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A case study on closed-loop recycling of co-polyester plates – Assessment of material quality and life-cycle energy and greenhouse gas performance

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A B S T R A C T

Material quality, and opportunities for multiple reprocessing, need to be considered when analysing the overall carbon footprint and energy efficiency of plastic products in life cycle assessments. This is rarely done today. This paper presents a case study evaluating a closed-loop recycling system involving a plastics manufacturer in Sweden which produces and reprocesses multiple-use plastic dining plates. The study involves (i) analysing the physical properties and food safety and (ii) assessing the life-cycle energy and greenhouse gas (GHG) performance of the closed-loop recycling system and three other conventional options. The results show certain deterioration in material quality of the plastic plates after six reprocessing cycles but maintained functionality and fulfilment of the food safety requirements. Furthermore, the results show that the life-cycle GHG emissions for the closed-loop recycling system correspond to 20–60% of those of the alternative systems. The primary energy use for the closed-loop recycling system amounts to 50–60% of that of two alternative systems, while it is higher compared to the system that involves one recycling loop followed by waste incineration with energy recovery. This study demonstrates the importance of taking material quality into account in life cycle assessments and confirms the GHG benefits of closed-loop systems.

1. Introduction

The need for a slower and circular flow of materials is widely recognised in the European Union (EU) and many other parts of the world. According to the European Commission's action plan for a circular economy, the value of products, materials and resources should be conserved in the economy for as long as possible, and the generation of waste should be minimised (European Commission, 2015). Concrete EU legislation in this area includes producer responsibility for certain products, such as end-of-life vehicles (2000/53/EC, 2000), waste electrical and electronical equipment (2003/96/EC, 2003), and packaging (94/62/EC). Despite the well-recognised need to promote reuse and circular material flows, the approach is still often waste-oriented (Rashid et al., 2013) and thus fails to take into consideration the preservation of the built-in material value of products, and limits the options for resource-efficient manufacturing, resulting in considerable value losses (Material Economics and ReSource, 2018).

Recent years have seen an increased focus on plastics due to its low circularity, but also due to plastic littering and its high fossil fuel dependence. In 2018 the EU launched its strategy for plastics in a circular economy. The strategy addresses the need to improve the economy and quality of plastics recycling, to reduce plastics littering and to promote innovation towards circular solutions. (European Commission, 2018). The volumes of plastics recycled remain very low considering that almost all consumer plastics are thermoplastics that could be recycled in several reprocessing cycles. Less than 30% of post-consumer plastic waste in the EU was collected for recycling in 2018 (European Commission, 2018). In Sweden, 42% of the plastic packaging that was placed on the market in 2016 was collected for recycling (Toniolo, 2012). In preparation for recycling following collection, losses are large and only part of the material is used in new, often simpler, applications (FTI AB, 2016). Furthermore, plastics recycling suffers from low demand for recycled feedstock according to, for example, the Ellen MacArthur Foundation (Ellen MacArthur Foundation, 2016) and industry organisations (Waste Sweden, 2017). Today, only 10% of the plastics produced are recycled and incorporated into new products (Ellen MacArthur Foundation, 2016), and only 8% of the value is preserved (Material Economics and ReSource, 2018).

One fundamental challenge to recycling is that most of the plastic waste stream is mixed with regard to the type of plastic and chemical

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composition (type of polymer and additives) of the components. Separating plastics for new quality products from such a mixture is a complex task, and value is lost due to the mixture of different types of plastics collected in open-loop recycling (Ellen MacArthur Foundation, 2016). Most plastics that are recycled today consequently become low-value products such as noise barriers, buckets or decking (FTI, 2018; Allerup and Fråne, 2016). However, when they are kept separate and recycled in closed-loop systems, value is better preserved and the recycling rate is considerably higher (Swedish EPA, 2018). A case in point is the deposit-return system for polyethylene terephthalate (PET) beverage bottles in Sweden for which the capture rate in collection was 83% in 2016 (Allerup and Fråne, 2016).

By preserving the material quality of the waste stream, closed-loop recycling systems facilitate recycling in not only one but multiple cycles. Several studies show that recycling of plastics into new plastic products saves resources and energy and reduces greenhouse gas (GHG) emissions compared to using virgin fossil feedstock (see e.g. (Craighill and Powell, 1996; Björklund and Finnveden, 2005; Gu et al., 2017)). Existing life-cycle assessments (LCAs) of plastics recycling mainly compare recycling in only one cycle to other waste handling options, without considering the quality impact on the material and thus opportunity for recycling in multiple cycles. Differences in material quality between closed- and open-loop recycling have been pointed out in some recent LCAs (see e.g. (Eriksen et al., 2018)). These studies do not, however, analyse the consequences of these differences on the environmental performance of the different systems. This paper aims to fill this research gap by analysing and comparing the environmental performance of a closed-loop recycling system with other recycling systems.

In this case study, we evaluate a closed-loop recycling system where plastics are recycled into new, high-quality products in a multiple reprocessing system. The case study involves a small plastics manufacturer in Sweden which produces and reprocesses multiple-use plastic dining plates in closed-loop reprocessing cycles. This plastics products manufacturer is a pioneer in improving sustainability in plastic production systems. The plastic plates are used in public canteens where plastic plates are often preferred to glass and porcelain plates for work environment reasons related to lower weight and less noise. The plastic consists of a co-polyester (PCTG) which resembles the polyester polyethylene terephthalate (PET) that is commonly used in packaging and as a fibre (Swedish National Consortium for Emissions Data (SMED), 2012). Today, the plastic plates run in a linear material flow in which they are used for approximently three to five years, washed daily, and thrown away when they become worn out. The purpose of this study is to evaluate a closed-loop recycling system with regard to the reprocessability of the plastics and the energy savings and carbon footprint of the closed-loop reprocessing system. This is done in two steps by 1) analysing the reprocessed plates with regard to physical properties and food safety and 2) assessing the energy and GHG performance of the closed-loop multiple reprocessing system from a life cycle perspective compared to other systems that involve conventional waste handling options.

