Environmental life cycle cost assessment: Recycling of hard plastic waste collected at Danish recycling centres

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Published in:
Resources, Conservation and Recycling

Link to article, DOI:
[10.1016/j.resconrec.2019.01.014](https://doi.org/10.1016/j.resconrec.2019.01.014)

Publication date:
2019

Document Version
Peer reviewed version

[Link back to DTU Orbit](#)

Citation (APA):
Faraca, G., Martinez Sanchez, V., & Astrup, T. F. (2019). Environmental life cycle cost assessment: Recycling of hard plastic waste collected at Danish recycling centres. *Resources, Conservation and Recycling*, 143, 299-309. <https://doi.org/10.1016/j.resconrec.2019.01.014>

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Abstract

Recycling of plastic waste is promoted by the European Union as an important step toward a circular economy. Recovered plastic waste is a complex and heterogeneous material, and the impurities and/or untargeted polymers associated to plastic waste may affect the recycling process and potentially decrease the intended benefits. An environmental and financial assessment was conducted on one tonne of hard plastic waste collected at Danish recycling centres. Three management scenarios were considered: two mechanical recycling (a simpler and a more advanced configuration, namely sMR and aMR) and a feedstock recycling (FR) scenario based on conversion through pyrolysis. Scenario aMR provided the largest savings in the highest number of impact categories (including global warming potential) and total costs; scenarios sMR and FR provided smaller savings (or even burdens), depending on the environmental impact category considered. A scenario analysis evaluating the type of energy provision, location of recycling facilities and the application of the recycled material confirmed the ranking of results with respect to global warming potential and total costs. A global sensitivity assessment of model data inputs demonstrated that three to nine parameters were typically sufficient to achieve more than 90% of total variance of the results; critical parameters were mainly related to sorting efficiencies, technical yields and market substitution factors. The study demonstrates that if high quality of the recycled plastic is achieved, both environmental savings and financial revenues are possible.

Keywords: Circular economy; Life Cycle Assessment; Global Warming Potential; Global Sensitivity Assessment; contamination; material recovery facility

Abbreviations

eLCC: Environmental life cycle costing; EU: European Union; ET: terrestrial eutrophication; FU: functional unit; GWP: global warming potential; HDPE: high density polyethylene; HTc: human toxicity (cancer); HTnc: human toxicity (non cancer); LCA: Life cycle assessment; LCC: Life cycle costing; LDPE: low-density polyethylene; MRF: material recovery facility; MSWI: municipal solid waste incinerator; NIR: near infra-red; PET: polyethylene terephthalate; POFP: photochemical ozone formation; PP: polypropylene; PS: polystyrene; PVC: polyvinyl chloride; SC: sensitivity coefficients; SR: sensitivity ratios; TA: terrestrial acidification; TD: triangular distribution

1 Introduction

Plastic represents a broad class of polymeric materials fulfilling a wide range of purposes in today's society, depending on its mechanical and chemical properties, e.g. for packaging, construction materials, transportation, electronics, agriculture, etc. The European demand for plastic in 2016 was 57 million tonnes (EC, 2018), of which 70% was made of six main polymers: polyethylene terephthalate (PET), high- and low-density polyethylene (HDPE and LDPE), polyvinyl chloride (PVC), polypropylene (PP) and polystyrene (PS). Plastic eventually becomes waste, which is estimated at about 25 million tonnes (in 2015; Plastic Europe, 2016) and about 10-15% of total municipal solid waste in Europe (e.g. Giugliano et al., 2011; Rigamonti et al., 2013; Miljøstyrelsen, 2014). About 85% of this plastic waste is made from the six polymer types mentioned previously (Villanueva and Eder, 2014). In Denmark, 33000 tonnes of plastic waste are generated every year (Miljøstyrelsen, 2017a). To increase circular economy solutions and resource efficiency, the recycling of these materials is important. However, little attention has been paid to identifying the most appropriate recycling options from an environmental and financial perspective.

