



Life cycle assessment of post-consumer plastics production from waste electrical and electronic equipment (WEEE) treatment residues in a Central European plastics recycling plant



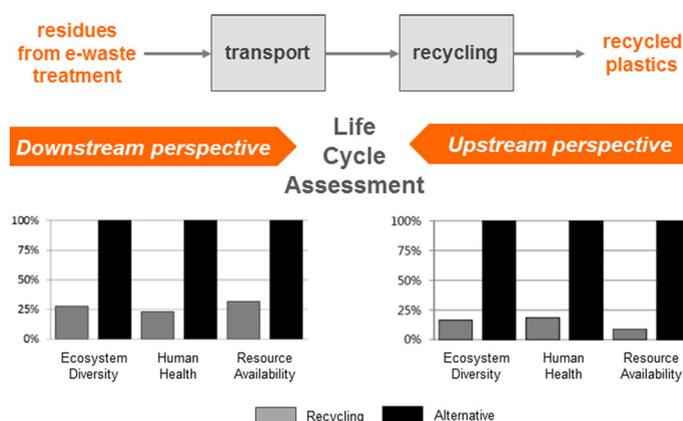
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HIGHLIGHTS

- LCA of plastics production from plastics-rich WEEE treatment residues
- Multiple stakeholder perspectives addressed via different research questions
- Plastics production from WEEE treatment residues clearly superior to alternatives
- Robust results as demonstrated by extensive sensitivity analyses

GRAPHICAL ABSTRACT



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ABSTRACT

Plastics play an increasingly important role in reaching the recovery and recycling rates defined in the European WEEE Directive. In a recent study we have determined the life cycle environmental impacts of post-consumer plastics production from mixed, plastics-rich WEEE treatment residues in the Central European plant of a market-leading plastics recycler, both from the perspective of the customers delivering the residues and the customers buying the obtained post-consumer recycled plastics. The results of our life cycle assessments, which were extensively tested with sensitivity analyses, show that from both perspectives plastics recycling is clearly superior to the alternatives considered in this study (i.e. municipal solid waste incineration (MSWI) and virgin plastics production). For the three ReCiPe endpoint damage categories, incineration in an MSWI plant results in an impact exceeding that of the examined plastics recycling facility each by about a factor of 4, and the production of virgin plastics has an impact exceeding that of the post-consumer recycled (PCR) plastics production each by a factor of 6–10. On a midpoint indicator level the picture is more differentiated, showing that the environmental impacts of the recycling options are lower by 50% and more for almost all impact factors. While this provides the necessary evidence for the environmental benefits of plastics recycling compared to

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existing alternatives, it can, however, not be taken as conclusive evidence. To be conclusive, future research will have to address the fate of hazardous substances in the outputs of such recycling systems in more detail.

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1. Introduction

Since the 1980s the mass share of plastics in electrical and electronic equipment (EEE) has continuously increased (APME, 2001). For waste electrical and electronic equipment (WEEE), the average mass share of plastics has been estimated to amount to about 21% by weight (Huisman et al., 2008; Ongondo et al., 2011), with great differences among the WEEE categories. According to Wäger et al. (2010), the plastics mass share varies between 3% in lighting equipment and up to 73% in toys, leisure and sports equipment; large household appliances, which constitute the most relevant fraction of WEEE with about 28% by mass in 2008, have a plastics mass share of about 19%.

At the same time, with the recast of the European Directive on waste electrical and electronic equipment (WEEE Directive) (European Union, 2012), which in particular aims at reducing the disposal of waste and to contribute to the efficient use of resources by re-use, recycling and other forms of recovery of such waste, the recycling targets will increase by August 15, 2015. Consequently, plastics play an increasingly important role in reaching the recovery and recycling rates defined in the European WEEE Directive.

However, plastics recycling itself is associated with a number of environmental issues. In particular, plastics from WEEE may contain substances considered harmful for human health and environment that should not be kept in a recycling loop but directed towards safe final sinks (Brunner, 2010; Buekens and Yang, 2014; Kral et al., 2013). Such substances are e.g. polybrominated diphenyl ethers listed in the Stockholm Convention on persistent organic pollutants (POPs) (Stockholm Convention, 2009a,b; UNEP, 2013a,b) and regulated in the Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (RoHS Directive) (European Union, 2011). Several studies have shown that substances of concern may be found in plastics from WEEE at levels that exceed maximum concentration values for new products defined in the European RoHS Directive. This includes substances that have deliberately been introduced into the plastics matrix during primary production as well as substances accidentally introduced into the plastics fraction through cross-contamination during pre-treatment of WEEE. Cadmium used in pigments (Schlummer et al., 2007) or commercial polybrominated diphenyl ethers (PBDE) like *c*-pentaBDE, *c*-octaBDE or *c*-DecaBDE used as flame retardants (Morf and Taverna, 2004; Morf et al., 2005; Peeters et al., 2014; Schlummer et al., 2007; Vyzinkarova and Brunner, 2013; Wäger et al., 2012) are examples for deliberately introduced substances, lead from printed circuit boards (Schlummer et al., 2007; Wäger et al., 2012) is an example for an accidentally introduced substance. Accordingly, the production of post-consumer plastics from WEEE requires a separation of these substances and/or of the plastics containing these substances to a level that is compliant with legal requirements (Sindikü et al., 2014; Wäger et al., 2012).

In this paper we present and discuss the results of a recent study that exhaustively investigated the life cycle environmental impacts associated with the production of post-consumer plastics from WEEE treatment residues in the Central European plant of a market-leading plastics recycler. The plant, which is located in Austria, treats material from all over Europe and includes numerous proprietary and patented processes. These allow the separation of the polymeric material from the other materials in the delivered plastics-rich WEEE treatment residues and the obtaining of post-consumer plastics sorted by type (ABS, HIPS, PP) for re-use in specific applications, such as consumer electronics (MBA Polymers, 2015). The processes include removal of non-plastics (metal, rubber, wood, glass, fluff, foam, textiles, dirt etc.), washing and

preparation (clean plastics and remove non-target plastics), polyolefin purification, styrenics purification (cleaning-sorting of ABS and HIPS) and formulation, blending and compounding (MBA Polymers, 2012). The life cycle environmental impacts were determined with a life cycle assessment (LCA) approach. In the last few years, many LCA studies related to waste management in general (Laurent et al., 2014a,b) and plastics recycling in particular (Lazarevic et al., 2010; Rajendran et al., 2012, 2013) have been performed. The recycling of plastics originating from WEEE treatment, however, only has been addressed in one study with a simplified representation of the recycling processes (Wäger et al., 2011).