To the best of our knowledge, no previous LCAs have been carried out on systems with multiple reprocessing cycles, while ensuring that material quality is preserved. Thus, the originality of this study is the LCA of recycled high-value plastic products combined with analyses of physical properties and food safety in order to confirm that material quality is preserved within the defined life cycle systems boundaries. This LCA focuses on GHG footprint and energy use.

2. Method

2.1. Reprocessability of the PCTG plastics

The reprocessability of the plastic plates was investigated by analysing the physical properties of the reprocessed plates and food safety. These analyses were carried out for seven reprocessing cycles. The plastic considered in this case study is the glycol-modified co-polyester poly-cyclohexylenedimethylene terephthalate (PCTG) (brand name Ecozen), consisting of the monomers ethylene glycol, terephthalate, cyclohexane-di-methanol and isosorbide. The melting point of PCTG is 240 °C and the glass temperature (the temperature at which the mobility of the molecules in the plastic increases) is specified by the manufacturer at 90 °C.

The mechanical properties (impact resistance and tensile tests) and the thermal property (glass transition temperature) of the reprocessed material were studied by analysing samples after every reprocessing cycle. The impact resistance tests were performed using Ceest impact testing equipment according to ISO 179 and the Charpy method. The tensile tests were performed in a Zwick Z1 universal testing machine and according to ISO 527. Differential scanning calorimetry (DSC) with a Mettler DSC7, containing STARe software, was used to investigate the thermal property. Before the first, second and third reprocessing cycles, ageing of the plates was performed by simulating daily dishwashing for three years of use. For this purpose, the plates were kept in a water bath with dish-washing detergent at a temperature of 62–68 °C for 12 h. The plates were then rinsed in water at a temperature of 85 °C. For more detailed information on the testing methods, see Supplementary Material.

Food safety was also studied for seven reprocessing cycles, with ageing simulated, as described above, between each reprocessing cycle. The overall migration to food was analysed from plates after the 3rd, 5th and 7th reprocessing cycles, by so-called “filling” or by total immersion in two different food simulants according to the standards EN 1186–3:2002, EN 1186–9:2002 and EN 1186–14:2002. All tests were performed as standardised for products for repeated use with three exposures and triple tests from each exposure. Characterisation and quantification of migrating molecules in the simulants were performed using liquid chromatography/mass spectrometry (LC/MS) and gas chromatography/mass spectrometry (GC/MS). The amount of migrating molecules was calculated by semi-quantification against an internal standard and identified utilising the NIST reference library (NIST, version 2.2, 2014). The simulants were also analysed for acetaldehyde,[1] metals and primary aromatic amines (PAA). For more detailed information on the testing methods see Supplementary Material.

2.2. LCA method and assumptions

LCA was used to analyse the energy performance and GHG emissions of the studied closed-loop recycling system for plastic plates as well as for three reference systems. The timeframe of this study was set to 18 years based on an estimated three-year lifespan of the individual plates in combination with the results from the material tests which show that the plates maintain functionality after 6–7 reprocessing cycles despite certain deterioration of physical properties (see section 3.1). Assuming six reprocessing cycles and a three-year lifespan gives a timeframe of 18 years. The functional unit (FU) in this study was set to “fulfilling the need for 100 dining plates every day for 18 years”. The final disposal of the circulated material after 18 years was also included.

LCAs were carried out for a system where PCTG plates are reprocessed every three years using closed-loop recycling (base case) and for three reference systems representing more common-practice waste handling alternatives. The reference systems involve: 1) PCTG plates recycled with conventional open-loop recycling to provide a different short-lived product, 2) PCTG plates recycled with conventional open-loop recycling to provide a different long-lived product, 3) PCTG plates in a “use and discard” system including waste incineration.

The choice of the parameters energy consumption and GHG emissions in the LCA is primarily motivated by the availability of life cycle inventory (LCI) data, which is currently more restricted regarding other

[1] From the first tests with ageing in the first three reproduction cycles.

Table 1
The amount of material required/consumed in the base case system and the three reference systems to fulfil the functional unit.

	Base case PCTG (closed-loop)	System 1 PCTG (open-loop to short-lived product)	System 2 PCTG (open-loop to long-lived product)	System 3 PCTG (use-and-discard)
No. of manufactured plates	600	600	600	600
Manufactured from virgin raw material (pieces)	100	600	600	600
Weight of each plate (kg)	0.134	0.134	0.134	0.134
Amount of virgin raw material consumed (kg)	13.4	80.4	80.4	80.4
Material to waste incineration (kg)	13.4	80.4	–	80.4
Material deposited at landfill (kg)	-	-	-	-
Material stored in new long-lived product (kg)			80.4	

environmental impact categories. As more comprehensive LCI data become available, other critical and relevant impact categories should preferably be considered, providing a more comprehensive assessment. However, we argue that the two parameters included here, energy and GHG performance, are critical in the development of a sustainable circular economy where both energy and GHG savings need to be achieved.

The whole life cycle of the plastic products was assessed including all the steps from raw material extraction to final waste handling, according to ISO 140 44 (ISO 140 44, 2006). All energy use in the calculations was specified as primary energy. When calculating the consumption of electricity in processes, primary energy factors of 1.7, characteristic of Swedish conditions (Gode et al., 2011), and 2.46, characteristic of average European conditions (International Institute for Sustainability Analyses and Strategy, 2015), were used. These primary energy factors were included in the calculation to reflect that energy losses in the generation and transmission of the electricity vary between geographical regions due to different electricity production mixes. Emissions of GHGs were calculated as global warming potential (GWP) and expressed as CO_2 equivalents (CO_2-eq), in a 100-year perspective (IPCC, Buendia et al., 2006) (see also Table 1).

All stages in the production of the studied products were covered by conventional technology. The technical systems boundaries are illustrated in Fig. 1 and include extraction of raw material, production of feedstock, manufacturing of plates/plastics products, transportation of raw material and recycled material and transportation of product to the user. All plates were assumed to have an average lifetime of three years, before being recycled or replaced.

Waste handling was also included. In the base case, it was assumed that the dining plates were incinerated after the sixth use phase, after 18 years of use. In the reference systems, it was assumed that the plates were incinerated after being recycled and used, or straight after use. The temporal system boundary was thus set to 18 years, as described previously.

