays KIDV 05/2016, PE DKR 329, PP DKR 324, Films DKR 310, MIX DKR 350, Beverage cartons DKR 510, Ferrous metals DKR 410 and Non-Ferrous metals DKR 420 [DKR, 2019; KIDV, 2016]. These sorted products are traded to recycling companies as baled goods, with the exception of

PET trays. This material was stored in 2017 for future recycling. In 2018 a dedicated recycling facility for PET trays was being constructed, but unready to commence operations. These recycling companies produced 5 types of main products (washed milled goods) with a combined weight of 103 ± 7 Gg net, 5 types of side-products with a combined weight of 31 ± 1 Gg net and 20 Gg of process waste [Brouwer, et al. 2019b, Table P]. The main products are the sinking fraction of the sorted product PET bottles and the floating fractions of the sorted products PE, PP, Film and MIX. These are traded as recycled PET (rPET), recycled PE (rPE), recycled PP (rPP), recycled film (rLDPE) and recycled polyolefin-mixture (rPO).

The majority of the plastic packages (216 Gg net) are discarded with the mixed MSW. Approximately 19 ± 3% of the Dutch mixed MSW was subjected to mechanical recovery in four mechanical recovery facilities. One new recovery facility commenced operations in the summer of 2017, but besides this new facility the mechanical recovery network hardly changed between 2014 and 2017. These recovery facilities make intermediate plastic concentrates of which the weight is not registered and which are subsequently sorted in similar sorted products as made from separate collection. The residues are forwarded to incinerators. The total gross weight of recovered sorted plastic products amounted to 23 ± 3 Gg gross in 2017. All these sorted products, with the exception of PET trays, were traded to mechanical recycling facilities. The recycling companies produced 11 ± 1 Gg net of main products, 4 ± 1 Gg net of side products and 4 Gg of process waste.

The complete PCPPW recycling network yielded 103 ± 7 Gg net of plastic products (main milled goods) and 31 ± 1 Gg of side products in 2017 [Brouwer et al. 2019b, Table P]. Hence, the net recycling chain yield for plastics is 38 ± 2% in case both the main and side products are considered, which lowers to 32 ± 2 % in case only the main products are regarded as recycling products and lowers even further to 26 ± 2% in case only the contribution of packages to these main products are considered.

3.2 Comparison of PCPPW recycling networks

The major difference in the PCPPW recycling network between 2014 and 2017 is the widening of the collection portfolio from only plastic packages to mostly PMD. The mechanical recovery part of the network hardly changed at all. The difference in performance between the PCPPW recycling networks in 2014 and 2017 is discussed with the three CPI's and several more detailed technical performance indicators.

3.2.1 Circular performance indicators

The three circular performance indicators of the PCPPW recycling network for 2014 and 2017 are listed in table 1. The net packaging recycling yield grew from 20 ± 2% to 26 ± 2%. The average polymer purity reduced slightly from 91 ± 6% to 90 ± 7% and the average polymer purity of the valuable fractions improved marginally from 94 ± 4% to 95 ± 3%. The marginal reduction in average polymer purity of the washed milled goods is a consequence of the added data to the model, that had more black packages in the sorted product Film and MIX. The polymeric purity of the valuable recycling products changed slightly. The purity of the PE- and PET- main washed milled goods increased slightly, whereas the purity of the PP washed milled goods decreased slightly for both the separate collection and the recovery system. In paragraph 3.2.6 the polymeric purity of the washed milled goods from separate collection of LWP is described in more detail. In short, the total amount of washed milled goods has increased between 2014 and 2017 and the average polymeric purity of the recycled plastics has only changed marginally.

3.2.2 Net collection yield

The net collection yield for plastic packages equals 38 ± 2% for 2017. This is substantially more than the previously reported 25 ± 3% for 2014 [Brouwer et al., 2018; Thoden van Velzen et al., 2018a]. This translates in a rise from 86 to 135 Gg net of separately collected plastic packages between 2014 and 2017. Thus the broadening of the collection portfolio by the municipalities successfully enhanced the net collection yields for PCPPW in the Netherlands. The rise in gross collected amounts of LWP is substantially larger, from 129 Gg in 2014 to 254 ± 14 Gg in 2017, and can only partially be attributed to an increase in the net weights of the targeted packaging materials. Besides the targeted plastic packages, beverage cartons and metal packages, also more residual wastes are collected with the LWP. The share of residual waste in the LWP that is accepted for sorting grew from 9% in 2014 to 12% in 2017 [Brouwer, et al. 2019b, Table J].

The growth in net collection yield as a consequence of the portfolio expansion is likely to be caused by an increase in the participation rate [Woodard, et al. 2006]. To verify this hypothesis, the minimal participation rate is calculated with a newly developed method [Thoden van Velzen et al., 2019]. The minimal participation rates amounts to 55 ± 5 % for 2017 and 37 ± 6 % for 2014. Therefore, the expansion of the collection portfolio indeed raises the participation rate which in turn raises the net collection rate.

3.2.3 Sorting division in terms of recovered masses

The sorting division in terms of recovered masses for 2014 and 2017 is shown in Table 2. Three new sorted products are introduced for the co-collected packaging materials; beverage cartons, ferrous metals and non-ferrous metals. Moreover, a new sorted plastic product (PET trays) is introduced. Consequently, the sorting division has adapted to accommodate these changes and the shares of main plastic products PET bottles, PE and PP have decreased slightly. The share of the sorted product Film has decreased much more than what would be expected to accommodate the newly included materials, whereas the ratio between the flexible plastic packages and all plastic packages in the collected material remain nearly constant [Brouwer, et al. 2019b, Table D and F]. This implies that the sorting fates for flexible packaging materials have changed. The share of MIX formed decreased from 37 ± 7% in 2014 to 26 ± 5% in 2017. This is predominantly caused by the introduction of the new sorted product for PET trays (7 ± 1% in 2017), which used to end up in the Mixed plastics and are now sorted in a separate product. Relatively much more sorting residues were produced from the collected materials in 2017 than in 2014, 22 ± 4% compared to 15 ± 1% (table 2). This can partially be explained by the slightly increased levels of residual waste in collected materials.

3.2.4 Sorting fates per packages type

A detailed comparison between sorting fates of separately collected LWP between 2014 and 2017 (Table 3) [Brouwer et al., 2019b Table L] reveals that many types of rigid plastic packages have been sorted to a larger extent to the correct sorted product. In contrast to the rigid packages, the flexible packages, and especially the relatively large PE flexible film categories, are sorted to a lesser extent to the sorted product Film and to a larger extent to the sorted product MIX. Furthermore, although a new sorted product has been created for PET trays, still substantial amounts of PET trays were found in the MIX. Also, for rigid PVC packages a remarkable change in the sorting fate is noticed. These rigid PVC packages should be added to the sorting residues, however, the comparison with 2014 reveals that in 2017 more of these packages end up in the MIX and less in the sorting residues. Packaging types that aren't targeted for collection and cannot be recycled, are laminated flexibles and drug blisters, these packages are sorted to a larger extent to the sorting residue, and therefore form a smaller source of contamination in 2017 than in 2014. The non-packaging plastics are sorted roughly in the same manner as in 2014. Residual waste present in the collected materials and categorised as "organics and undefined" was sorted to a lesser extent to the MIX in 2017 as compared to 2014 and to a larger extent to the sorting residues. Since more residual waste was present in the collected material, this implies that more effort had to be performed by the sorting companies to remove this waste to maintain the quality.