2. Methodology

Life cycle assessment (LCA) is a method to assess the potential environmental impacts and resource consumption throughout a product's life cycle, i.e., from raw material extraction to waste management, including the production and use phases. LCA is generally seen as the most established and well-developed method in this area (see e.g. (Ness et al., 2007)). The related ISO 14040 standard (ISO, 2006) distinguishes four main steps within an LCA study: goal and scope definition, inventory modelling, impact assessment, and final interpretation phase. In the first step, the boundaries of the study are defined – as a study is always established relative to the objectives that are to be achieved (for a more detailed description see e.g. (Rebitzer et al., 2004)). The second phase is often the most time-consuming part, as the input and output values of each process within the boundaries have to be collected here, before the totality of all these material and energy flows is assessed in the third step based on ecological criteria.

2.1. Goal & scope

The goal of the study was to perform an in-depth life cycle assessment of the recycling of mixed, plastics-rich residues from WEEE treatment in an operational, state-of-the-art plastics recycling plant. In order to do so, two different perspectives were applied, which are reflected in the following two research questions:

1. How do different recovery and disposal routes for plastics-rich WEEE treatment residues originating from the mechanical treatment of WEEE perform from an environmental point of view?
2. How do post-consumer recycled (PCR) plastics originating from the investigated plastics recycling plant perform in comparison to the respective virgin plastics production from an environmental point of view?

While research question 1 allows taking into consideration the perspective of recyclers producing this type of waste fraction, research question 2 addresses the perspective of customers purchasing plastic granulates to manufacture plastic parts for their products.

The LCAs performed in this study can be considered to be of the accounting type, as they focus on the comparison of an existing recycling process with alternatives producing a similar reference flow of materials. Regarding research question 1, the environmental impacts of the recycling process were compared with the impacts of a “final disposal” of the same residual material through incineration in a state of the art European Municipal Solid Waste Incineration (MSWI) plant. Landfilling has not been considered, as legislation is not encouraging this type of “final disposal” any more for organic fractions (European Union, 2012). Regarding research question 2, the environmental impacts of the plastics recycling process were compared with those resulting from primary production of an identical amount of (plastics) material.

Attributional LCA models using a common “basket of products” through system expansion were used in both cases (ISO, 2006). Such a “basket of products” approach makes the evaluated treatment and production pathways comparable by extending each of them in such a way that they cover a common set of products and services (Fleischer, 1994).

2.2. System boundaries and functional unit

In order to answer the two research questions, appropriate system boundaries and functional units were defined for each of them. For research question 1, a functional unit¹ of 1 tonne of plastics-rich WEEE treatment residues was defined, which is equivalent to the reference flow² for the calculation. As the actual collection/separation of the WEEE devices is identical for both treatment options (i.e. plastics recycling vs. MSWI), the corresponding processes have not been included in the systems represented. In view of having an identical product output or basket of products, respectively, the systems were each expanded with additional processes where necessary. The related system boundaries are shown in the upper part of Fig. 1.

For research question 2, a functional unit of 1 tonne of a defined mixture of plastics has been considered, which again is equivalent to the reference flow for the calculation and addresses the complete treatment chains for both options investigated (PCR plastics production vs. virgin plastics production). The contribution of collection/separation of WEEE devices to the production of plastics from plastics-rich residues has been determined by economic allocation between the different output fractions from the WEEE treatment activities. The related system boundaries are shown in the lower part of Fig. 1.

The “basket of products” for research question 1 include more products than the basket of products for research question 2, because incineration in the MSWI plant is part of the examined system, which results in the production of a certain amount of heat and power (see Fig. 1, upper part). For both research questions the combustion of the plastics fraction directed to the cement kiln could not be addressed, because the existing cement kiln model in ecoinvent does not include bromine and heavy metals transfer coefficients. Alternatively, it was assumed that the plastics fraction going into the cement kiln is replacing a 50:50 mix of heavy fuel and coal with an energy content similar to the plastics fraction. Regarding the substitution of virgin plastics with PCR plastics, we assumed a 1:1 replacement.

2.3. Inventory analysis

In this study, primary and secondary data from different sources have been used. Primary data, i.e. direct information about material and energy flows as well as material compositions related to the plastics recycling process were obtained from the company running the recycling plant. In all cases where actual, i.e. measured numbers for the process were not available, we made estimations based on the inputs of the company representatives and verified these assumptions with sensitivity analyses. For secondary data we referred to the database ecoinvent v2.2 (ecoinvent Centre, 2010), which includes average European data for most existing materials and energy supply processes and/or services (such as transport or waste treatment). The consideration of a possible recovery of metals in the MSWI process would have required establishing new data models, which was not possible in the frame of this study. Table S1 in the Supporting information compiles the data used to model the four different systems shown in Fig. 1.

¹ According to ISO (2006), the functional unit is the “quantified performance of a product system for use as a reference unit”.

² According to ISO (2006), the reference flow represents “measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit”.