The European Union (EU) promotes recycling as an important step toward a circular economy (EEA, 2016; EPRS 2016), and in early 2018 a strategy was released to address challenges related to plastic waste recycling (EC, 2018). In 2015, EU Member States recycled 30% of plastic waste generated by consumers, i.e. post-consumer plastic waste (Plastic Europe, 2016). In Denmark, around 20% of post-consumer plastic waste is recycled, while the rest is incinerated (less than 5% is landfilled; Plastic Europe, 2016); only hard plastic waste (rigid products such as bottles and containers, pots, pipes, crates, toys, etc.; see definition in Appendix B) is generally recycled in Denmark. This relatively low recycling rate reflects a traditionally low coverage of separate collection for plastic waste in many municipalities (Miljøstyrelsen, 2017b; Sweco Danmark A/S, 2015) and the presence of a deposit-refund scheme for plastic bottles, which until recently was not accounted in the national recycling rates (DRS, 2015). In many parts of Denmark, recycling centres represent an important collection method for plastic waste (Larsen et al., 2010). Recycling centres are manned collection points where the waste is sorted into around 40 material fractions. This collection method is used widely across the UK, Denmark, Norway and Sweden (Krook et al., 2010), with 44-50% of source-separated post-consumer plastic waste collected from Danish recycling centres

in 2015 (Miljøstyrelsen, 2017b; Sweco Danmark, 2015). In the rest of Europe, recycling centres account for 10-15% of collected recyclables (Villanueva and Eder, 2014). As the composition and quality of collected plastic waste is affected by the collection system itself, specific attention has to be paid to materials originating from these recycling centres. Today, such studies are scarce in literature.

An assortment of plastic recycling options exists, differing in number, type and sequence of processes. Recycling processes for plastic waste have been classified by the scientific literature and legislation standards as a) primary (re-extrusion), b) secondary (mechanical recycling), c) tertiary (feedstock or chemical recycling) and d) quaternary (energy recovery) routes, according to the technology used and the output produced by the process (e.g. ASTM D5033; Brems et al., 2012; Ignatyev et al., 2014). While primary recycling routes are applied to pre-consumer (industrial) plastic only, due to the high level of homogeneity requested, traditionally secondary recycling routes have been the main option for processing post-consumer plastic waste. Secondary recycling involves sorting and reprocessing steps to produce single-polymer pellets, granulates or flakes for use in new plastic products. Tertiary routes represent technologies involving breaking down the polymer chains into smaller molecules. Technologies like hydrolysis and pyrolysis produce liquids that can be used in the production of new plastics, synthetic fibres, lubricants and fuels, or other products used by the chemical industries (Ignatyev et al., 2014; Yu et al., 2016). Finally, quaternary routes involve any type of energy recovery. The European Waste Framework Directive (WFD) recognises as recycling only those processes producing outputs that can be used as secondary raw materials (EC, 2008). Mechanical recycling certainly falls into this category, whereas the classification of tertiary technologies depends on the final use of the product (Villanueva and Eder, 2014). Quaternary routes are not defined as recycling by the WFD and will not be addressed in this study. Table B.1 in the Appendix describes key differences between recycling routes.

Recycling is generally accepted as a sustainable waste management strategy; however, plastic waste recycling is often more complicated than for other material fractions (e.g. glass and metals; Alwaeli, 2009; Rigamonti et al., 2015). Difficulties arise mainly from presence of impurities that may act as technical, economic and/or market barriers to substitution of virgin plastic (RDC Environment, 2003). Impurities in plastic waste may be grouped into three classes: non-plastic components (foreign materials), non-targeted plastic (different polymers than the one targeted for reprocessing, including multi-

plastic products and polymer blends) and chemical impurities (bound to the plastic matrix, such as pigments, additives, stabilizers, etc). In the EU, the presence and concentration of impurities is regulated in plastic products (i.e. “input” to the waste phase) and in the outputs from the reprocessing industry (the recyclates; EN 153-42 to -48; ISRI, 2017). Business-to-business specifications are also widely used (Villanueva and Eder, 2014). These standards reflect that impurities may have various consequences on mechanical recycling, from decreasing processing efficiencies, through lowering the mechanical performance of recyclates, to inclusion of unwanted (and potentially hazardous) chemicals in the product matrix. The purity of plastic waste is the primary factor determining the quality and economic value of recyclates (ISRI, 2017), and it is in turn dependent upon the type and number of sorting and reprocessing steps. The presence of impurities influences recycling costs not only in terms of investments in machinery required for sorting, but also in terms of potential revenues from selling the recycled materials: the lower the quality of the final product, the lower the revenues (e.g. Rigamonti et al., 2015; Villanueva and Eder, 2014). On the contrary, tertiary routes like pyrolysis can adapt to high levels of waste contamination (Brems et al., 2012; Lofti, 2003), can replace multipurpose products with material and energy application (Gear et al., 2018) and prevent the recycling of chemical impurities along with the plastic material, thus becoming an attractive choice (Yu et al., 2016).