Globally, the value chain of plastics production consists of a few major primary producers of feedstock and many small product manufacturers. In this study, all primary plastic material production was assumed to be based on average European data, while product manufacturing, use and recycling were assumed to be located in Sweden, illustrating the geographical systems boundaries. These assumptions were made to eliminate the impact on the results arising from differences in transportation distances or in power generation systems, since the main interest of the LCA part of this case study is to analyse the differences between closed-loop and open-loop recycling systems. Transportation from the primary producer to the product manufacturer (cargo ship), and from the product manufacturer to the user (truck) was included in the study through an equivalent estimated distance and best-estimate transport measure (Fig. 1).

The daily use of the plates in the canteen, including washing, was assumed to be the same in all systems, and was therefore excluded from the assessment. Transportation to an incineration plant was assumed to be over a short distance, resulting in negligible energy consumption and GHG emissions, and was thus excluded. The amount of industrial waste from manufacturing was assumed to be the same in all systems and was therefore also excluded.

2.2.1. *Description of base case and reference systems*

The PCTG plates are produced in two steps which include production of the co-polyester resin, and injection moulding at the plate manufacturer. The plates are manufactured in an extruder working at a locking pressure of 160–200 tonnes at a temperature of 240 °C. The recycling system investigated involves the processing of plates from virgin plastic granules and use in six use phases with reprocessing of the material before every new use phase. In the base case, the material is recycled back to the plate manufacturer, ground, dried and injection moulded to produce new plates. The need to replace the plates every three years means that 600 plates will be manufactured during the 18-year period. At the end of the 18-year study period, the plates are incinerated. The amount of virgin raw material required is the material needed to produce 100 plates. The following reprocessing (5 x 100 plates) is based on this re-used raw material. As the weight of each plate is 0.134 kg, 13.4 kg virgin raw material is needed, assuming negligible losses in reprocessing.

In Systems 1 and 2, the plates are recycled with other plastics waste and feedstock after three years of use. In System 1, the discarded plates are used to manufacture a short-lived, lower-value product, such as a plastic bag or a packaging material, which is incinerated after use. In System 2, the discarded plates are used to manufacture a long-lived product, such as storm-water pipes or composite building materials. This product is expected to last several decades and is not incinerated within the timeframe of this study (18 years). In Systems 1 and 2, the plastics products are likely to be sent straight to recycling, without passing a mechanical sorting plant. Expansion of systems boundaries was applied in these reference systems, including substitution of alternative short-lived and long-lived products, respectively. In System 3, it was assumed that the PCTG plates were incinerated, with energy recovery, directly after being used for three years, replacing alternative fuels for district heat production (see Section 2.2.3). In Systems 1–3, new PCTG plates are produced from virgin raw material, and the amount of virgin raw material required for the 600 plates in each system totals 80.4 kg. The material consumption/requirement in each of the compared systems to fulfil the FU is summarised in Table 1.

2.2.2. *Calculations of energy use and GHG emissions*

The energy use and total GHG emissions for each system were calculated using the following equation (1). The data used in these calculations are described in Section 2.2.3.

$$E_i = \sum e_i p_i c_i \tag{1}$$

where:

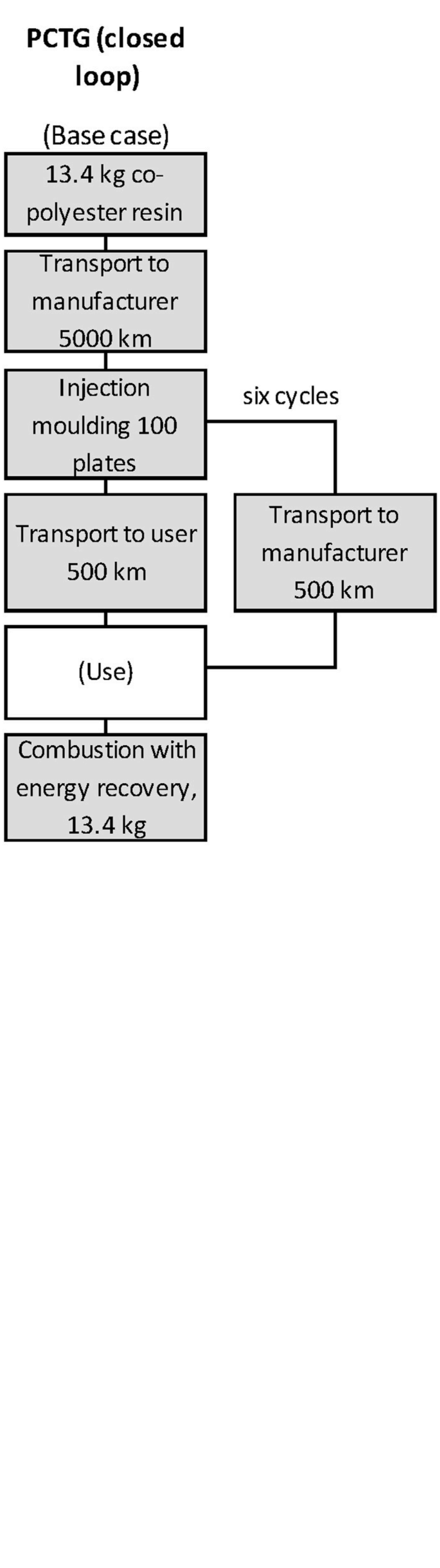

Fig. 1. The process tree of the four systems studied, with the base case on the left.

Table 2
Description of the characteristics of the life cycle inventory data utilised in the study.[a].

References utilised for conducting best estimate input data	Year	Geographical scope	Impact assessment method	Origin
(Plastics Europe, 2015a, 2015b, 2015c)	2015	Europe	*GWP 100*	Industry
(Wall, CEO Mälarplast, 2018)	2018	Sweden	*GWP 100*	Plant specific
(Swedish National Consortium for Emissions Data (SMED), 2010)	2010	Sweden	*GWP 100*	Industry
(Lantz and Börjesson, 2014)	2014	Sweden	*GWP 100*	Industry
(Swedish Energy Agency, 2018)	2018	Sweden	*GWP 100*	Industry
(Network for Transport measures (NTM), 2018)	2018	Europe	*CO_2-eq*	Industry
(IPCC, Buendia et al., 2006)	2006	Sweden	*GWP 100*	Plant specific

[a] It has not been possible to evaluate the allocation rules used in the input data described in this table due to limited description in the references, except for two references using mass allocation (ref. (Lantz and Börjesson, 2014) and (Swedish Energy Agency, 2018)).