2.4. Impact assessment

The impact assessment was performed with one of the most recent and up-to-date LCA methods, the ReCiPe method (Goedkoop et al., 2009), both on the mid- and end-point levels. Addressing both levels allows getting a comprehensive view of the examined processes; the use of the same method for mid- and end-point levels is considered to be more consistent. On the mid-point level, the following commonly used categories were included in this study: terrestrial acidification potential (TAP), global warming potential (GWP), freshwater eutrophication potential (FEP), photochemical oxidant formation potential (POFP), ozone depletion potential (ODP), fossil depletion potential (FDP), freshwater ecotoxicity potential (FETP), marine ecotoxicity potential (METP), human toxicity potential (HTP), and terrestrial ecotoxicity potential (TETP). The relevance of these midpoint indicators was further examined by applying a normalisation step, based on the situation in Europe. With regard to the end-point level, the default (European H/A) perspective from ReCiPe was used, taking into account the actual geographical situation of the recycling plant, which treats post-consumer material originating from various European WEEE treatment facilities. In addition to the aggregated total, the three damage categories “Human Health”, “Ecosystem Diversity” and “Resource Availability” are shown and discussed individually as well. It has to be kept in mind, however, that such results represent potential and not actual environmental impacts.

3. Results and discussion

3.1. Comparison of treatment options for plastics-rich residues

Fig. 2 shows the results of the various mid-point indicators from ReCiPe for the two treatment options recycling and incineration in an MSWI plant. The upper part of the figure shows the total values relative to the option with the higher impacts, while the lower part addresses the split into the individual process steps that belong to each of the two options.

With two exceptions, the incineration in an MSWI plant results in impacts that exceed those of recycling by a factor of 4 and more. The exceptions are the factors “FEP” (freshwater eutrophication potential), with a difference of only factor 2 between the recycling and the MSWI option, and “ODP” (ozone depletion potential), with a recycling impact exceeding the corresponding score for the MSWI option by about 25%. 70% of this impact are due to the additional “heat & power production” (caused to a large extent by Halon 1211 and Halon 1301 releases in the fossil fuel supply) required to establish the common basket of products. Similarly, in the case of the FEP impact, the additional “heat & power production” contributes to the overall impact with about 50%; caused by phosphorus emissions into ground water (related mainly to the copper production chain). In the case of GWP and FDP the processes for the additional “heat & power production” contribute to about 70% of the overall impact of the recycling option; but in both cases the overall impact of the recycling option nevertheless stays about 4 times lower than the incineration in an MSWI plant. The high impact of the MSWI option is due to the virgin plastics production and the incineration process itself (GWP, caused by CO₂ emissions) and the primary metal production (FDP – caused by consumption of fossil fuels in whole metal supply chain), respectively. In case of TAP and POFP, the additional “heat & power production” contributes to about 50–60% of the recycling option, again resulting in an about 4 times lower impact than for the MSWI option. In both cases, virgin plastics and primary metals production are responsible for about 80% of the impact of the MSWI option (which in the case of TAP are dominated by the releases of sulphur dioxides along the supply chains). The four toxicity impact factors (FETP, HTP, METP, and TETP) show a clearly different picture, especially for the MSWI option. In all four cases, the impacts of the MSWI option are largely dominated by the actual incineration of

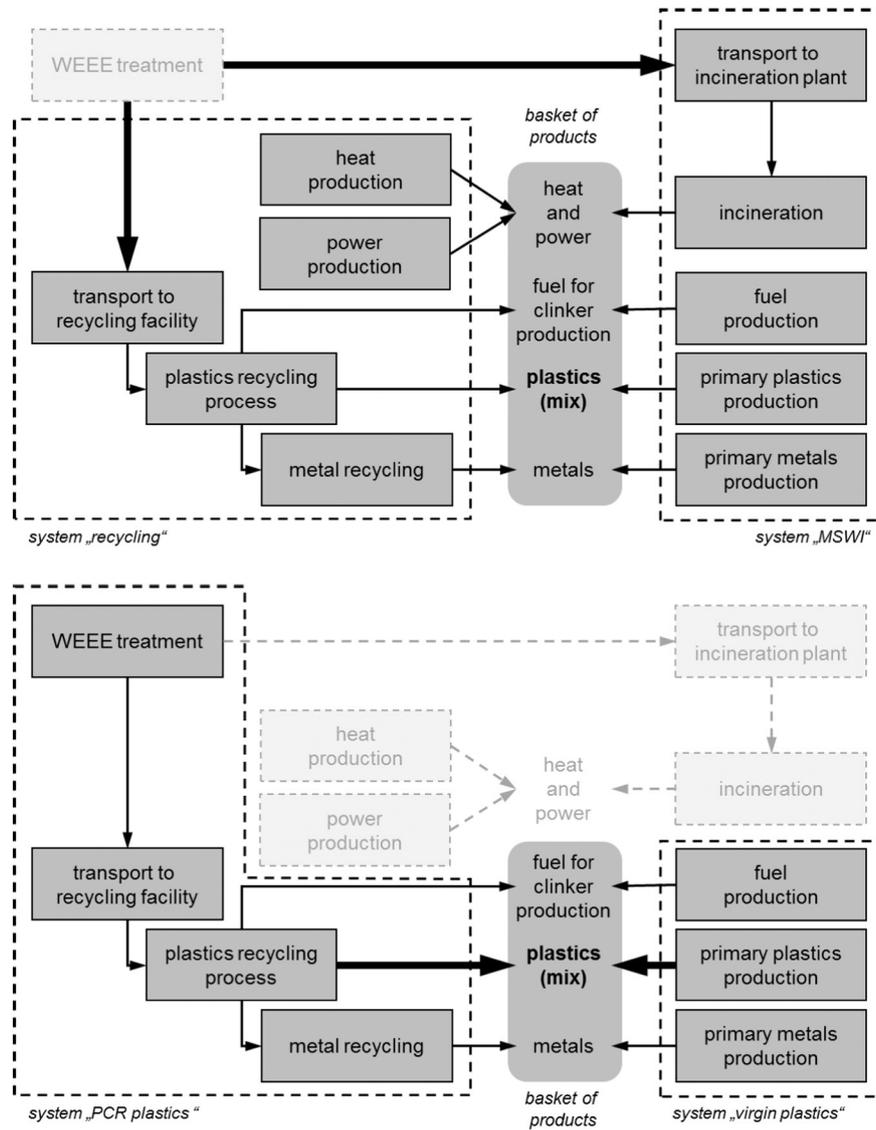


Fig. 1. System boundaries for the comparison of different treatment options for plastics-rich WEEE treatment residues (research question 1, upper part) and of PCR plastics with virgin plastics (research question 2, lower part). The bold arrows represent the corresponding reference flows.

the plastics-rich WEEE treatment fraction (65% for HTP, 95 and more % for the three other toxicity factors, mainly caused by direct emissions of copper (FETP, METP), bromine (TETP), and antimony (HTP) in the incineration step). As a consequence, the overall result for the MSWI option is up to almost 400 times higher (FETP) than for the recycling option. The process steps related to the actual recycling activities (i.e. transport to the plant and all subsequent separation, pre-treatment and recovery processes, summarized as “plastics recycling process” in the figure) only contribute to about 5 (TETP) to 15% (METP) of the total impact of the recycling option. All further impacts are due to the additional processes taken into account here in order to establish a common basket of products.