The selection of the plastic waste recycling route usually depends upon factors such as location, polymer type, product type, method of collection, presence of impurities, desired quality of the product and raw material prices offered by companies sorting or recycling the waste (Ragaert et al., 2017; Valentino, 2016; Yu et al., 2016). The export of recyclables is common in Europe, as the plastic recycling industry is characterised by a large number of entities, each performing a certain role in the recycling chain, linked by market agreements which are very volatile and depend on “who-offers-more” as well as the price of virgin materials (Villanueva and Eder, 2014). The main European importers of plastic waste are Germany, the Netherlands, and Italy (EC, 2018). Although an international market for recyclables may promote more recycling, international exchange of recyclable materials may also reduce transparency and traceability of waste materials, thereby making more difficult the accounting of environmental impacts and total management costs.

The environmental consequences of plastic waste recycling and management have been addressed through life cycle assessment (LCA) in numerous studies

in the literature, e.g. Al-Salem et al. (2014), Benavides et al. (2017), Eriksson and Finnveden (2009), Ferreira et al. (2014), Gear et al. (2018), Rigamonti et al. (2014), Shonfield (2008) and Valentino (2016). Two reviews (Bernardo et al., 2016; Lazarevic et al., 2010) compared existing studies and concluded that generally mechanical recycling is the environmentally preferred management option. However, as the net environmental benefits provided by recycling processes cannot stand alone as a basis for decision-making, an evaluation of financial considerations is also required. This may be achieved through Life Cycle Costing (LCC), which allows for consistent and integrated evaluation of both environmental (LCA) and financial aspects (Martinez-Sanchez et al., 2015; Swarr et al., 2011). Unfortunately, economic assessments focusing on recycling routes for source-sorted plastic waste are scarce in the literature (Lea, 1996; WRAP, 2009). None of these studies involved LCA. Furthermore, none of the studies in literature have addressed plastic waste collected at recycling centres.

Existing environmental assessments of plastic waste recycling in the literature have covered primarily plastic packaging collected at households, focusing only on a few polymer types (e.g. Arena et al., 2003; Da Cruz et al., 2012; Perugini et al., 2005), or have been included as a part of a broader waste system evaluation (e.g. Massarutto et al., 2011; Sadhukhan et al., 2016). Most LCA scenarios considered in these studies have addressed “macro-options” such as recycling vs. incineration vs. landfilling; only a few cases have reported detailed information about recycling processes (e.g. Shonfield, 2008). Existing economic analyses of plastic recycling have focused mainly on collection methods and approaches for increasing collection rates, as opposed to the recycling processes themselves (Beigl et al., 2004; Hage and Söderholm, 2008; Larsen et al., 2010; Massarutto et al., 2011). To improve plastic waste recycling further, specific focus on the environmental and economic performance of representative recycling options is needed while reflecting the actual composition and quality of plastic waste collected at recycling centres.

The overall aim of this study is to provide an integrated environmental and financial assessment of key recycling routes for hard plastic waste from Danish recycling centres. The intention is to provide an improved basis for decision-making in relation to plastic waste from recycling centres. This is achieved through environmental life cycle costing (eLCC) focusing on the following specific objectives: i) definition of arch-type recycling process configurations relevant for a European context, to illustrate key differences in approaches, ii) eLCC modelling of selected scenarios representing these arch-type recycling

configurations, iii) ranking of scenarios and identification of critical process parameters influencing the performance of each scenario, and iv) evaluation of the influence of the quality of post-consumer plastic waste on the results.

2 Materials and methods

2.1 Goal and scope definition

In an eLCC a financial analysis is complemented by an LCA for the same system. Therefore goal, scope, functional unit (FU) and system boundaries of the LCA and LCC must be identical (Hunkeler et al., 2008). The FU adopted in the modelling was the management of 1 tonne of post-consumer hard plastic waste collected at recycling centres and with the following composition: 53% PP, 19% PE, 2% PET, 1% PS, 10% other polymers and 15% impurities (including PVC corresponding to 6% of total PW) (data from Miljøstyrelsen, 2015, 2017b; Swaeco Danmark, 2015). Plastic films and PVC are collected separately from hard plastic at Danish recycling centres; any presence of these two plastic types was accounted for as misplacements (impurities) within the recycled plastic. The geographical scope was Denmark; the temporal scope was 2017-2030.