E_i Result for the functional unit throughout the process tree of data category i
e_i Emission factor for a unit process (kg/kg) per unit energy consumed (MJ/kg) for data category i
p_i Weight of one plate (kg)
c_i Consumption of plates (n)
i Data categories calculated are: kilograms of CO_2-eq and use of primary energy, in MJ

2.2.3. Data collection

LCI data were collected from scientific publications, reports, industrial databases and interviews, with a preference for recent plant-specific data related to the processes in question. For transparency, the characteristics of the data utilised are described in Table 2. If no data could be found, assumptions and analogues to similar processes or materials were used. The data used in the calculations are European or Swedish, from no earlier than 2012, unless otherwise stated. We consider the quality of the data used for the recycling scenarios to be generally high, as modern industrial or plant-specific data were available throughout the systems.

The co-polyester PCTG is a recently developed material, and the only known data for PCTG are thus from an LCA provided by the producer SK Chemical (SK Chemicals, 2013). They quantify the GHG emissions for PCTG in relation to the emissions for the production of polystyrene (PS), polymethyl methacrylate (PMMA) and acrylonitrile butadiene styrene (ABS) resin. Based on the SK chemical comparison study, the eco-profiles for PS, PMMA and ABS from PlasticsEurope (Plastics Europe, 2015a, 2015b, 2015c) were used to estimate the GHG emissions from, and the consumption of primary energy in, the production of PCTG resin. The data used in the calculations are presented in Table 3, and a full description of data collection and use is presented in the Supplementary Material.

Table 3
Used data regarding production, manufacturing and recycling (Chemicals, 2013; Plastics Europe, 2015a; Plastics Europe, 2015b; Plastics Europe, 2015c; WallCEO Mälarplast, 2018; Swedish National Consortium for Emissions Data (SMED), 2010).

Activity	GHG emissions (*kg CO_2-eq/kg*)	Energy use (*MJ/kg*)
PCTG resin production	2.37	41.1
PCTG plate manufacturing		
drying and extruding	0.124	16.1
grinding[a]	0.128	16.5
Conventional recycling		
washing, grinding, re-granulation	6.6	3.1
extruding new product	0.124	16.1

[a] Only in reprocessing.

The primary energy consumption and GHG emissions for the manufacturing of the PCTG plates were calculated based on the installed power of the dryer, extruder (used in drying and injection moulding of PCTG), and grinder, together with the time of use. The installed power and time of use (WallCEO Mälarplast, 2018) are presented in full in the Supplementary Material.

The heat value and emission factor for plastic waste were assumed to be 30 MJ/kg plastic and 70 g CO_2-eq/MJ (Swedish National Consortium for Emissions Data (SMED), 2010). This gives an emission of 2.1 kg CO_2-eq/kg combusted plastic. PCTG is 10% carbon neutral by weight since one of the monomers is bio-based, leading to a reduction of 10% in GHG emissions from combustion.

The alternative fuel replaced by waste incineration with energy recovery was assumed to be forest residues. This is based on Swedish conditions since wood fuels, in form of residues from forestry or forest industry byproducts, are the dominant fuel in the district heating sector (Lantz and Börjesson, 2014). The life-cycle GHG emissions for forest residues (primarily logging residues from final felling) are assumed to be 2.2 g CO_2-eq/MJ (Swedish Energy Agency, 2018).

The energy consumption of a general cargo ship was assumed to be 0.159 MJ/tonne per km and the GHG emissions 12.4 g CO_2-eq/ton per km (Network for Transport measures and NTM, 2018). The energy consumption in transportation by truck was assumed to be 1.90 MJ/tonne per km, and the GHG emissions were assumed to be 0.130 kg CO_2-eq/ton per km (Network for Transport measures and NTM, 2018).

2.3 Sensitivity analysisThis study include a sensitivity analyses to evaluate the rubustness of the GHG performance. The sensitivity analyses include the following three parameters: 1) geographical system boundary, 2) transport distance and 3) number of reprocessing cycles.

3. Results

3.1. Reprocessability of the PCTG plastics

The results of the tests of the mechanical and thermal properties are presented in Table 4. The results show that impact resistance was affected by the reprocessing. After the third reprocessing, the material became somewhat harder and displayed a significant reduction in impact resistance, from about 150 J to about 50 J (Joule). In the last two

Table 4
Results of the measurements of the impact resistance, tensile parameters and glass temperature in the virgin material and in the material after 1–7 reprocessing cycles. Glass temperature was analysed for the virgin material and after the third and seventh reprocessing cycles.

Sample	Impact resistance	Tensile parameters		Glass temperature	
	Break energy (J/mm^2)	Max force (MPa)	Elongation at max force (%)	Onset (°C)	Mid-point (°C)
K0	178	51.2	5.9	85.4	87
K1	151	51.3	6.0		
K2	90.5	53.2	5.9		
K3	146	48.1	5.3	83.9	85.7
K4	51.0	53.0	5.9		
K5	48.7	53.1	5.8		
K6	1.60	53.3	5.8		
K7	1.77	47.0	4.7	83.1	85.2