Fig. S1 in the Supporting information shows the normalised ReCiPe midpoint indicator values for the two treatment options (recycling, MSWI). Normalisation, which consists in weighing the scores of the impact factors for the examined system against the scores of the corresponding impact factors of a reference system (e.g. Europe), gives an indication of the relevance of the impact factors addressed. The higher a normalised impact factor, the more relevant it is. The highest normalised impacts were found for the categories FETP, HTP, and METP, followed by the category FEP and, with clearly lower values, the categories FDP, GWP, TETP and TAP. As a consequence, environmental impacts

related to toxicity and eutrophication appear to be more relevant than e.g. the global warming potential for the examined system.

On the level of the ReCiPe endpoint method, the comparison between the two systems results (on the level of the relative total values, see Fig. 3) in a similar picture for all three damage categories and the total impact: incineration in an MSWI plant results in impacts exceeding those of the examined plastics recycling facility by a factor of about 4 (see Fig. 3). Again, the process steps of the actual recycling activity (transport to the plant and the subsequent pre-treatment and recovery) are only responsible for about 20% of the total impact from recycling; the remaining impacts are again due to the expansion of the system required to establish the common basket of products. For the MSWI option, the damage categories “Ecosystem Diversity” and “Human Health” are dominated by the actual disposal process (i.e. the incineration of the plastics-rich WEEE treatment fraction in an MSWI plant), while the damage category “Resource Availability” is dominated by the virgin plastics production.

3.2. Comparison between PCR plastics and virgin plastics

The comparison between PCR plastics and virgin plastics production, which uses 1 tonne of produced plastics as the reference flow, neither

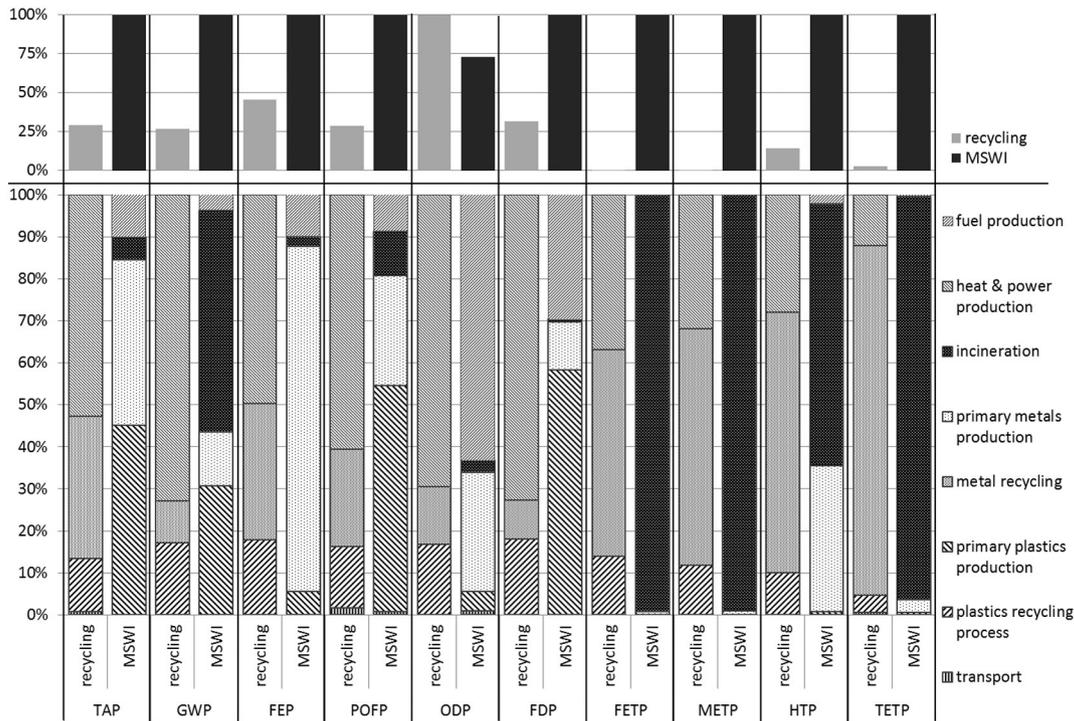


Fig. 2. Environmental impacts associated with the two treatment options (recycling, MSWI) for 1 tonne of plastics-rich WEEE treatment residues (upper part: total values, relative to the option with the higher impacts; lower part: split into the individual process steps for each option). Shown are the ReCiPe mid-point indicators terrestrial acidification potential (TAP), global warming potential (GWP), freshwater eutrophication potential (FEP), photochemical oxidant formation potential (POFP), ozone depletion potential (ODP), fossil depletion potential (FDP), freshwater ecotoxicity potential (FETP), marine ecotoxicity potential (METP), human toxicity potential (HTP), and terrestrial ecotoxicity potential (TETP).