2.1.1 Definition of scenarios

This study addresses “typical” European solutions for the management of post-consumer plastic waste (i.e. leaving aside process details, as detailed evaluations have been provided in existing literature), therefore three arch-type system configurations were selected among those suggested in the literature (see Figure 1 and Appendix C for details): i) two mechanical recycling scenarios (one simple configuration and one involving a more advanced system) and ii) one feedstock recycling scenario (pyrolysis). The aim was not to include an exhaustive list of possible scenarios but rather to evaluate main types of recycling systems representing key differences in approach.

Scenario 1: simple mechanical recycling (sMR)

The plastic is sorted at a material recovery facility (MRF) comprising two near infra-red (NIR) scanners. The targeted plastic outputs are PP and PE, while the remaining part of the FU is incinerated. Other studies report HDPE and PET as MRF-targeted output (e.g. Sadhukhan and Martinez-Hernandez, 2017); however, in our case the PET fraction in plastic waste from recycling centres is too low to be targeted in a simple MRF. Recovered PP and PE enter individual reprocessing lines that include washing, grinding, ferrous and non-

ferrous metal separation, floating and pelletizing into recycled pellets substituting for virgin counterparts.

Scenario 2: advanced mechanical recycling (aMR)

The plastic is sorted at an MRF comprising a range of steps: manual sorting (aiming at separating PVC and impurities from the main stream), shredding, sieving, ferrous and non-ferrous metal separation and four NIR scanners. The targeted outputs are PP, PE, PET and PS. PVC and metal impurities separated out from the plastic flows are sent to further processing and recycling. Any remaining materials are incinerated. The plastic reprocessing steps include grinding, washing, drying and pelletizing into recycled pellets that substitute for corresponding virgin materials.

Scenario 3: Feedstock recycling (FR)

The plastic undergoes a manual sorting step whereby PVC is separated from the main stream (PVC must be < 2 % of the input to avoid processing problems

Figure 1. System boundaries for scenarios sMR (simple mechanical recycling), aMR (advanced mechanical recycling) and FR (feedstock recycling). Elec = electricity; NIR= near infra-red separator; MSWI = municipal solid waste incinerator. Light-grey boxes represent processing steps; dark-grey ovals represent co-products; white-dashed boxes represent avoided processes.

(Al-Salem et al., 2014; Elli and Pagani, 2017)). After being shredded to the right dimension (10 to 100 mm) and separated from metals, the pre-treated plastic waste is fed to a pyrolysis chamber. The waste is heated without oxygen until it breaks down into oils and gases, leaving a residue (char) rich in carbon and inorganic content from the original waste (Astrup and Bilitewski, 2010). The yield of gas and liquids depends on the specific pyrolysis system and the molecular composition of the plastic feedstock (Benavides et al., 2017), but generally exceeds 70% by weight (Alston and Arnold, 2011). Although not included herein, due to lack of data, some pyrolysis plants involve the use of catalysts to improve process efficiency (Benavides et al., 2017; Elli and Pagani, 2017). As catalyst amounts are low compared to the treated plastic (about 0.7% per weight) and can be regenerated many times (Benavides et al., 2017), this omission is found reasonable. The oil is used as secondary raw material displacing crude oil; the steam is utilised on-site to heat the pyrolysis chamber.

2.1.2 System boundaries

The system focuses on the sorting and reprocessing parts of the plastic waste recycling chain. Therefore, the system boundaries (Figure 1) start by unloading the collected PW at the MRF and end at the point of substitution of secondary materials or final management of any residual fractions. This excludes the collection phase at the recycling centres and transport between recycling centres and the MRF; these are identical for all scenarios and the exclusion has no consequences for the comparison between scenarios. For the two scenarios involving mechanical recycling (sMR and aMR), the model ends with the substitution of virgin plastic pellets. Similarly, the pyrolysis scenario (FR) ends by substituting crude oil based on the generated pyrolysis oil; further upgrading of the pyrolysis oil was assumed similar to the substituted crude oil and not included in the assessment.

2.2 Assessment method

The environmental and financial assessments followed a consequential approach to support decision-making and reflect potential changes in the management of plastic waste (Weidema et al., 2009; ISO14044, 2006). Co-products relative to the functional unit were addressed by system expansion and substitution (ISO 14040, 2006). The modelling was carried out with the dedicated waste LCA model EASETECH developed by DTU Environment, Technical University of Denmark (see Clavreul et al., 2014; Martinez-Sanchez et al., 2015).