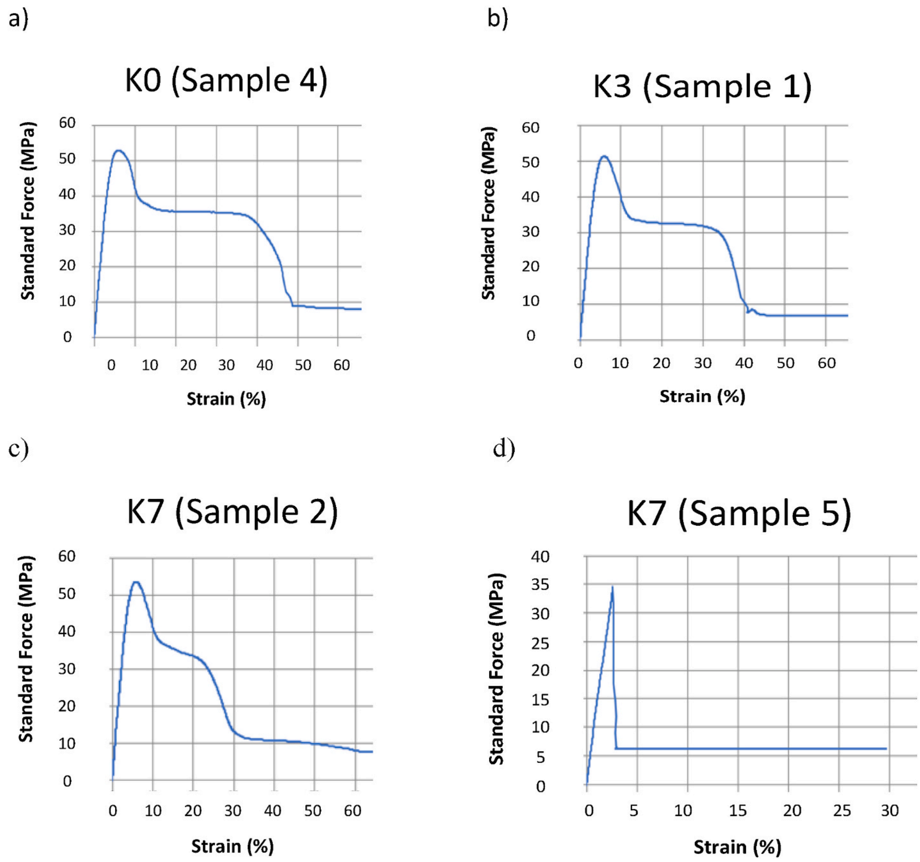

Fig. 2. Tensile graph for one of the samples from a) virgin material (K0), b) third reprocessing cycle (K3), c) and d) seventh reprocessing cycle (K7).

reprocessing cycles there is a further significant reduction in impact resistance. Glass temperature, analysed as spot-checks for the virgin material and after the third and seventh reprocessing cycle, decreased from 87 °C (K0) to 85.2 °C (K7). This means that the plates can still be dishwashed at a conventional temperature of 60–65 °C.

The tensile tests show no major changes at maximum force and elongation at maximum force with the exception for the last reprocessing cycle. This is illustrated in Fig. 2, where the slow break with a clear yield point (the flat mid-area of the curve) regarding virgin material is clearly visible in a) (K0). After reprocessing, this yield point became smaller, as seen in b), and the samples were more brittle (K3). After the seventh reprocessing cycle (K7), only two samples of the five tested showed an indication of a yield point as in c) and most samples showed more brittle fracture as in d).

The character of the changes observed in the reprocessed material is to be expected, and they correspond to those that have been observed for reprocessed PET (López et al., 2014; Venkatachalam et al., 2012). However, this co-polyester seems to maintain its properties better through the reprocessing cycles. Most importantly, the observed changes do not affect the usability of the plates after seven reprocessing cycles. In addition, the processability of the plastic feedstock in the extruder is retained throughout the seven reprocessing cycles.

Table 5
Overall migration measured after repeated use in the simulants 95% ethanol and 3% acetic acid in accordance with the standards EN 1186–3:2002, EN 1186–9:2002 and EN 1186–14:2002. The result is expressed as mean value from triple tests from three exposures.

	Mean results from triple tests of three exposures	
	95% ethanol *0.5 h, 40°C*	3% acetic acid *2 h, 70°C*
Reprocessing cycle 3	<2 mg/dm^2	<2 mg/dm^2
Reprocessing cycle 5	<2 mg/dm^2	<2 mg/dm^2
Reprocessing cycle 7	3.0 mg/dm^2	2.4 mg/dm^2

The total migration of substances detected from the plastic plates is shown in Table 5 and is below the statutory limit The overall migration limit permitted under EU rules is 10 mg/dm^2 (Commission Regulation 10/2011/EC). No acetaldehyde, metals or polycyclic aromatic hydrocarbons (PAH) were found to migrate to the simulants. However, the fact that some migration was detected led to an additional reprocessing procedure where the samples of plastics for migration analyses were not cut before testing. In this case, no overall migration was detected. Hence, it seems likely that the overall migration detected in the previous tests (shown in Table 5) was caused by very small fragments of the plastics leaving the cut due to the increased brittleness and not due to migrating molecules.

The material quality tests and their results presented here were described in an application to the European Food Safety Authority (EFSA), which has to approve the production of recycled plastic materials and articles intended to come into contact with foods (Commission Regulation (EC) No 282/2008). The application also described a quality management system for ensuring maintained material quality in the loop. EFSA recently approved the closed-loop recycling system presented in this case study which can now be implemented (European Food Safety Authority (EFSA), 2021).

3.2. Life cycle GHG emissions and energy performance

Multiple reprocessing of the PCTG material by closed-loop recycling led to significantly lower GHG emissions and less energy consumption per FU than in the three reference systems studied, as can be seen in Fig. 3. The GHG emissions from the base case are approximately 25% of those in System 1 (conventional open-loop recycling to a short-lived