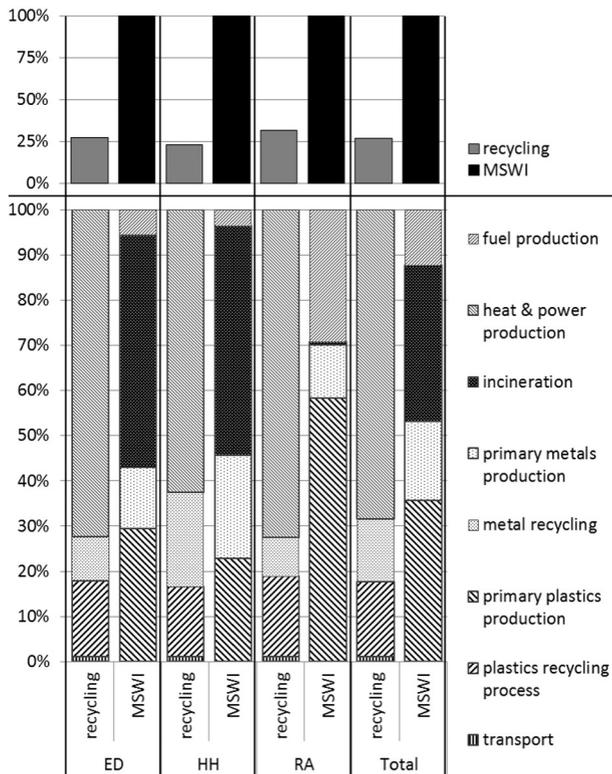


Fig. 3. Environmental impacts associated with the two treatment options (recycling, MSWI) for 1 tonne of plastics-rich WEEE treatment residues (upper part: total values, relative to the option with the higher impacts/lower part: split into the individual process steps for each option). Shown are the ReCiPe endpoint results for the damage categories “Ecosystem Diversity” (ED), “Human Health” (HH), and “Resource Availability” (RA), as well as the total impact.

includes the MSWI process nor the heat and power production steps (see Fig. 1). As shown in Fig. 4, this results in much smaller differences for the four toxicity categories than the previous comparison of the treatment options for plastics-rich residues. For FETP, METP and HTP, differences by about a factor of 3–4 can be observed between the production options (PCR, virgin), in comparison to a difference by a factor of almost 400 between the treatment options (as discussed above). In case of TETP, the impacts of the virgin plastics option only exceed those of the PCR plastics option by about 25%. In all four cases the primary metals production is the main contributor for the impacts, being responsible for more than 80% of the impacts in case of the various toxicity categories. The process identified to be the main contributor in research question 1 (i.e. the actual incineration of the plastics-rich shredder fraction) is not part of the system examined in research question 2.

Compared to the outcomes for research question 1, an accentuation of the difference between the two plastics production options (PCR plastics vs. virgin plastics) can be observed for all the non-toxicity indicators. Even the ODP indicator now results in a clear advantage for the PCR plastics option, i.e. a value that is less than half of the value for the virgin plastics option. Still, however, the ODP results are dominated by the plastics recycling process and the fuels production, respectively. In case of the FEP results, the included metal processes (recycling and primary production) are dominating the overall impact. All other non-toxicity factors in the virgin production option are dominated by the primary production of the three different types of plastics (being responsible for 60% and more of the total impact).

Fig. S2 in the Supporting information shows the normalised ReCiPe midpoint indicator values for the two plastics production options. The picture is quite similar to Fig. S1, i.e. the highest normalised impacts can be observed for the categories FEP, FETP, HTP, and METP, followed by the indicators for FDP, TAP and GWP.

Fig. 5 shows the results for the ReCiPe endpoint method. The scores for the three damage categories and the total impact in the upper part of

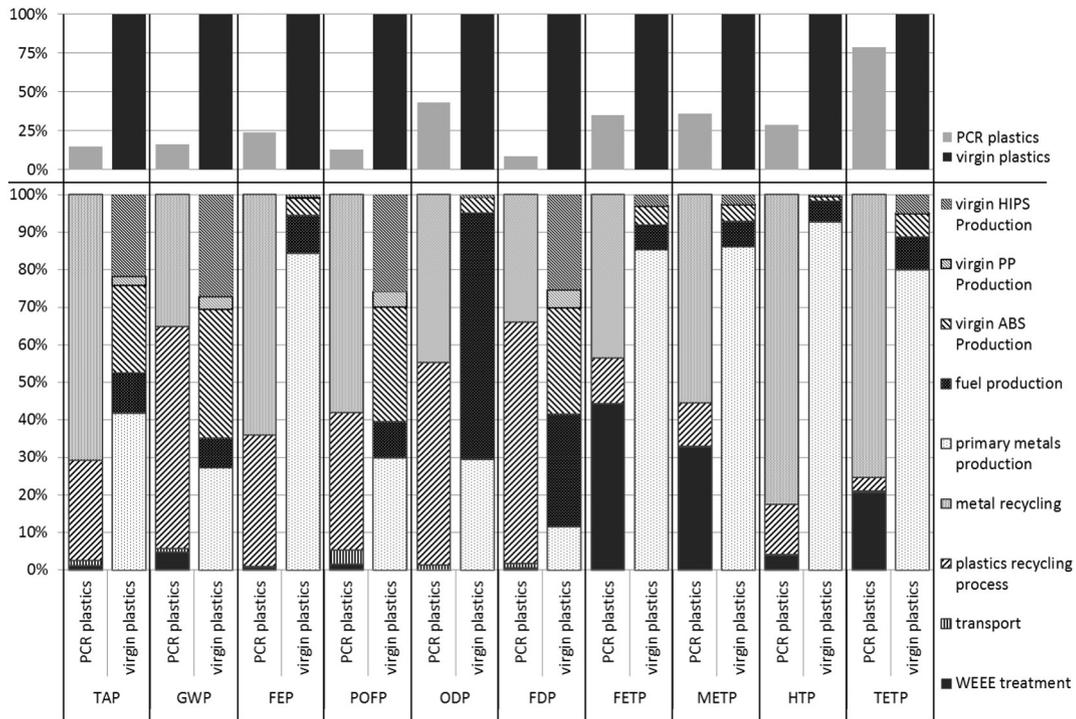


Fig. 4. Environmental impacts associated with the production of 1 tonne of plastics from post-consumer recycling (PCR plastics) and primary production (virgin plastics) (upper part: total values, relative to the option with the higher impacts; lower part: split into the individual process steps for each option). For an explanation of the abbreviations of the reported ReCiPe midpoint indicators, see Fig. 2.