2.2.1 Environmental assessment

The impact assessment followed the ILCD recommended method 2013 (Hauschild et al., 2013). Assessed impact categories included: global warming potential (GWP100), human toxicity (cancer, HTc, and non-cancer, HTnc), photochemical ozone formation (POFP), terrestrial acidification (TA), and terrestrial eutrophication (ET). No normalisation or weighting was applied. In the interpretation of results emphasis was placed on GWP, as this impact category is based on widely accepted concepts and well-established characterisation factors. Detailed results for all impact categories can be found in Appendix H.

2.2.2 Financial assessment

The financial part of the eLCC addressed monetary flows associated with the FU (i.e., how costs and revenues were affected by the FU as represented by the individual scenarios). As the assessment was not intended to evaluate monetary flows between concrete actors within the recycling chain, it was assumed that all necessary services were provided by a single actor. The assessment included budget costs and transfers but did not include externalities, according to Hunkeler et al. (2008). Budget costs included capital costs (expenditure for machinery, maintenance of equipment) and operational costs (manpower wages, energy and material consumption). Transfers included taxes, subsidies and fees applied to the recycling chain (mainly energy taxes, emission taxes, salary taxes and social security taxes). Costs were obtained for 2017 as a reference year. Please refer to Martinez-Sanchez et al. (2015) for further methodological details related to the LCC of waste management systems.

2.3 Uncertainty, sensitivity and scenario analysis

A number of approaches is available in the literature for sensitivity and uncertainty analysis (e.g. Clavreul et al., 2012; Groen et al., 2014; Laurent et al., 2014). In this study, uncertainty on the foreground system (including waste management activities) was addressed through global sensitivity assessment, which is not commonly used within waste management LCAs and was demonstrated to allow identifying important aspects governing LCA results (Bisinella et al., 2017); uncertainty on the background system (interacting with the foreground system by supplying energy and material inputs, including avoided energy and virgin material production) was addressed by scenario analysis.

2.3.1 Uncertainty analysis

In order to define the uncertainty associated with the input values, inventory data were inserted as ranges by defining all parameters describing the foreground system as probability distributions (Bisinella et al., 2017). Due to the diversity of the information available in the scientific literature, triangular distributions of the type TD (min,max,mode) were assumed. When the mode (the value appearing most often) could not be calculated, the median (the value separating the higher half from the lower half of the data sample) was used instead. The mode/median represented the baseline value used in the model, while the entire distribution was used for uncertainty propagation, where uncertainties associated with each input parameter were propagated to calculate the uncertainty associated with the results. Based on Monte Carlo analysis, input values were sampled from the probability distribution associated with each parameter, and the results were calculated following the approach of Bisinella et al. (2016).

2.3.2 Sensitivity analysis

Sensitivity analysis identifies how results vary as a consequence of a change in input values (Bisinella et al., 2016). It comprises a perturbation analysis, which is performed by increasing each parameter by 10% of its value in a “one-at-a-time” manner while keeping all other parameters fixed at their baseline values. Results from the perturbation analysis enable the calculation of sensitivity coefficients (SCs) and sensitivity ratios (SRs) associated with each parameter (Equations F.1 and F.2 in Appendix F). While the calculation of SRs provides information about the sensitivity of the model to each parameter, SCs are used to calculate and rank the contribution of individual parameters to total variance, thereby evaluating the importance of individual parameters relative to the results and on this basis identifying the most critical parameters.

2.3.3 Scenario analysis

A range of assumptions regarding the background system were necessary in order to complete the modelling: i) the type of energy provision, ii) the geographic location of facilities, and iii) the materials substituted by the recycled products. The robustness of the model to these assumptions was tested by scenario analysis. In the baseline situation, marginal electricity production was assumed to consist of wind and biomass and marginal heat production to represent natural gas boilers. In the scenario analysis, one “fossil” alternative was evaluated. Two alternative geographic locations for the facilities were evaluated in addition to the baseline situation (Denmark): Germany (scenarios

“sMR-DE”, “aMR-DE”, and “FR-DE”) and Italy (scenarios “sMR-IT”, “aMR-IT”, and “FR-IT”), as the two major importers of plastic waste in Europe. Regarding the selection of substituted virgin materials, an alternative situation for the sMR scenario was evaluated with the assumption that recycled plastic pellets substituted wood plastic composites (WPC) instead of virgin plastic pellets (scenario “sMR-WPC”). This alternative reflected a situation in which the purity of the separated plastic entering the reprocessing step was too low to ensure substitution of virgin polymers into pure plastic applications (Villanueva and Eder, 2014). Appendix G presents an overview of the considered parameters and their variations.