3.2.5 Composition of sorted products

The composition of the sorted products made from the separately collected materials is given in table 4. From this table it is apparent that in general the composition of the sorted products has only changed to a limited extent. The share of targeted plastic packages in the sorted products has increased slightly for most sorted products, with the exception of MIX. In the MIX the share of non-packaging plastics, beverage cartons and metals has increased. In the sorted products PP and Film a slight increase was observed in beverage cartons, paper & board and metals. Hence, some cross-contamination is occurring in the sorted products as a consequence of the expansion of the collection portfolio. Nevertheless, these amounts are relatively small in comparison to the amounts of non-targeted plastic packages and non-packaging plastics in these sorted products.

The average composition of the sorted products made from separately collected LWP was also compared to the DKR-specifications [Brouwer, et al. 2019b, Table O]. Sorted products regularly did not comply to the specifications and this situation hasn't changed much between 2014 and 2017. Therefore, the expansion of the collection portfolio, which increased the average level of residual waste in the collected LWP, apparently had little influence on the level of compliance to the specifications of the sorted products.

The levels of attached moisture and dirt (LAMD) of sorted products decreased between 2014 and 2017, see table 4. Hence, the co-collection of different packaging materials does not result in a noticeable exchange of moisture and product residues between these materials.

3.2.6 Composition of washed milled goods

The composition of the main washed milled goods in terms of polymers and other materials is given in Table 5 [Brouwer et al., 2019b, Table S]. Only minor changes in the modelled compositions of 2014 and 2017 are observed.

The polymer purity of the PE washed milled goods from separate collection increases the most, due to a decrease in PP contamination. Less PP non-beverage bottles and PP thermoforms and rigid packages are faultily sorted into the PE sorted product, as a result of a reduction of these PP packages in the collected materials and improved sorting fates.

The polymer purity for the PP washed milled goods from separate collection decreases the most, due to an increase in PE contamination. More PE based packages (beverage bottles, non-beverage bottles and rigid packages) end-up in the PP sorted product, as result of raised sorting fates of these packaged towards the PP sorted product.

3.2.7 Other performance indicators

The same pattern in collection fates, sorting fates of mechanical recovery and EoL-fates is observed in the data for 2017 as in the data of 2014 [Brouwer et al., 2019b, Table K, M and N]. The only difference is that these parameters are in general higher for both correctly and faultily sorted and consequently less packages are not recycled and incinerated. The composition of the main washed milled goods is further analysed in terms of the share of desired polymer from targeted and non-targeted packages and objects, non-intended polymers and contamination from different materials [Brouwer et al., 2019b, Table T and U]. These compositions hardly change between 2014 and 2017. This is expected since these compositions are governed by the compositions of the sorted products and the material composition per packaging type. The former hardly changed between 2014 and 2017 and the latter is constant by assumption. Similarly, also the origin of the contaminants and the recovered masses hardly change [Brouwer et al., 2019b, Table V, W, Q and R].

3.3 Impact of the portfolio expansion

As a direct consequence of the portfolio expansion, the gross collected amounts of LWP almost doubled from 129 Gg gross in 2014 to 254 Gg gross in 2017. This translated in an increase of the uncompressed volume of LWP at the households from 0.26 to 0.50 $m^3.cap^{-1}.a^{-1}$, assuming a LWP density at consumers of 30 $kg.m^{-3}$. Simultaneously the National VANG policy encouraged municipalities to lower the specific capture rate for mixed MSW. To accommodate the increase in LWP volume and to reduce mixed MSW generation, the 388 municipalities responded in various manners, for example by increasing the collection frequency for LWP from four weekly to fortnightly and decreasing the collection frequency for mixed MSW, by changing the carrier of LWP from bags to mini-containers and by implementing reversed collection schemes. The latter implies that recyclable materials are collected with kerbside collection systems and that mixed MSW changed to drop-off collection. Both the introduction of mini-containers and the implementation of reversed collection systems have been reported as risk factors for larger shares of residual waste in the LWP [Leenaars & Boer, 2017]. Indeed the share of residual waste in the collected LWP, which was accepted for sorting, grew from 9% to 12%. The rise in the share of the residual waste in the LWP is therefore not directly caused by the portfolio expansion itself, but rather indirectly by the concomitant changes in the collection schemes of LWP and mixed MSW such as carriers, methods and frequencies.

The larger share of residual waste in the collected LWP increased the challenge for sorting companies to make sorted products that comply with the specifications. They responded by enforcing tougher visual quality inspections of each batch of collected LWP material [Vereniging Afvalbedrijven, 2017] and hence a new material stream of rejected LWP material was created, which amounted to 15 ± 5 Gg in 2017. Secondly, sorting facilities started to produce much larger amounts of sorting residues (19 ± 1 Gg gross in 2014 and 55 ± 9 Gg gross in 2017). Both the rejected LWP and the sorting residues were incinerated. Therefore, a downside of the collection portfolio expansion, is an increase in the waste streams that originate at the cross-docking facilities and sorting facilities from 19 Gg in 2014 to 70 Gg in 2017. Hence, the portfolio expansion resulted in a quantity-quantity trade-off. The additional quantity of recycled plastics produced (+28 Gg net) traded off against an additional quantity of waste being separated off (+51 Gg gross). This trade-off has serious ramifications on the performance of the complete recycling network, since substantial amounts of rejected LWP and sorting residues are first collected and then transported to incineration facilities. It is recommended to future environmental studies of PCPPW networks to include these material flows in their analysis.

Although the sorting facilities had to do more effort in 2017 than in 2014 to produce sorted products that comply with the specifications, still the quality of the sorted products did not change significantly between both years. The quality of several sorted products (PET bottles and PE) even improved slightly in the sense that these contained less faultily sorted packages and so did the sorting fates of the corresponding packaging types. Nevertheless, achieving compliance to the specification remained a challenge for the sorting facilities, especially for the sorted products PET trays, PP, Film and MIX [Brouwer et al., 2019b, Table O].

No evidence was found for a quantity-quality trade-off in the circular performance indicators (see Table 1). Moreover, only limited evidence for cross contamination between co-collected packaging materials has been found (see 3.2.5). The manifestation of this quantity-quality trade-off has been avoided by the monthly quality inspections of sorted products [Nedvang, 2010] and the financial penalties for non-compliance [Nedvang, 2016]. This forced sorting facilities to perform as good as possible, although their feedstock became more complex and challenging. The overall consequence was a quantity-quantity trade-off; to create 28 Gg net more recycled plastics, 51 Gg more waste had to be managed.

4. Discussion

4.1 Officially reported recycling yields

The officially reported recycling yield remained constant between 2014 and 2017 and was 50% in both years [Afvalfonds, 2018] and is hence substantially higher than the net recycling yield of 26 ± 2% found in this study. The official yield is calculated as the gross amounts of sorted plastic products made from post-industrial and post-consumer plastic waste corrected by the share of non-packaging objects and the share of impurities in the sorted products above the DKR-specifications (resp. 9.8% and 4.3%) and divided by the net amounts of plastic packages introduced on the Dutch market. The net recycling yield in this study was calculated based on the total net amount of plastic packages present in the main recycled products (washed milled goods) from post-consumer plastic waste, divided by the net potential for post-consumer plastic packages on the Dutch market.