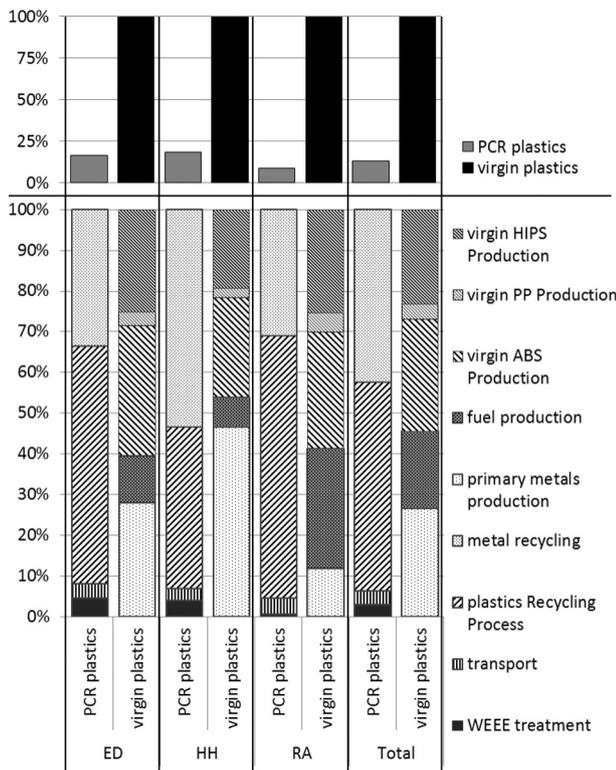


Fig. 5. Environmental impacts associated with the production of 1 tonne of plastics from post-consumer recycling (PCR plastics) and primary production (virgin plastics) (upper part: total values, relative to the option with the higher impacts; lower part: split into the individual process steps for each option). For an explanation of the abbreviations of the reported ReCiPe endpoint indicators, see Fig. 3.

this figure show a similar pattern as the respective figure from research question 1 (Fig. 3). Virgin plastics production has an impact exceeding that of the PCR plastics production by a factor of 6–10. The process steps of the actual recycling activity (i.e. WEEE treatment, transport and the subsequent plastics recycling process) are responsible for about 70% of the total impact of the PCR plastics system. Within the virgin plastics production system, the production of the primary plastics accounts for 50–60% of the total impact. The remaining part is in both cases largely dominated by the respective metal treatment process step.

3.3. Sensitivity analyses

Despite a close collaboration with the plant owner, some parameters used in this comparison are still based on assumptions only. In order to evaluate the relevance and influence of these parameters, several sensitivity analyses were performed for each of the two research questions by varying each of these assumed parameters. In both cases, only aspects directly related to the plastics recycling process have been examined. Possible influences due to variations in the processes of the examined basket of products have not been taken into account; those processes have been defined in function of the actual situation of the examined recycling plant.

Table 1 gives an overview of the established sensitivity analyses and the corresponding parameter settings for research question 1 (how do different recovery and disposal options perform?).

Sensitivity analysis A addresses the actual composition of the input material. The composition of the input material, which is subject to changes over time due the varying composition of delivered plastics-rich WEEE treatment residues and fluctuations in the supplier market, has a direct influence on a variety of output fractions (plastics, metals, fuel for clinker, etc.) and on the amount of heat/power that can be produced in the MSWI route. Sensitivity analysis B addresses the emission pathways of impact-relevant substances, namely bromine, heavy metals and NMVOC by assuming a worst-case scenario of releases into the air (compared to no such emissions in the default case, due to a lack of respective information from the plant owner). The assumption

Table 1
Parameter settings for the sensitivity analyses regarding the investigated recovery and disposal options for plastics-rich WEEE treatment residues.

Sensitivity analysis	Parameter settings	
A: composition of delivered plastics-rich WEEE treatment residues	Default	Data from recycling plant
	A1	“High” metals content of 18% and low plastics content of 77%
	A2	Ecoinvent default data of treatment residues with 93% plastics content
	A3	High plastic content (96%) and low content of other materials (2%)
B: emission pathway of impact-relevant substances	Default	Bromine (originating from the brominated flame retardants (BFRs)) and heavy metals leaving the system as a part of the fuel
	B	10% of the bromine (originating from the BFRs) as “emissions to air” + 10% of all heavy metals emitted as “emissions to air”. NMVOC emissions to air correspond to 100% of the waste oil fraction + 0.001% of the plastics fraction (assumed worst case)
C: increased amount of recycled plastics	Default	Data from recycling plant
	C1	10% of plastics currently going into clinker production as a fuel are also recycled
	C2	25% of plastics currently going into clinker production as a fuel are also recycled
	C3	50% of plastics currently going into clinker production as a fuel are also recycled

that 10% of the original bromine content is released into air during the recycling of the plastics can be seen as a worst case situation – modern clinker production sites show transfer coefficients e.g. for chlorine or antimony of less than 0.1% (Bösch et al., 2009; Vermeulen et al., 2009), which is 100 times lower than the factor used for bromine here. Finally, sensitivity analysis C shows the effect of a higher plastics recycling yield on the overall results. The influence of all these three sensitivity analyses on the investigated recovery and disposal options for plastics-rich WEEE treatment residues is shown in Fig. 6 for the three damage categories of the ReCiPe endpoint method.

For sensitivity analysis A, the results clearly show that the actual composition of the input material has no relevant influence on the overall result – independent of the changes in the composition that are

assumed. In case of sensitivity analysis B, the damage on the ecosystem diversity is hardly influenced (change of less than 5%); however, these bromine emissions have a significant impact on the human health damage, resulting in about 160% higher impacts. But despite this high increase the MSWI option still shows a value that is about 2 times higher. Finally, according to sensitivity analysis C, a higher recycling efficiency does not have a big influence on the overall impact for the recycling option. This is due to the fact that the overall impact is dominated by the additional “heat & power production” (see Fig. 3). For the MSWI option, the overall impact would increase by 10 to 12% if 50% of the currently “lost” plastics fraction (i.e. the fraction going as fuel into the clinker production) was recycled in addition. The higher impact compared to the recycling option can be explained by the greater amount of primary plastics that is taken into account. A higher recycling efficiency could e.g. be achieved by including a solvent extraction process for the elimination of BFRs (Freegard et al., 2006; Schlummer et al., 2012).