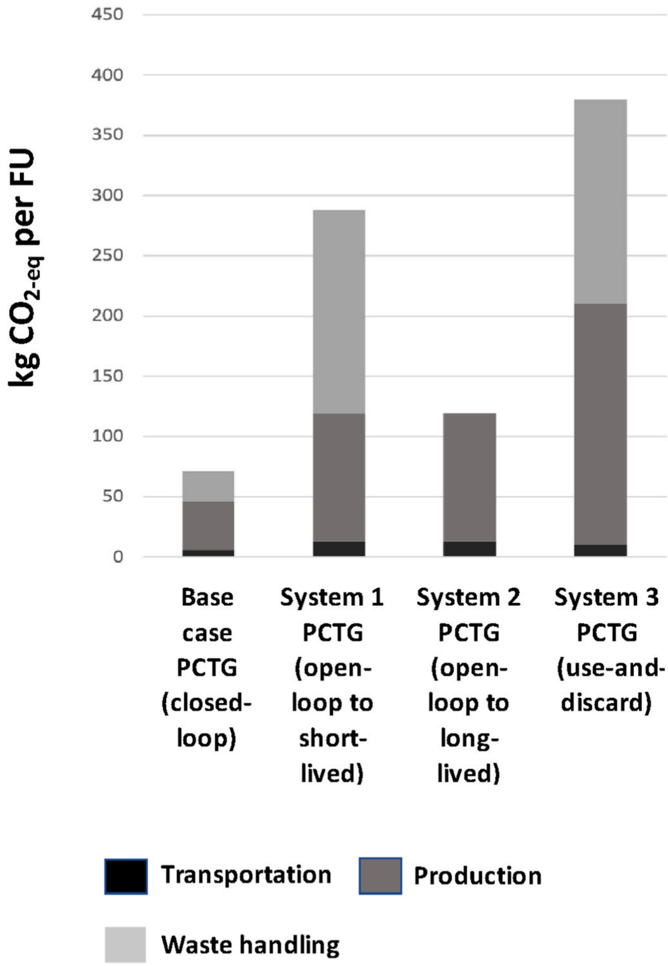

Fig. 3. GHG emissions (kg CO_2-eq/FU) from the base case and the three reference systems included for comparison. The FU in this study was “fulfilling the need for 100 dining plates every day for 18 years”.

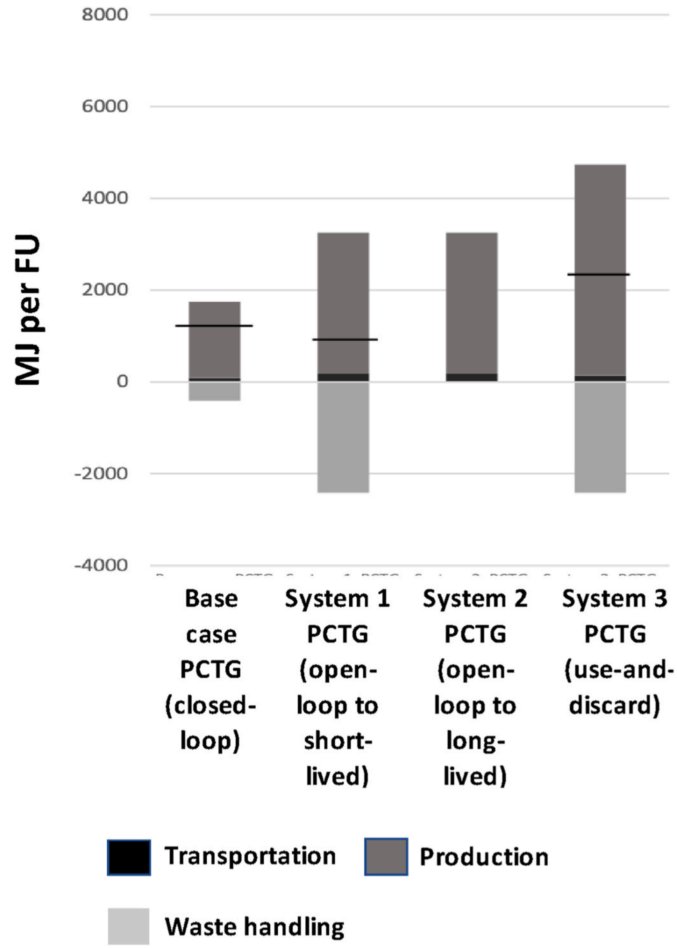

Fig. 4. Primary energy consumption (MJ/FU) in the base case and the three reference systems. The energy recovered (district heat) from combustion of the discarded PCTG in waste incineration plants is illustrated by the negative bars. The horizontal lines indicate the potential net energy consumption in each system. The FU in this study was “fulfilling the need for 100 dining plates every day for 18 years”.

product), and 61% of the emissions in System 2 (conventional open-loop recycling to a long-lived product). The difference between the two conventional recycling options is caused by the emissions from incineration of the short-lived product. The GHG emissions of the base case amount to 19% of those in System 3 in which the PCTG plates are not recycled.

The primary energy consumption is also lower with closed-loop recycling (Fig. 4). The primary energy consumption in the base case is 54% of the energy consumption in Systems 1 and 2 with conventional recycling. Compared to System 3 (a recycling), energy consumption is 37%. If the energy from combustion can be utilised, it is relevant to compare the net energy consumption. The net energy consumption of the base case was found to be 160% of the net energy consumption in System 1 (recycling to a short-lived product) since this system contributes more (fossil) energy by waste incineration. The net energy consumption of the base case is 41% of that for System 2, which involves recycling to a long-lived product where the product is not incinerated within the timeframe of this study. The net energy consumption of the base case is 58% of that for System 3. The net energy consumption of each system is indicated by the horizontal lines in Fig. 4. The results are also presented in Tables S1 and S2 in the Supplementary Material.

3.3. Sensitivity analyses of life-cycle GHG performance

As described in section 2.3 sensitivity analyses were done including geographical system boundary, transport distance and number of reprocessing cycles. The results for GHG emissions are presented in Fig. 5. The results of the sensitivity analyses are also presented in Table S3 in the Supplementary Material.

Product manufacturing uses electricity, and the GHG emissions from the production of electricity in Sweden are lower than the European average (European Environment Agency (EEA), 2020). Furthermore, the primary energy factor for Swedish average electricity production is lower than the European average. To understand the impact of the location of reprocessing in Sweden, the geographical system boundary was changed so that both production and manufacturing are based on European average data. As seen in Fig. 5, the change in geographical system boundary led to higher GHG emissions but did not change the overall results regarding the ranking of the systems.

The base case has a shorter total transport distance than the reference systems since the raw material in most cases is derived from the user (recirculated material) instead of from a resin producer (virgin material) which requires long-distance ship transport. To analyse the impact of this advantage for the base case, the long-distance ship transportation of the virgin feedstock was replaced by truck transport over a distance equal to that between the manufacturer and the user. This change had only a marginal effect on the GHG emissions in all systems and did not change the overall results (Fig. 5).