A higher official recycling yield was expected for 2017, as the amount of recycled plastics made from PCPPW clearly increased between 2014 and 2017. However, the official yield did not increase in this time period, which is attributed to a reduction in the amount of recycled post-industrial packaging materials in this time period [Afvalfonds, 2018].

4.2 Implications for the circular economy

Both the Dutch government and the EU commission strive towards a more circular economy for plastic packaging materials. Progress has been made between 2014 and 2017 with respect to the quantity of recycled plastics made of PCPPW, but no progress was achieved with respect to the quality of the recycled plastics made (table 1). The purity of the recycled PET is quite good at 99% but the purity of the other recycled plastics (compositions shown in Table 5) remains insufficient for circular applications such as packages, household appliances and other consumer products.

As soon as the concentration of another polymer in the main polymer exceeds 2%, the material is considered a polymer blend [Utracki, 2002]. All polymers – even PP and PE – will form thermodynamically immiscible blends [Manias & Utracki, 2014]. While some synergistic blends do exist, mechanical properties are typically reduced for the blends occurring in the modelled washed milled goods under consideration. The cross contaminations of PE and PP in one another are between 5 and 10%, which is at best dubious in terms of acceptability. Striving towards a recycling system that would allow to go below the 5% cross-contamination threshold might severely increase the qualities of the resulting PP and PE secondary materials. The main recycled products made of Film and MIX are typical mixed polyolefin (PO) products. These polyolefin blend materials are regularly contaminated with PET, PVC, PS, and other polymers from laminates such as PA, EVOH, PVdC, etc. and hence these recycled plastics can typically only be applied in relatively thick-walled bulk products like garden furniture.

On top of the effects caused by the occurrence of immiscible blends, PVC contaminations in recycled polymers should be avoided at all cost. PVC is much less thermally stable than PP, PE or PET [Yu et al., 2016] and the virgin stabilizers are not guaranteed to still be active in the polymer, which could lead to the development of hydrochloric acid in the extrusion process, which is known to cause heavy corrosion on equipment as well as accelerate degradation of the main recycled polymer [Ragaert et al., 2017]. For mechanical recycling of PET, 50 ppm of PVC contamination is an accepted limit. PVC and PET will strongly accelerate each other's degradation [Awaja & Pavel, 2005]. Considering the material composition in Table 5, both the PP and MIX fractions exceed these tolerances for PVC contamination, thus damning them to low-quality applications. In the recycling products made from PET, Film and PE, PVC contaminations could be considered tolerable.

Recycling companies can apply more advanced recycling processes and produce recycling products with lower levels of polymeric contamination. Fairly common is the use of melt-filtration. This technique can help to remove inorganic contaminants from the recycled plastics and even PET contaminants from recycled PE and PP, although it can create substantial material losses [Brouwer et al., 2018]. Alternatively, the purity of recycled PET, PE and PP can be improved with flake sorting techniques. Most of these flake sorting machines sort on colours, NIR spectra and laser reflectivity. A single flake sorting machine can produce a recycled plastic with polymeric purities that exceed 99.5%, but at the expense of substantial losses of material and at quite low sorting speeds. Hence, this can only be operated in situations when there are outlets for recycled plastics with mediocre polymeric purities (80-95%) as side products. To minimise the production of recycled plastics with lower polymeric purities, multiple series of flake sorting machines are applied in a few advanced recycling companies. Although recycled plastics with high polymeric purities (>99%) at relatively high yields can be obtained [Langen, 2018], the required investments are high and hence this is not standard, yet.

Hence, although more recycled plastics were produced from Dutch PCPPW in 2017 than in 2014, they were hardly suited for fully circular, more closed loop applications. Currently, open loop recycling pathways are chosen, in which the applicability of the produced blends is determined by their blend composition, colour and odour. On the short term, there is a need for more open loop recycling end markets, but more importantly, there is a need to achieve higher polymer purity's in order to allow more high-end applications.

An alternative approach to advance towards purer recycled plastics is design for recycling. Multiple improvements will be needed with regards to the packaging design, to lower the polymeric contamination of the recycled plastics. First of all, the share of non-recyclable plastic packages has to be reduced. This share was estimated to be 28% in 2014 [Brouwer et al., 2017] and mainly consists of black/non-NIR identifiable packages, PS packages, packages of 'other' materials (PC, PLA, etc.), laminates and drug blisters. Their presence in the collected materials causes the need for a MIX sorted product. In case the share of non-recyclable packages can be reduced, less MIX has to be produced and less good recyclable plastic PE and PP packages will be lost to the MIX [Brouwer et al., 2017]. Secondly, the designs of 'good recyclable' plastic packages also need to be improved, since within these good recyclable packages design components are still present that cause polymeric contamination. Previous research showed that by improving the designs, to allow for improved sorting fates and simple removal of unwanted components during mechanical recycling, the polymeric purity of the main recycling products PET, PE, PP and film can be improved. It will require a concerted action of all stakeholders to progress towards not only more recycled plastics but also more recycled plastics with a high polymeric purity [Thoden van Velzen et al., 2018a].

4.3 Future outlook

The recycling network for Dutch PCPPW is dynamic and in 2018 and 2019 two large new mechanical recovery facilities will start to become fully operational, with a combined input capacity of 440 Gg [Thoden van Velzen et al., 2018a]. This will create an enormous increase in the quantity of recycled plastics produced. Furthermore, a new sorting facility for the sorting of mechanically recovered plastics just opened and the construction of an additional sorting facility for separately collected LWP has been started.

In 2018 a 24 Gg recycling facility for sorted films opened which produces a relatively pure recycled LDPE for film applications. Furthermore, a new 24 Gg PET tray recycling line is being tested. Unfortunately it has not yet commenced operations. Simultaneously, a 10 Gg depolymerisation facility for coloured PET bottles, textiles and PET trays is being build, which is planned to be operational in 2019. A large PE and PP recycling plant, aiming for pure PE and PP flake-sorted products is being constructed and another recycler is gradually expanding its production capacity from 35 to 100 Gg for producing highly pure flake-sorted PE and PP recycled products.

Hence, all these developments show that in coming years the recycling network for PCPPW will expand and more recycled plastics are expected.

Simultaneously with the efforts of the sorting and recycling companies, multiple producers of packaged goods are engaged in redesigning their packages for recycling. Most of them strive to only use recyclable packages by 2025 or 2030.

5. Conclusion

The Dutch PCPPW recycling network was modelled for 2017 from the households to recycled flakes. The quality of the model was good, as was eminent from the high parameter for the data reconciliation process and the relatively small errors in most of the calculated results. The net recycling yield for Dutch PCPPW grew from 20 ± 2 to 26 ± 2% between 2014 and 2017 in case only the main recycling products made from packaging plastics are considered. In 2017 approximately 103 Gg of recycled plastics were produced from Dutch PCPPW, as compared to 75 Gg in 2014. This rise in recycling yield was accomplished by the expansion of the collection portfolio from only plastic packages to also beverage cartons and metal objects. The consequential change in volumes of mixed MSW and LWP caused the collection agencies to change the collection schemes, which in turn caused the amount of residual waste in the collected LWP to rise. As a consequence, collected LWP had to be rejected and also more sorting residues had to be separated off (in total 70 Gg in 2017 compared to 19 Gg in 2014). Hence a clear quantity-quantity trade off was found; the rise in recycled plastics traded off against a larger amount of waste that was created at the sorting facilities. The quality of the recycled plastics in terms of polymeric composition changed only marginally, implying that most of the recycling products are too impure for circular applications and still substantial design-for-recycling efforts or subsequent flake sorting steps are required. Therefore, the Dutch PCPPW network has progressed in the last years in terms of quantities produced, but not in terms of the qualities produced. Substantial efforts need to be made to bridge the gap in the qualities that are required for a circular economy and those currently produced.