Table 2 gives an overview of the established sensitivity analyses and the corresponding parameter settings for research question 2 (how do post-consumer recycled plastics perform in comparison to the respective virgin plastics?).

The results of the four sensitivity analyses for research question 2, again expressed via the three damage categories of the ReCiPe endpoint method, are summarized in Fig. 7. It can be clearly seen that the influence on the overall results for the PCR plastics option is rather small. For sensitivity analyses D and E, the production of 1 kg of PCR plastics results in variations of the results for the PCR plastics option of 0.4 to 12%. Only in the case of sensitivity analysis F, which addresses the split between the different recycled metals, an influence of up to almost 30% on the result for PCR plastics could be observed. The changes in the metal split also affect the results for the “virgin plastics” option, resulting in variations ranging from – 10 to + 36%. Last but not least, sensitivity analysis G is dominated by the amount of ABS that could be recycled: the higher the amount of ABS, the higher the overall impact of the virgin plastics option.

All in all, the four sensitivity analyses show that variations in the different parameters do not result in a change of the overall conclusion – i.e. the PCR plastics option results in all cases still in a clearly lower impact compared to the virgin plastics option.

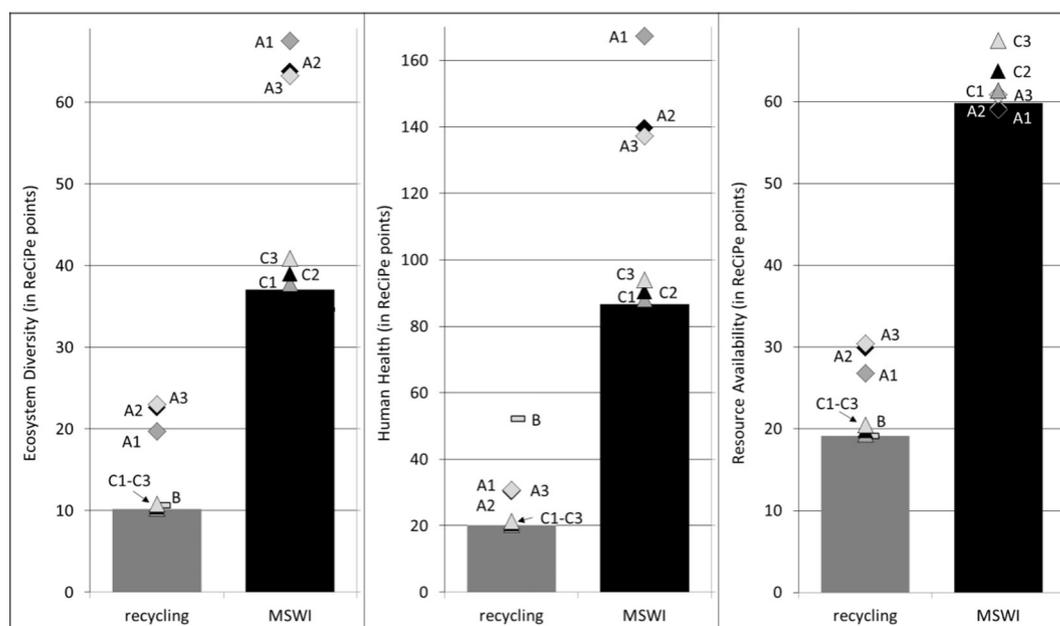


Fig. 6. Environmental impacts associated with the two treatment options (recycling, MSWI) for 1 tonne of plastics-rich WEEE treatment residues, including the influence of sensitivity analyses A1 to C3.

Table 2
Parameter settings for the sensitivity analyses regarding the investigated plastics production options.

Sensitivity analysis	Parameter settings
D: allocation factor of the WEEE separation step	Default 15%, based on an economic allocation (plastics-containing fractions vs other parts)
	D1 25%, mass-based allocation (plastics containing fractions vs other parts)
	D2 0% (as WEEE recycler has to pay for the treatment in the plant)
E: transport modes and distances for the input material	Default Distance Switzerland to plant; i.e. an average transport distance of 250 km
	E1 Average transport distance of 50 km, representing a local supplier structure
	E2 Average transport distance of 500 km, representing short European distances
	E3 Average transport distance of 1000 km, representing long European distances
F: split between recycled metals	Default Split between steel, copper & aluminium according to plant information
	F1 High share of steel (68%); copper & aluminium having been reduced accordingly
	F2 High share of copper (56%); steel & aluminium having been reduced accordingly
	F3 Inclusion of precious metals (gold, silver) – keeping default values for other metals
G: split between recycled plastics types	Default Split between ABS, HIPS & polypropylene according to plant information
	G1 High share of ABS (55%), having 20% PP and 25% HIPS
	G2 High share of ABS (45%) and PP (40%), having 15% HIPS
	G3 High share of PP (60%), having 30% ABS and 10% HIPS

4. Conclusions

In this paper, we calculated the environmental impacts associated with the recycling of shredded plastics-rich by-products created from WEEE treatment activities. Hereby, we applied two perspectives, one looking “downstream” of the plastics containing fractions to be recycled, the other “upstream” of the secondary plastics produced out of the same fractions.

Our approach allows the evaluation of the environmental benefits (or burdens) of recycling from the perspective of key stakeholders on the level of interpretation (ISO, 2006). In the case of plastics recycling from plastics-rich WEEE treatment residues, these key stakeholders are the customers providing the feedstock for the recycling plant and the customers purchasing the recycling product.

Applying multiple stakeholder perspectives to answer the question, if plastics recycling from plastics-rich WEEE treatment residues is environmentally beneficial, is a possible approach to better address the complexity of socio-technical systems. In any case, a clear advantage of the chosen approach is that the operators of the recycling plant will be able to selectively address and inform their key stakeholders on the environmental benefits of the plastics recycling option.