2.4 Life cycle inventory

Data applicable to Denmark were used when available; otherwise European data were applied.

2.4.1 Operational data

A clear relationship between operational parameters (sorting efficiencies, technical yields, substitution factors, energy and material consumption) and MRF/reprocessing facility configurations (number and types of individual unit processes) could not be found in the available literature. Therefore, data ranges were identified for relevant operational parameters and divided into two subintervals: a lower interval [min; mode] representing the simple MRF and an upper interval [mode; max] for the advanced MRF. As such, two subintervals representing individual distributions were defined for each parameter. The means for each of these subintervals (a "lower" mean and an "upper" mean) were used as baseline values in the sMR and aMR scenarios, respectively.

Table 1 lists the baseline values for sorting efficiencies, technical yields and market substitution factors of targeted polymers adopted in the sMR and aMR scenarios, as well as conversion efficiencies adopted in the FR scenario. The marginal electricity provision for Denmark was calculated according to Muñoz et al. (2015) with data from EC (2016) and consisted of 61% wind energy and 39 % energy from biomass. All residual fractions from the MRF (including unrecovered plastic and impurities) were assumed incinerated at MSWI with electricity and heat recovery (fly and bottom ashes were assumed landfilled, while iron scrap recovered from bottom ashes was assumed recycled). In the case of the feedstock recycling (FR) scenario, pyrolysis char was assumed landfilled. A large number of publications was used as data sources: see Appendix D and E for a full list of included inventory data and processes.

Table 1. Baseline values for sorting efficiencies, technical yields and market substitution factors adopted in scenarios simple mechanical recycling (sMR), advanced mechanical recycling (aMR) and feedstock recycling (FR). Additional information is provided in Table D.1.

Polymer	Sorting efficiency	Technical yield	Market substitution factor
Scenario sMR			
PP	55%	69%	0.48
PE	62%	76%	0.66
Total recovery		36%	
Scenario aMR			
PP	81%	86%	0.83
PE	84%	91%	0.91
PET	65%	88%	0.95
PS	65%	77%	0.66
Total recovery		67%	
Scenario FR			
Crude oil	-	80%	0.90
Steam	-	12%	1.00

2.4.2 Cost data

Cost data were retrieved from scientific literature and reports, converted to €/tonne of treated material as described in Martinez-Sanchez et al. (2015; 2016), and inserted as triangular distributions when possible. All budget costs were annualised and related to the annual capacity of the relevant facility, assuming 15000 tonnes/year (Clean, 2017). The cost of machinery was normalised by the annual usage rate of the machinery (Martinez-Sanchez et al., 2015). Discounting of future costs and benefits were not included, according to Larsen et al. (2010). The price of recyclable plastic was obtained by multiplying the market substitution factor with the price of the virgin counterparts. Transfers included energy taxes, emission taxes, salary taxes and social security taxes. Taxes on energy consumption excluded refundable taxes, levies and VAT that generally can be recovered by non-household users (Eurostat, 2017a; 2017b). Environmental taxes on energy included energy taxes, CO₂ taxes, NO_x taxes but not Public Service Obligation taxes as these are expected to be phased out within 2022 (IEA, 2017). VAT on capital goods was not included as this is typically recovered by business entities. Corporate income taxes were likewise excluded as most waste management facilities in Denmark are publicly owned and not allowed to generate profits. As costs

related to collection and transport prior to the MRF were beyond the assessment scope, these costs were not included (see Section 3.2). Detailed inventory cost data can be found in Table D.7 in Appendix D.2.

3 Results and discussion

3.1 Environmental assessment

The characterised impacts from the three scenarios are shown in Figure 2 for selected impact categories (results for additional categories are provided in Table H.1); the error bars represent variability in impact scores associated with uncertainty in the input data. Considering data uncertainties, scenario aMR provided significantly lower net impacts in all impact categories. While scenarios sMR and FR generally had higher impacts than scenario aMR, scenario sMR had significantly lower impacts than FR in four out of six impact categories. Across all impact categories and accounting data uncertainties, the scenario rank order was: 1) aMR, 2) sMR and 3) FR. While the error bars in Figure 2 indicate a wide range of potential impact scores for the individual scenarios, differences between individual scenarios for most of the impact categories were significant (except in the case of HTnc). This indicates that the conclusions regarding environmental impacts are relatively robust in view of known parameter uncertainties.