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Abbreviations

CPI circular performance indicator, DKR Deutsche Kunststoff Recvcling, EoL End-of-Life, EVOH poly ethylene-co-vinyl alcohol, LAMD level of attached moisture and dirt, LWP light-weight packaging waste, LDPE low density poly ethylene, MFA material flow analysis, MSW mixed municipal solid waste, MIX mixed plastics, PA polyamide, PC polycarbonate, PCPPW post-consumer plastic packaging waste, PD plastic packages and beverage cartons, PLA poly lactic acid, PMD plastic packages, metal packages and beverage cartons, PE polyethylene, PET poly(ethylene terephthalate), PO polyolefin, PP polypropylene, PS polystyrene, PVdC poly vinylidene chloride, PVC poly vinyl chloride, PVOH poly vinyl alcohol.

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7. Figures

Figure 1: Schematic overview of the modelled PCPPW recycling network in the Netherlands in 2017.

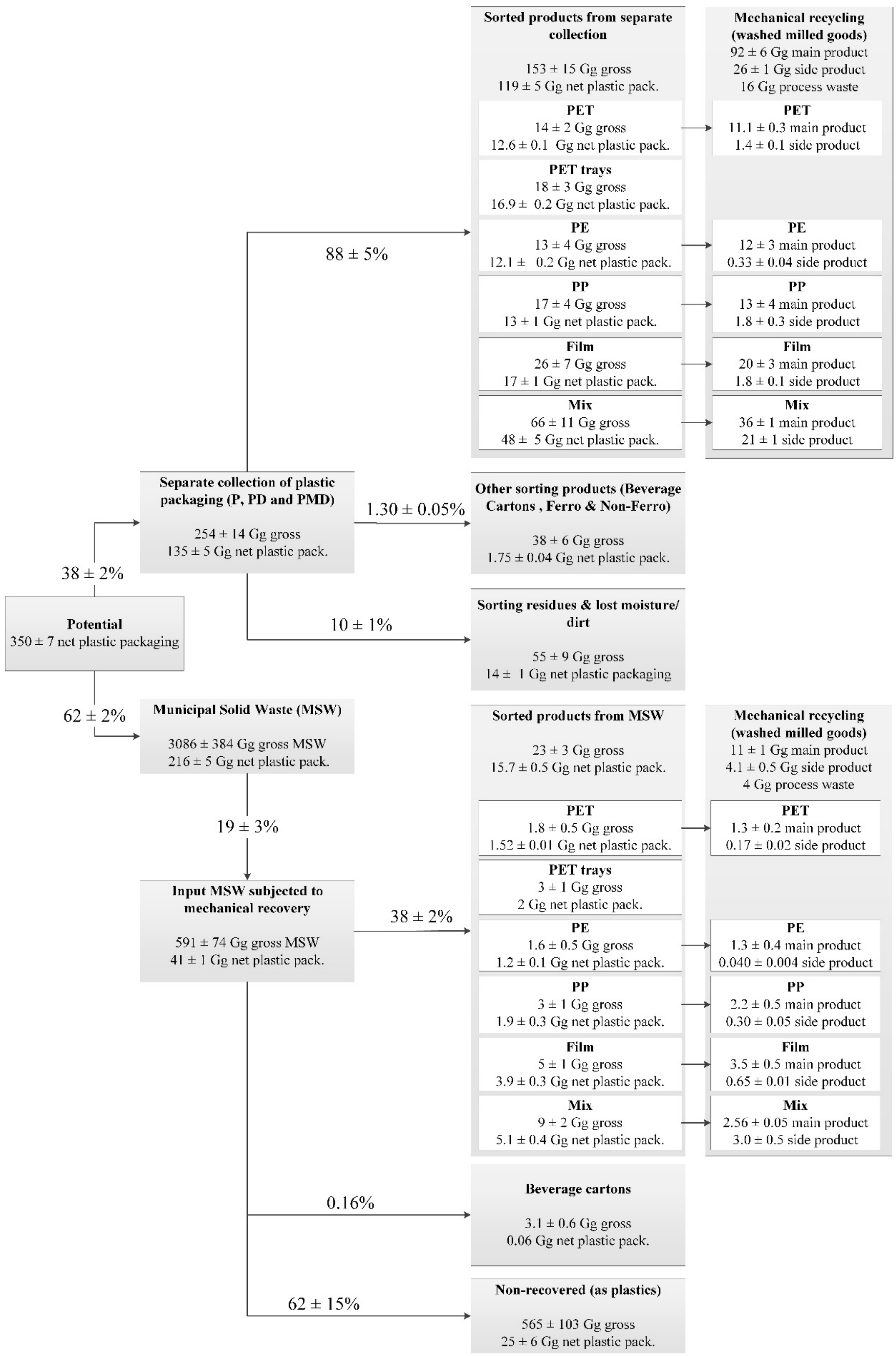

Figure 2: Sankey diagram of the PCPPW recycling network in the Netherlands in 2017. The numbers shown on the left of the "Mechanical recycling" step are net packaging weights. Due to the model´s complexity, to the right of the "Mechanical recycling" step, the difference between packages, non-packaging objects and residual waste can no longer be made and, therefore, the sum of the output amounts and the incinerated plastics results higher than the potential.

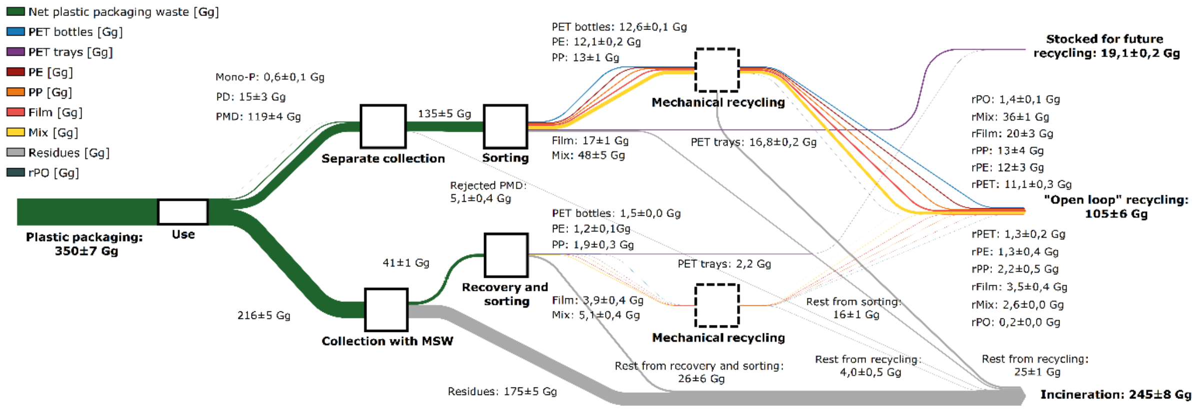

Table 1: Circular performance indicators of the Dutch PCPPW recycling for 2014 and 2017 [Brouwer et. al., 2018; Thoden van Velzen et al, 2018a].