With the exception of the review process, where only one reviewer was involved, the study has been performed in accordance with the requirements of the ISO 14040 series for life cycle assessment. The study also complies with most recommendations for a better LCA practice in LCA studies of solid waste management systems recently presented (Laurent et al., 2014b). In particular, much effort was spent in collecting data for the foreground processes, which was done iteratively with the representatives of the plastics recycling plant, and in performing extended sensitivity analyses to address uncertainties. Still, the limited data availability did not allow to check mass balances for individual waste components and to track waste and substance flows with tools

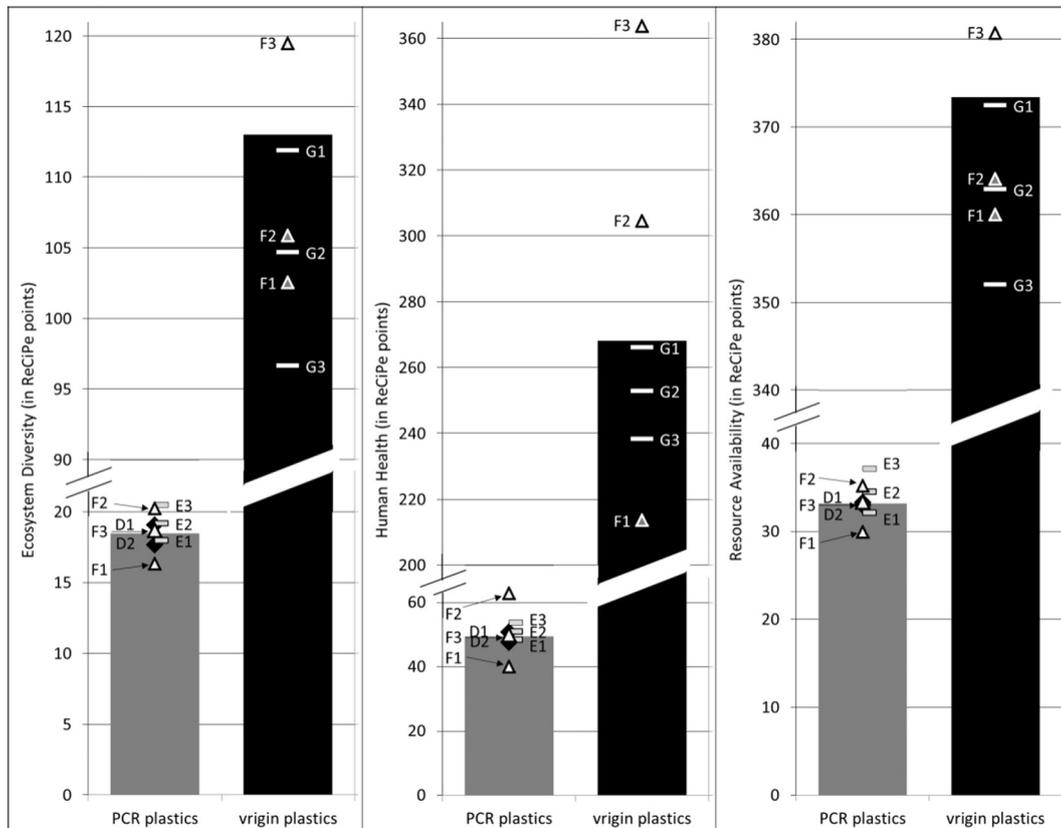


Fig. 7. Environmental impacts associated with the production of 1 tonne of plastics from post-consumer recycling (PCR plastics) and primary production (virgin plastics), including the influence of the sensitivity analyses D1 to G3.

combining substance flow analysis and LCA such as EASETECH or ORWARE (Clavreul et al., 2014; Eriksson et al., 2002). This was further accentuated by the fact that on the level of inventory data some processes in ecoinvent are not able to calculate substance specific emissions, e.g. bromine in cement kilns. Nevertheless, it has to be emphasized that the investigated recycling plant is committed to prevent hazardous substances to enter the recycling loop and complies with relevant national and international legislations, e.g. the RoHS Directive (European Union, 2011).

The results of our study show that the recycling of plastics from plastics-rich WEEE treatment residues is clearly superior to alternative disposal and production routes. This holds true both from a downstream perspective (research question 1), i.e. the perspective of the customer delivering plastics-rich WEEE treatment residues (with the incineration of the residues in an MSWI plant as the alternative) and the upstream perspective (research question 2), i.e. the perspective of the customer purchasing the recycling product (with virgin plastics production as the alternative). For the ReCiPe endpoint assessment method, for example, the recycling of plastics-rich WEEE treatment residues results in impacts that are about 4 times lower than those for the disposal in an MSWI plant and 6 to 10 times lower than those for the virgin plastics production. This outcome is indifferent to variations in key parameters such as allocation factors in the WEEE separation step, transport modes and distances, split of recycled metals and plastics or the emission pathways of impact-relevant substances.

The results of our study are in line with other studies on plastics recycling, which however mainly focus on other, mostly pure plastics fractions (Lazarevic et al., 2010; Rajendran et al., 2012), and provide the necessary evidence for the environmental benefits of plastics recycling compared to existing alternatives. Yet they cannot yet be taken as conclusive evidence. Firstly, the technical data of the produced PCR plastics do not exactly correspond to those for virgin plastics. Accordingly, the produced PCR plastics only can replace virgin plastics in those cases, where they comply with the specifications of the customer or where they can be modified with standard additives (such as impact modifiers) to meet the desired specifications. Second, we could not systematically address the fate of hazardous substances in the outputs of the plastics recycling plant. Future research will have to address the fate of hazardous substances in the outputs of such recycling systems in more detail to allow for a conclusive scientific evaluation of different plastics recycling systems, including systems operating in a more informal context (Buekens and Yang, 2014; Sindiku et al., 2014). In particular, this will require further efforts in the compositional characterisation of the outputs of such systems and the application of appropriate substance flow analysis and risk assessment methodologies.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.scitotenv.2015.05.043>.

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