The number of reprocessing cycles and, consequently, the time frame of the functional unit, were reduced to investigate whether the number

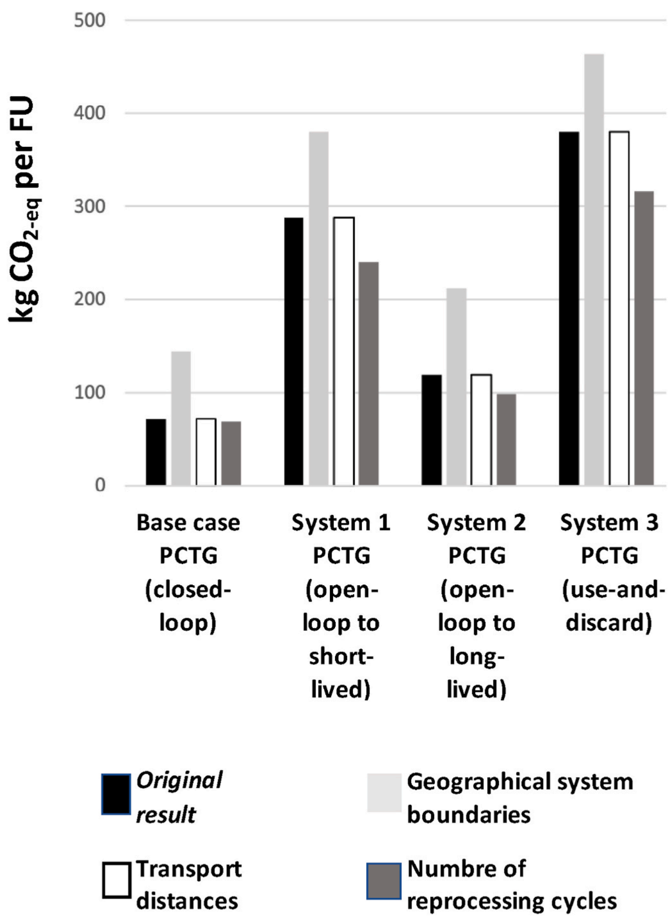

Fig. 5. Results of the sensitivity analysis after changing the geographical system boundaries, transportation distances and number of reprocessing cycles, in terms of the emission of GHGs. (Note that in the variation Number of reprocessing cycles the FU is changed in relation to the other variations to 5 reprocessing cycles covering 15 years).

of reprocessing cycles in the base case had an effect on the overall results. Calculations were performed with five cycles, leading to a temporal system boundary of 15 years. This change led to lower GHG for all systems, but the overall results remained the same.

The overall conclusion drawn from the sensitivity analyses was that none of the variations investigated changed the overall results regarding the ranking of the systems with regard to GHG emissions and energy consumption. In all cases, the closed-loop recycling of PCTG dining plates showed the best GHG performance.

4. Concluding discussion

This case study evaluates a closed-loop recycling system where plastic dining plates are reprocessed six times. The evaluation was carried out with regard to material quality and life-cycle GHG emissions and energy performance. The material tests show limited deterioration of the mechanical and thermal properties of the reprocessed material and that fully functional plastic plates that fulfil food safety standards can be produced after at least six reprocessing cycles. Furthermore, the results show that the life-cycle GHG emissions for the closed-loop recycling system correspond to 20–60% of those of the alternative systems. The primary energy use for the closed-loop recycling system amounts to 50–60% of that of two alternative systems. These findings were shown to be robust when varying critical parameters in the sensitivity analyses.

The study clearly demonstrates the PCTGs ability to be recycled time after time and the climate benefits of a closed-loop recycling system of plastics that enable multiple reprocessing cycles. It should nevertheless be noted that the specific results will differ for other closed-loop recycling systems for plastics. This case study involves six reprocessing cycles of PCTG, a PET modified to withstand heat and scratching better than PET. The number of reprocessing cycles that is possible for other types of plastics and plastic products has to be determined based on their specific properties and quality requirements. A limitation of the LCA study is that it only addresses GHG emissions and energy efficiency. Partly depending on the type of plastic, it could also be relevant to study other environmental impacts such as toxicity. A difficulty in this regard, and for LCAs in general, is that the life cycle environmental performance of many new plastic materials with specific characteristics are often poorly documented, especially those produced outside Europe.

The results of this case study are of interest to several stakeholders such as "early birds" in the plastics manufacturing industry which involves many local and regional producers. These producers may be suitable pioneers in the development of circular material flows since the local/regional structure may facilitate material circulation between producer and user. This case study shows that recycled plastics can meet food safety standards (and obtain EFSA approval) as long as the quality of the recycled material is kept under control. This is something that could inspire development of closed-loop recycling systems of plastics in food applications and transform the waste-management perspective of today's plastics recycling into a resource-management perspective, ensuring the maximum possible use of each kind of plastic.

The results of this study are also relevant for LCA practitioners. To fully exploit the potential of LCA as a tool for analysing processes in a circular economy, the quality of recycled materials and potential for use in multiple reprocessing cycles should be taken into account. We therefore recommend users of the LCA methodology within circular material systems to consider whether the material is circulated in a way that preserves the material quality and allows multiple lives with equivalent functions. This may also imply adjustments of the systems boundaries, including the temporal system boundaries, which must be expanded to incorporate the expected future applications of the material.

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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 data