CPI's	2014	2017
net packaging recycling yield	20 ± 2%	26 ± 2%
average polymer purity of the washed milled goods	91 ± 6%	90 ± 7%
average polymer purity of the valuable washed milled goods	94 ± 4%	95 ± 3%.

Table 2: Sorting division of separately collected Dutch LWP, after data reconciliation for 2014 and 2017 in terms of recovered masses of sorting [%]

	2014	2017
PET bottle	7±1	6±1
PET trays	NA	7±1
PE	8±1	5±1
PP	10±2	7±2
Film	21±4	10±3
Mix	37±7	26±5
Beverage cartons	NA	8±1
Ferrous metals	NA	6±2
Non-ferrous metals	NA	1±0
Sorting residue	15±1	22±4
Lost MAD	2±1	3±0

NA: Not applicable

Table 3: The approximated sorting fates of 15 representative packaging types and one type of residual waste for the LWP collection portfolio in the Netherlands in 2014 and 2017, [%]. [Brouwer et al. 2019b, Table L].

	Ideal sorting fate		**Correctly sorted**		**Faulty sorted to Mixed plastics**		**Faulty sorted to other sorted products**		**Not recycled**	
	2014	**2017**	**2014**	**2017**	**2014**	**2017**	**2014**	**2017**	**2014**	**2017**
PET bottle clear ≤ 0.5 litre	PET	PET	69%	76%	20%	12%	2%	6%	9%	6%
PE beverage bottles	PE	PE	81%	85%	12%	11%	1%	3%	5%	2%
PP beverage bottles	PP	PP	84%	79%	14%	21%	1%	0%	0%	0%
PET non-beverage bottles	PET	PET	67%	65%	23%	23%	2%	5%	7%	8%
PE non-beverage bottles	PE	PE	86%	86%	9%	9%	1%	3%	4%	2%
PET thermoforms	Mix	PET	73%	61%	NA	27%	16%	6%	11%	7%
PET other rigid packages	Mix	trays	85%		NA		12%		3%	
PE thermoforms	PE	PE	14%	28%	42%	33%	24%	6%	20%	33%
PE other rigid packages	PE		36%		50%		9%		5%	
PP thermoforms	PP	PP	52%	51%	29%	32%	14%	4%	5%	13%
PP other rigid packages	PP		69%		22%		6%		4%	
PVC thermoforms	Rest	Rest	N.A.	N.A.	15%	48%	2%	8%	83%	44%
PVC other rigid packages	Rest		N.A.		16%		5%		79%	
PE flexible packages > A4	Film	Film	70%	52%	27%	39%	2%	3%	1%	6%
PE flexible packages < A4		Film		24%		66%		5%		5%
PP flexible packages > A4	Film	Film	27%	27%	60%	64%	8%	6%	5%	4%
PP flexible packages < A4		Film		20%		64%		10%		6%
Laminated flexible packages and blisters	Mix	Mix	60%	56%	NA	NA	30%	31%	10%	13%
Organics & undefined	Rest	Rest	N.A.	N.A.	60%	19%	2%	7%	38%	74%

N.A.: Not Applicable.

Table 4. The material composition of sorted products made from separately collected LWP in 2014 and 2017, [%]. The average levels of attached moisture and dirt (LAMD) of representative plastic packages in sorted products in 2014 and 2017 are shown separately, [%].

	PET	PET Trays	PE	PP	Film	Mix	BC	F-metal	NF-metal
2014									
Targeted plastic packages	88.7%		89.8%	73.0%	77.3%	79.1%			
Other plastic packages	10.7%		8.3%	14.1%	11.7%	1.2%			
Non packaging plastics	0.3%		1.6%	12.2%	10.2%	7.6%			
Metals	0.1%		0.1%	0.2%	0.5%	0.8%			
Beverage cartons, paper & board	0.0%		0.1%	0.2%	0.1%	4.2%			
Other residue	0.2%		0.1%	0.4%	0.2%	7.1%			
LAMD	*12%*		*15%*	*11%*	*22%*	*15%*			
2017									
Targeted plastic packages	93.7%	90.8%	92.0%	73.9%	67.3%	74.0%	91.7%	89.1%	76.5%
Other plastic packages	5.4%	7.7%	5.8%	11.9%	7.8%	1.3%	4.6%	5.8%	13.1%
Non packaging	0.2%	0.4%	1.8%	10.0%	23.4%	10.7%	0.4%	1.1%	1.2%
Metals	0.0%	0.1%	0.2%	0.6%	0.2%	2.6%	0.2%	NA	NA
Beverage cartons	0.0%	0.1%	0.2%	0.4%	0.2%	2.1%	NA	0.9%	5.6%
Paper & board	0.0%	0.5%	0.0%	0.5%	1.0%	4.2%	2.3%	1.2%	2.4%
Other residue	0.7%	0.3%	0.1%	2.7%	0.2%	5.1%	0.8%	1.9%	1.1%
LAMD	*9%*	*5%*	*4%*	*10%*	*13%*	*4%*	*36%*	*8%*	*14%*

Table 5: The material composition of the washed milled goods of 2014 and 2017 [%].

		PET	PE	PP	PS	PVC	Paper	Metal	Glass	Other	Rest
PET SC	**2014**	98.6 ± 22.3	0.1 ± 0.0	0.1 ± 0.2	0.5 ± 0.2	0.1 ± 0.1	0.1	0.2	0.0	0.2	0.1
	2017	98.8 ± 3.4	0.1 ± 0.0	0.1 ± 0.1	0.3 ± 0.0	0.0 ± 0.1	0.1	0.1	0.0	0.1	0.4
PE SC	**2014**	0.0 ± 0.0	89.4 ± 11.1	9.8 ± 1.8	0.1 ± 0.0	0.1 ± 0.1	0.0	0.0	0.0	0.6	0.0
	2017	0.0 ± 0.0	92.6 ± 38.9	7.1 ± 2.0	0.0 ± 0.0	0.0 ± 0.0	0.0	0.0	0.0	0.3	0.0
PP SC	**2014**	0.1 ± 0.1	3.8 ± 1.3	93.1 ± 19.9	0.2 ± 0.1	0.4 ± 1.6	0.0	0.0	0.0	2.4	0.0
	2017	0.0 ± 0.1	5.4 ± 1.8	92.0 ± 46.2	0.1 ± 0.0	0.3 ± 0.4	0.0	0.0	0.0	2.3	0.0
Film SC	**2014**	0.0 ± 0.1	81.7 ± 26.0	14.7 ± 4.8	0.3 ± 0.2	0.2 ± 0.3	0.0	0.0	0.0	3.1	0.0
	2017	0.0 ± 0.0	82.8 ± 20.6	13.0 ± 2.1	0.1 ± 0.0	0.1 ± 0.2	0.0	0.0	0.0	4.0	0.0
MIX SC	**2014**	0.6 ± 0.9	47.5 ± 20.3	39.6 ± 14.7	3.9 ± 2.0	1.0 ± 1.4	0.0	0.0	0.0	7.4	0.0
	2017	0.2 ± 0.3	51.2 ± 2.3	36.4 ± 1.6	1.7 ± 0.3	0.8 ± 1.1	0.0	0.0	0.0	9.7	0.0