For scenario aMR, net savings (excluding uncertainty) in HTc and TA were 239% and 203% larger, respectively, compared with scenario sMR, and 74% and 326% larger compared with scenario FR. Consistent savings were observed for the impact categories POFP, GWP and ET, where the savings from scenario aMR were always at least 159% larger than scenarios sMR and FR. The process that contributed the most in the majority of impact categories was material substitution for scenarios sMR and aMR and the pyrolysis process itself in scenario FR. The difference between scenarios aMR and sMR (typically ranking second) was due to lower substitution of virgin plastic, because of lower recoveries and lower substitution factors associated with a lower quality of the recovered plastic.

In terms of GWP, the results showed that only scenario aMR provided savings (-717 kg CO₂eq/FU), whereas scenarios FR and sMR resulted in a net burden to the environment (374 and 940 kg CO₂eq/FU respectively). This ranking of scenarios is in accordance with a UK study (Al-Saleem et al., 2014) demon-

Figure 2. Characterised results for scenarios simple mechanical recycling (sMR), advanced mechanical recycling (aMR) and feedstock recycling (FR). “Other” in the legend includes landfill and building of facilities. GWP: global warming potential; POPF: photochemical ozone formation; TA: terrestrial acidification; ET: terrestrial eutrophication; HTc: human toxicity, cancer effects; HTnc: human toxicity, non-cancer effects; MSWI: municipal solid waste incinerator. Yellow diamonds represent net results; error bars represent the min and max results from the Monte Carlo analysis. See also Table H.1.

strating that if the recycling chain (MRF + reprocessing) had a substitution factor below 0.5, the pyrolysis technology would be environmentally preferable. Rigamonti et al. (2014) obtained results in the same order of magnitude for two mechanical recycling scenarios, while Shonfield (2008) demonstrated that mechanical recycling of PP, PE, PET and PVC would save around 500-700 kg CO₂eq/tonne of plastic waste. Shonfield (2008) and Al-Saleem et al. (2014) found a net load of 30-100 kg CO₂eq/tonne plastic for a pyrolysis scenario, considering the replacement of paraffin and naphta, while Gear et al. (2018) provided results of 1-62 kg CO₂/tonne plastic for a thermal cracking option, depending on the replaced product.

Avoided virgin material production was the largest contributor to GWP savings in scenario aMR (59% of total impacts), while in scenarios sMR and FR this accounted only for 15% and 11% of total impacts, respectively. Indeed, the simpler configuration of the MRF in scenario sMR increased the presence of untargeted polymers in the mixture to be recycled (the PP-stream and the PE-stream included 10% and 25% of untargeted polymers that were incorrectly sorted). When this happens, the recycler reprocesses the (impure) mixture at the highest melting point of the polymers, causing the degradation of components with lower melting points and the formation of burnout solids (Ragaert et al., 2017): the market substitution factor in the modelling lowered to take into account that these elements are undesirable in recycled materials, as they interrupt the structure of final products and reduce mechanical properties.

The largest contributor in scenario sMR was incineration which constituted 67% of the environmental impacts. This was an indirect effect of the low sorting efficiencies at the MRF (mainly due to the presence of a single technology handling a very heterogeneous mix) and of the low reprocessing yields (due to softening, surface contamination and biodegradation processes), causing 641 kg of the FU to be sent to energy recovery. As the baseline situation mainly included natural gas for heat production, CO₂ emissions from incineration (14% of total GWP impacts) were not outbalanced by avoided emissions from substituted energy production. Similar results have been documented by other studies (e.g. Alston and Arnold, 2011; Rigamonti et al., 2014). For the same reason, CO₂-eq emissions from the pyrolysis process represent an important contribution (71% of total impacts). From a climate impact perspective pyrolysis may therefore be similar to incineration processes; this supports the classification as tertiary recycling only when the

final product is used in material applications (new plastic, lubricants, etc.; Villanueva and Eder, 2014).

3.2 Financial assessment

Scenario aMR was the only scenario providing net financial savings (-90 €/FU). Scenarios FR and sMR ranked second and third, representing net costs of 16 €/FU and 87 €/FU respectively (Figure 2).