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References

2000/53/EC, European Legislation on EPR for End of Life Vehicle (ELV).
2003/96/EC, European Legislation on EPR for Waste Electrical and Electronic Equipment (WEEE).
94/62/EC 94/62/EC, European Legislation on EPR for Packaging.
Allerup, J., Fråne, A., 2016. Sveriges återvinning av förpackningar och tidningar - uppföljning av producentansvar för förpackningar och tidningar 2014. In: Report from Svenska MiljöEmissionsData, Naturvårdsverket 2016.
Återvinningsstatistik, F.T.I.A.B.. Förpacknings- och Tidningsinsamlingen (Only available in Swedish). http://www.ftiab.se/180.html. (Accessed 2 April 2018), 2016.
Björklund, A., Finnveden, G., 2005. Recycling revisited—life cycle comparisons of global warming impact and total energy use of waste management strategies. Resources. Conserv. Recycl. 44, 309–317.
Chemicals, S.K., 2013. Life cycle assessment – eco friendly materials: ECOZEN Co-polyester. SK chemicals, biesterfeld. Ferdinandstrasse 41 (Hamburg, Germany).
Craighill, A.L., Powell, J.C., 1996. Lifecycle assessment and economic evaluation of recycling, a case study. Resources. Conserv. Recycl. 17 (2), 75–96.
Ellen MacArthur Foundation, 2016. The new plastics economy - rethinking the future of plastics. https://www.ellenmacarthurfoundation.org/publications/the-new-plastics-economy-rethinking-the-future-of-plastics. (Accessed 30 October 2018).
Eriksen, M., Damgaard, A., Boldrin, A., Fruergaard, T.H., Astrup, T., 2018. Quality assessment and circularity potential of recovery systems for household plastic waste. J. Ind. Ecol. 23 (1), 156–169.
European Commission, 2015. "Closing the Loop - an EU Action Plan for the Circular Economy,". COM, p. 614 (final).
European Commission, 2018. A European Strategy for Plastics in a Circular Economy. COM, p. 28, 2018.
European Environment Agency (EEA). https://www.eea.europa.eu/data-and-maps/figures/co2-electricity-g-per-kwh. (Accessed 3 January 2020).
European Food Safety Authority (EFSA), 2021. Safety assessment of the process Green Loop System, used to recycle polycyclohexylene dimethylene terephthalate glycol-modified (PCTG) plates for use as food contact materials. Scientific opinion. EJ EFSA J. https://doi.org/10.2903/j.efsa.2022.7002.
FTI, A.B., 2018. Återvinningsprocessen. Förpacknings och Tidningsinsamlingen. http://www.ftiab.se/182.html. (Accessed 2 April 2018).
Gode, J., Martinsson, F., Hagberg, L., Oman, A., Hoglund, J., Palm, D., 2011. Environmental facts 2011-Estimated emissions from fuel, electricity, heat and transportation in Stockholm. Varmeforsk.
Gu, F., Guo, J., Zhang, W., Summers, P.A., Hall, P., 2017. From waste plastics to industrial raw materials: a life cycle assessment of mechanical plastics recycling practice based on a real-world case study. Sci. Total Environ. 601–602, 1192–1207.
International Institute for Sustainability Analyses and Strategy, 2015. Development of the primary energy factor of electricity generation in the EU-28 from 2010-2013. https://www.researchgate.net/publication/273636570_Development_of_the_Primary_Energy_Factor_of_Electricity_Generation_in_the_EU-28_from_2010-2013. (Accessed 30 April 2019).
IPCC, 2006. IPCC guidelines for national greenhouse gas inventories. Prepared by the national greenhouse gas inventories programme. In: Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), The Intergovernmental Panel on Climate Change. Institute for Global Environmental Strategies, Kamiyamaguchi Hayama, Japan.
ISO 140 44, 2006. Environmental Management – Life Cycle Assessment.
Lantz, M., Börjesson, P., 2014. Greenhouse gas and energy assessment of the biogas from Co-digestion injected into the natural gas grid - a Swedish case-study including effects on soil properties. Renew. Energy 71, 387–395.
López, M.M.C., Pernas, A.I.A., López, J.A., Latorre, L.A., Vilarino, J.M.L., Rodriguez, V. G., 2014. Assessing changes on Poly(ethylene terephthalate) properties after recycling: mechanical recycling in laboratory versus postconsumer recycled material. Mater. Chem. Phys. 147, 884–894.
Material Economics and ReSource, 2018. A value conservative Swedish material handling system. (Only available in Swedish). http://databas.resource-sip.se/storage/vardebestmtrlsystemrapport180118.pdf. (Accessed 19 May 2019).
Network for Transport measures, NTM. Calculation made 2018-10-15 v4/advanced/index.html?signout=disabled. https://www.transportmeasures.org/ntmcalc/v4/advanced/index.html?signout=disabled#/.
Plastics Europe, 2015a. Eco-profiles and environmental product declarations of the European plastics manufacturers, expandable polystyrene EPS. Plastics. February 2015.
Plastics Europe, 2015b. Poly methyl methacrylate (PMMA) – eco-profiles and environmental product declarations of the European plastics manufacturers. Plastics. January 2015.
Plastics Europe, 2015c. Styrene acrylonitrile (SAN) and acrylonitrile butadiene styrene (ABS) – eco-profiles and environmental product declarations of the European plastics manufacturers. Plastics. February 2015.
Rashid, A., Asif, F.M.A., Krajnik, P., Nicolescu, C., 2013. Resource conservative manufacturing: an essential change in business and technology paradigm for sustainable manufacturing. J. Clean. Prod. 57, 166–177.
SMED, MiljöEmissionsData, Svenska, 2012. Kartläggning Av Plastavfallsströmmar I Sverige. Sveriges Metereologiska Och Hydrologiska Institut, Report No 108. (Only Available in Swedish).
Waste Sweden, 2017. Life Cycle Assessment of Mechanical Soring of Residual Waste – Energy Consumption and Emissions of GHG. Avfall Sveriges Utvecklingssatsning, Rapport U 2017:09 (Only available in Swedish).
Swedish, E.P.A., 2018. Swedish recycling of packaging and newsprint – monitoring the EPR-system 2017 (Only available in Swedish). https://www.naturvardsverket.se/upload/sa-mar-miljon/mark/avfall/forpackningsrapport-2017-20181112.pdf. (Accessed 20 August 2019).
Swedish Energy Agency, 2018. Energy in Sweden - facts and figures 2018. Swedish National Energy Agency. Sweden. http://www.energimyndigheten.se/en/news/2018/energy-in-sweden–facts-and-figures-2018-available-now/. (Accessed 31 March 2018).
Swedish National Consortium for Emissions Data (SMED), 2010. Updating Climate Related Emission Factors. SMED Rapport Nr 92, p. 21, 2010. (Only available in Swedish).
Toniolo, S., 2012. Comparative LCA to evaluate how much recycling is environmentally favorable for food packaging. Recourses Conserv. Recycl. 77, 61–68.
Venkatachalam, S., et al., 2012. Degradation and recyclability of Poly (ethylene terephthalate). Intec. open space open minds (https//doi.10.5772/48612). https://www.intechopen.com/books/polyester/degradation-and-recyclability-of-poly-ethylene-terephthalate-. (Accessed 5 January 2018).
Wall, P., CEO Mälarplast, A.B., April 2018. Personal Communication.