Supplementary materials

Appendix A: Origin of the data and mathematical modelling procedure

A.1. Origin of the compositional data

The compositional data of the collected materials and sorted product used in the model is a combination of new analyses (2016-2018) and the previously published data [Brouwer, et al 2018]. The new compositional data relates to the separate co-collection of PD and PMD materials and to the sorted products that were produced from the mixed input of PCPPW, PD and PMD materials. The composition of the separately collected plastics (PCPPW) was estimated to be similar as in 2014, as only a minor part of the municipalities still operate this collection system and this will not affect the overall model. The composition of the recovered sorted products from MSW was estimated to be similar as those in 2014 [Brouwer, et al. 2018]. Therefore, the compositions of only a few recovered sorted products were determined in 2017-2018 and these were added to the dataset. The composition of MSW was expanded with additional measurements from 2017 and 2018, which were added to the data from 2013 which was still considered valid. Older measurements (prior to 2013) were not used as input data for the new model. The complete overview of the modelling data for the model of the Dutch Recycling Network of Post-Consumer Plastic Packaging Waste of 2017 is presented in a dataset [Brouwer, et al. 2019b, Table A t/m I].

All the samples sorted between 2016 and 2018 were sorted according to the new sorting protocol [Thoden van Velzen et al, 2018b] in terms of 45 plastic packaging types, 10 non-packaging plastic articles, 14 types of beverage cartons, ferrous metal articles, non-ferrous articles and 4 types of residual waste. The levels of attached moisture and dirt were determined on those packaging categories mostly present in the respective samples for all studied samples according to the protocol [Thoden van Velzen et al., 2018b].

In the previous model the non-packaging plastics were combined to one single category, without distinction on the polymer level. This resulted in erroneous compositions of the washed milled goods [Picuno, 2017]. Therefore in the 2017 model the non-packaging plastics are now split into 5 different polymer types (PET, PE, PP, PS, PVC), which correlates to the 'big five' of plastics packaging materials. Additionally, flexible packages were divided in two categories: smaller than A4 and larger than A4. These adjustments result in a list of 37 plastic packaging types, 7 non-packaging plastic articles, beverage cartons, metal articles and 4 types of residual waste which is used in the model [Brouwer et al. 2019b, Table D t/m I].

A.2. Origin of the gross amounts of collected, sorted and recovered LWP

The total gross amounts of separately collected PMD, PD and mono-P in the Netherlands in 2017 were obtained from Stichting Afvalfonds, the Dutch extended producers responsibility organisation for packaging waste, the data is presented in a dataset [Brouwer, et al. 2019b, Table A]. The total gross amounts of sorted products made in 2017 from this mixed feedstock were partly obtained from Stichting Afvalfonds (PET, PE, PP, Film and Mix) and partly (PET trays, Beverage Cartons, Ferrous Metals, Non Ferrous Metals, Crude residues and Fine residues) reconstructed based on a weighted average sorting distribution of two major sorting facilities of Dutch LWP [Brouwer et al. 2019b, Table A and C].

The total amount of collected LWP that was rejected at cross-docking stations and at sorting facilities in 2017 is not officially registered and hence unknown. The collected material is rejected in case too much contaminants are visually present during the unloading. Operators of these facilities approximated their rejection levels between 0 and 15%. The national averaged rejection level for Dutch LWP in 2017 was estimated to be 6%, based on a comparison between the nationally reported weights of sorted products PET, PE, PP, Film and MIX and the calculated weights based on the sorting distribution of two major sorting facilities [Brouwer et al., 2019b, Table C]. At a 6% rejection level these amounts were most similar. The composition of three collected PMD samples with high levels of residual waste (>20%) were averaged to describe the composition of rejected PMD material [Brouwer et al., 2019b, Table D].

The total amount of mixed MSW collected in the Netherlands in 2017 was obtained from Statistics Netherlands – the Central Agency for Statistics CBS [CBS Statline, 2018]. The amount of mixed MSW subjected to mechanical recovery and the total amounts of sorted products made from the recovered MSW were obtained from the management of the four active recovery facilities, this data is presented in the dataset [Brouwer et al., 2019b, Table B].

A.3. Mathematical modelling

The structure of the Dutch LWP recycling chain in 2017 has changed with respect to the one in 2014 in two aspects: the collection portfolio expanded and the collected material was sorted in more sorted products. Moreover, fine sorting residues were now differentiated from the crude sorting residues in the sub-model that describes the sorting of separately collected packaging materials.

<u>A.3.1 Modelling approach</u>

The calculations in this model were similar to the calculations already described for the previous model [Brouwer, et al. 2018]. Nevertheless, the change in collection system (from PCPPW collection to a combination of P, PD and PMD collection portfolios) resulted in some additional changes in the handling of the crude input data. Firstly, the compositional analyses were further categorised to enable merging of datasets, that would otherwise render comparison of the two models unfeasible. The percentage of the main components (Plastics, Metals, Beverage cartons and Residual waste) per sample were used to calculate the average composition of the collected materials and sorted products on this aggregated material level of main components. The average composition of packaging types within a main component was determined from the samples that were completely sorted, by multiplying the average percentage of the main components with the complete composition of main components in terms of packaging categories. Secondly, the expanded collection portfolios resulted in different types of packages in the collected materials and the sorted products, with different levels of attached moisture and dirt (LAMD). In this model the LAMD was equally subtracted from the gross amount of all packaging types and non-packaging objects, with the adjustment that for the plastics, beverage cartons and metals different average LAMD values were used, in order to deal with the substantial differences in LAMD between these main components. The amounts of materials and packages in the collected material flows was reconciled with the amounts of materials and packages in the sorted product flows using STAN software, which is further explained in paragraph A3.2. The resulting compositions are provided in the dataset [Brouwer et al., 2019b, Table D t/m I].

The net potential (amount of plastic packages at consumer households) of plastic packages originating from the Dutch households was calculated from the sum of the net amount of PCPPW in the collected materials that were accepted for sorting and the net amount of PCPPW present in the mixed MSW, as the amount of rejected LWP will most likely be recombined with mixed MSW and incinerated.

The subsequent mechanical recycling steps are described with transfer coefficients on the material level [Brouwer et al., 2018]. This modelling was done for the main plastic sorted products: PET, PE, PP, Film and Mix. The other (and new) sorted products were not further analysed, as the focus of the model is plastic packaging material recycling. Since there was no recycling company in 2017 able to recycle the sorted product "PET trays", this material was treated a stocked material for future recycling and hence it did not further contribute to the recycling system.

A.3.2 Data reconciliation with STAN software

The amounts of materials and packages in the collected material flows was reconciled with the amounts of materials and packages in the sorted product flows using STAN software [Cencic, 2012; Stan, 2018]. Minor manual adjustments to the data were made for packaging types that are not directly sold on the Dutch market, or were only found in very small amounts such as miscellaneous beverage bottles, miscellaneous non-beverage bottles, PS flexible packages > A4 and tubes for silicone kit.

The data reconciliation within STAN was performed without making use of the so-called "sub-good-layers" to keep the results comparable to the model of 2014. The consequence of this methodical choice is that the total weight of all the layers in the collected materials and sorted products doesn't precisely equal the weight of the "goods" (total gross weight of the collected and sorted products). Therefore, in the model the sum of all layers (reconciled packaging types, objects and moisture and dirt) were used as the total gross weight of the collected materials and sorted products.

A.4 Error analysis

In all instances other than the calculations for aggregated material level (i.e. for the total amount of Plastics, Metals, Beverage cartons and Residual waste), the standard deviation was derived by applying the basic principles of the error propagation law. For the aggregated numbers, the covariance (a measure of the strength of the correlation between two or more sets of random variates) was taken into account. Only in case of the EoL-fates the errors were so large in comparison to the average numbers that, these were treated as indicatively calculated results and were presented without errors.

The errors in the key performance indicators of this predictive model were calculated with the statistical theory of variance [Pollard, 2014]. In the specific case of the measurements available for this model, the distribution was proven to be normal (see normality tests below) and thus symmetric with respect to the average value (μ), which represents exactly the expected value. From the expected value, it is possible to derive a description of the variance, defined as the average of the difference between the actual values and the average or, in mathematical terms [Pollard, 2014]:

$$var(X)= \mathbb{E}\,[(X- \mu)^2] \qquad \text{(Eq. 1)}$$

In a normal distribution, the variance gives a measure of the distance of the values from the mean value. The square root of the variance represents the standard deviation.

In addition, given two variables, X and Y, each with sample size N, it is possible to define the covariance between these two variables [Pollard, 2014] as:

$$cov(X,Y)=E[(X-\mu X)(Y- \mu Y)] \qquad \text{(Eq. 2)}$$

In other terms, the covariance provides a measure of the strength of the correlation between two or more sets of random variates. If the variables are correlated – as it is the case of the compositional data used for this model - then their covariance will be non-zero and its value contributes to the calculation of the standard deviation. Only in case of the EoL-fates the errors were so large in comparison to the average numbers that, these were treated as indicatively calculated results and were presented without errors.

The major drawback of data reconciliation is that information on covariance after reconciliation is not retrievable anymore. As mentioned, covariance plays a significant role for aggregated numbers throughout the entire model for the calculation of all the performance indicators and, therefore, for the determination of the standard deviation of aggregated numbers only, variance-covariance matrixes were built up using the not reconciled data. The random variables that in Equation 2 are termed X and Y are represented by the 50 layers and the sample size N reflects the number of rough measurements resulting from the sorting analyses.

Different variance covariance matrixes were built with respect to the level considered. At packaging type level, one variance-covariance matrix was built for each collected material as well as for each sorted product. For the sorted products of fine residues, measurements from only one sorting analysis were available and a variance-covariance matrix could not have been implemented. These matrixes, allowed the calculation of the standard deviations of the following aggregated numbers: "net tonne plastic", "net tonne plastic packaging" and "tonne residual waste".

At the material level, a variance-covariance matrix was built for each of the ten materials composing each main plastic sorted product. This resulted in sixty variance-covariance matrixes which were used to assess the standard deviation of aggregated numbers for the origin of contaminants in sorted products, i.e. "components of targeted packages"; "components of non-targeted packages & objects"; "non-targeted packages & objects, outside the collection portfolio"; "non-targeted packages & objects, inside the collection portfolio".

In all instances other than the calculation of aggregated numbers, the standard deviation was derived by applying the basic principles of the error propagation law. The same procedure was followed for both sub-models.

A.5.1 Normality tests

A prerequisite for using STAN is that some of the data are normally distributed [Cencic, 2012]; the normal distribution of the measurements obtained from the sorting analyses was studied through a graphical method and an analytical method. Normality tests were made only on samples which size is bigger than 3.

Firstly, Quantile-Quantile (Q-Q) plots were drawn for the dominant packaging types within the collected products and the sorting products. Q-Q plots compares the quantiles of a variable's distribution and the quantiles of a given normal distribution verifying whether the distribution of a variable matches the given distribution. If this occurs, the points cluster around a straight line.

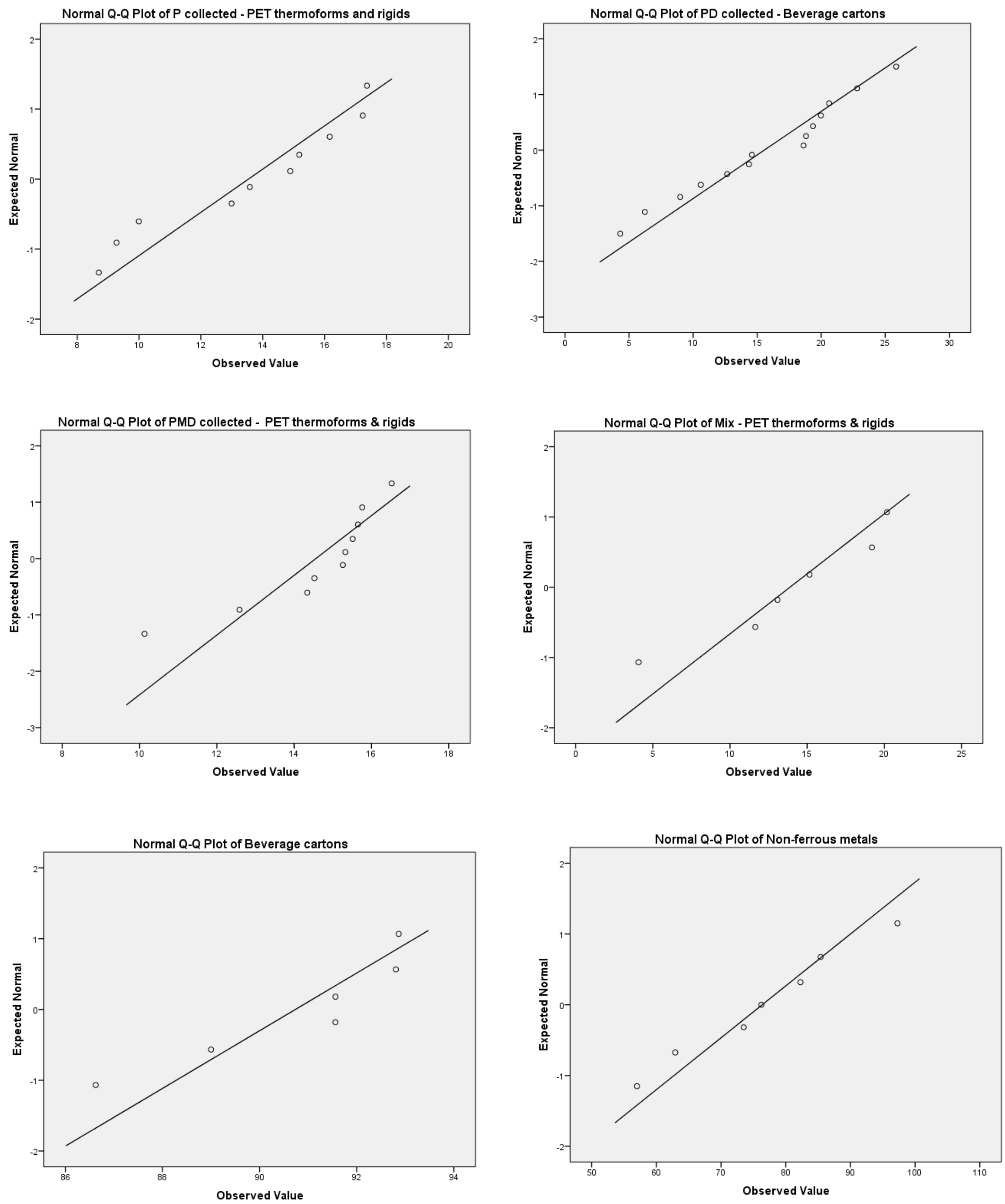

Figure A.1. Q-Q plots of data from separately collected and sorted PCPPW.

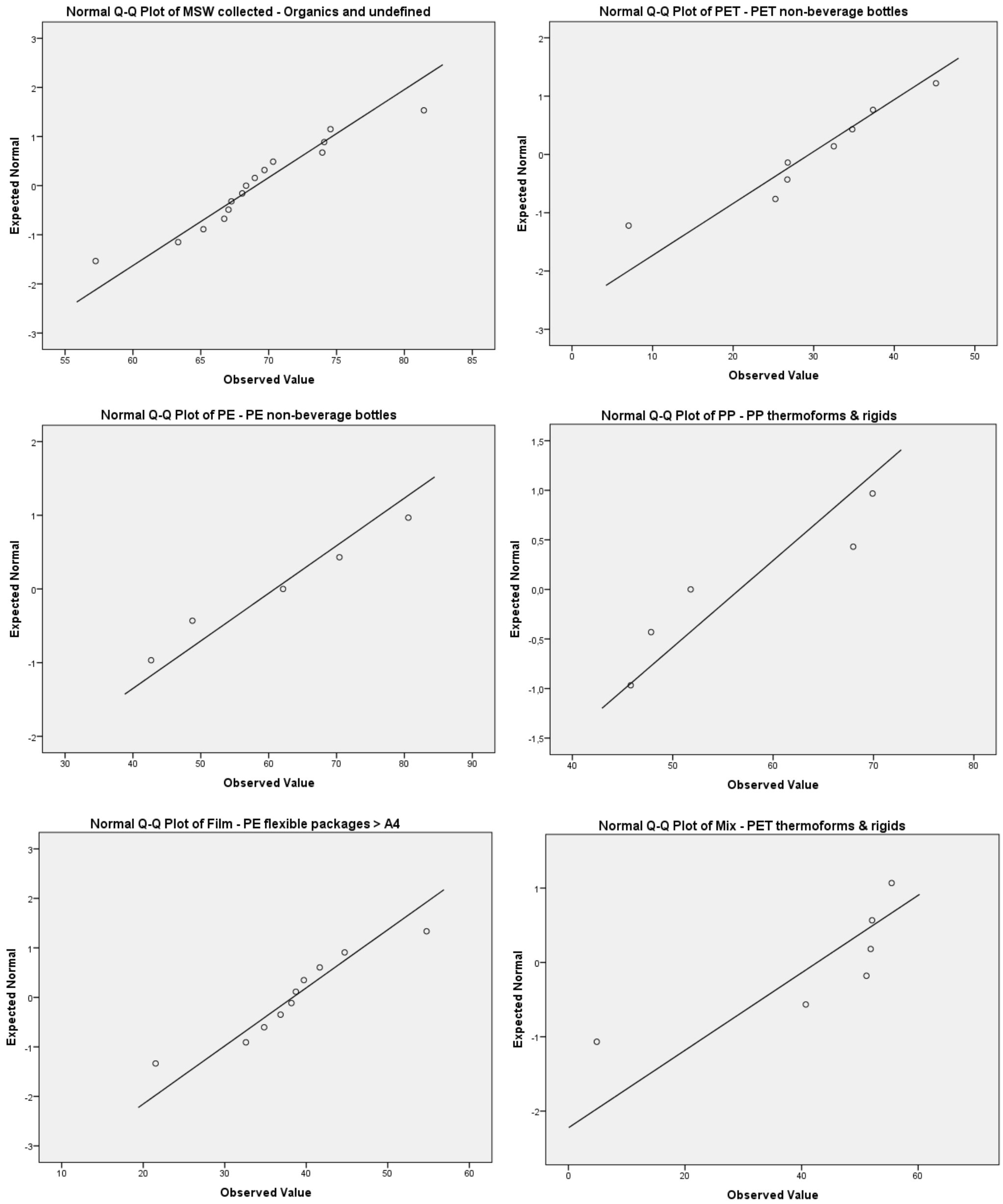

Figure A.2. Q-Q plots of data from MSW and mechanical recovered PCPPW.

Along with to the Q-Q plots, an additional test was performed to analytically verify the distribution of the data: the Kolmogorov-Smirnov (KS) test. A simple visual test is often helpful to have a rapid confirmation of the data distribution, however an analytical estimation is usually considered a tool that is more reliable than a subject visualisation of the plots.

The KS test works on the basis of statistical inference and was chosen over other normality tests because used on small sample sizes delivers better results [Lilliefors, 1967]. In the specific case of this model, the null hypothesis – being that the data are actually drawn from a normal distribution - is ***not*** to be rejected and the Lilliefors test – a modification of the KS test - is the goodness-of-fit test which also defines the test statistics D_n. Only if D_n is sufficiently large the null hypothesis can be rejected and this depends on the sample size, on the distribution form being fit and, above all, on the significance level of the test [Wilks, 2011]. In practice, considering a 95 confidence level, D_n is required to be located below the region defined from the quantile α = 0.05.

The KS test is applicable only if the parameters of the theoretical distribution (e.g. mean and variance) have not been estimated from the same data used to apply the test [Wilks, 2011]. For this reason, a modification of the KS test was operated by Lilliefors whose definition of critical values was used to test the available data [Lilliefors, 1967]. In Tables A.1 and A.2, results of KS normality tests made on the main packaging types for each sorted product are listed.

Table A.1. Test of normality for compositional data of separately collected PCPPW.

	Test statistic D_n	*D_n critical value**	*Degrees of freedom (sample size)*
	Collected packages - P		
PET thermoforms & rigids	0.163	0.258	10
	Collected packages - PD		
Beverage cartons	0.184	0.227	14
	Collected packages - PMD		
PET thermoforms & rigids	0.253	0.258	10
	Mix		
PET thermoforms & rigids	0.184	0.319	6
	Beverage cartons		
Beverage cartons	0.298	0.319	6
	Non-ferrous metals		
Non-ferrous metals	0.131	0.300	7
	Coarse residues		
Paper & cardboard	0.249	0.285	8

*data extracted from Lilliefors, 1967. Level of Significance: 0.05

Table A.2. Test of normality for compositional data of PCPPW recovered from mixed MSW.

	Test statistic D_n	*D_n critical value**	*Degrees of freedom (sample size)*
	Collected packages		
Organics & undefined	0.145	0.220	15
	PET		
PET non-beverage bottles	0.229	0.285	8
	PE		
PE non-beverage bottles	0.184	0.337	5
	PP		
PP thermoforms & rigids	0.265	0.337	5
	Film		
PE flexible packages > A4	0.150	0.213	10
	Mix		
PET thermoforms & rigids	**0.337**	**0.319**	6

*data extracted from Lilliefors, 1967. Level of Significance: 0.05

Results indicate that in no case the null hypothesis is rejected, meaning that it is correct to presume that the data is actually normally distributed. The only exception lies in the case of the sorted product MIX from recovered fractions from mixed MSW. However, it appears to still be feasible using STAN for data reconciliation; the requirements for the input data are clear in this sense: not the entire data set has to be normally distributed [Cencic, 2012].