Avoided virgin material production was the largest contributor to the results (29%, 50% and 31% in scenarios sMR, aMR and FR respectively) and determined the net savings in scenario aMR: even if budget costs of aMR were larger compared to scenarios sMR and FR (+182% and +202% respectively), they were off-set by larger revenues provided by selling recycled pellets (+301% and +323% respectively). These revenues were larger in scenario aMR than in sMR not only because a larger quantity of plastic waste was processed into recycled pellets (671 kg in scenario aMR vs 359 kg in scenario sMR), but also because better material quality was obtained in scenario aMR than in sMR, resulting in higher homogeneity at the polymer level, which was reflected by higher prices. On the other hand, scenario sMR allowed larger revenues from electricity and heat production from energy recovery (+256%) than scenario aMR, albeit such revenues were outbalanced by incineration and landfilling costs from handling residues. The results demonstrated that insufficient sorting in scenario sMR may lead to a recovered plastic mixture for which recycling is not economically feasible (see also Section 3.3), as also suggested by Villanueva and Eder (2014). Ragaert et al. (2017) also reported that insufficient sorting can increase the total costs of recycled resins. When the quality of recycled pellets is not high enough to compete in the market, possible solutions include filler (e.g. talc) addition, compatibilisation, or virgin resins addition (Ragaert et al., 2017; Sommerhuber et al., 2016; Vazquez and Barbosa, 2016), which may increase costs further. Revenues from scenario FR came from substituting crude oil and heat (31% and 18% of total financial impacts, respectively). However, the pre-treatment and processing of plastic waste represented 14% and 33% of financial impacts respectively, leading to net costs for the FR scenario. Other studies have reached similar conclusions and reported that plastic waste pyrolysis is cost-effective only in the case of large facility capacities (e.g. Axion Consulting, 2014; Yu et al., 2016).

Overall, the budget costs represented larger contributions than transfers (93%, 95%, and 92% of absolute results in scenarios sMR, aMR and FR respectively). Taxes on labour costs represented the main part of transfers; environmental

taxes on energy consumption were almost negligible overall. Uncertainty in all input data made results vary between 40 and 141 EUR/FU, between -158 and -41 EUR/FU, and between -33 and 64 EUR/FU for scenarios sMR, aMR and FR respectively. While the observed variability in financial results represents the quality of input data, it also illustrates the importance of systematically including financial costs as part of an integrated assessment of recycling solutions – in view of the potential political sensitivity of decision-making.

As collection is not included in the assessment, total costs were smaller than identified by other studies (RDC Environment, 2003; Massarutto et al., 2011). However, collection and transport have been shown previously to have minor financial importance in the case of recycling centres (Larsen et al., 2010). Overgaard et al. (2018) documented the price for reprocessing Danish hard plastic waste to pellets (details unknown) at 80 EUR/tonne plastic waste, which is very similar to our result for scenario sMR. The obtained results were compared with the “management fee” that Danish waste collectors have to pay for treatment of plastic waste. Based on a range of Danish waste collectors, typical fees are estimated to be in the range of 0-200 €/tonne plastic waste. While this range includes the results found for scenarios sMR and FR (although pyrolysis plants are not common in Europe), the large variability in prices also illustrates the volatility of the recycled plastic market. As scenario aMR indicates net savings, and considering the substantial variability in MRF performance across Europe, current Danish management fees suggest that considerable improvements should be possible. This also indicates that by moving toward higher quality in terms of output and recycling, the management fees could be lowered or turned negative, i.e. representing a revenue for municipalities collecting plastic waste, as it is now the case for metals.

3.3 Sensitivity and scenario analysis

3.3.1 Importance of process parameters

The contribution of individual parameters to the overall variance of the results is illustrated in Figure 3, including the information about the three most critical parameters for each scenario. Overall, three to nine parameters were sufficient to achieve more than 90% of the total variance in the environmental and financial results, although for some impact categories and scenarios only three parameters were sufficient (e.g. GWP and ET). Scenarios sMR and FR appeared to be affected by fewer parameters than aMR. This difference depended partly on the scenario modelling itself, but also on the complexity of

Figure 3. Contribution of individual parameters to overall variance of environmental and financial results (top to bottom: scenarios sMR, aMR and FR). Values in the legend represent the three most important parameters per impact category and the share of variance covered by the corresponding parameter type. GWP: global warming potential; POFP: photochemical ozone formation; TA: terrestrial acidification; ET: terrestrial eutrophication; HTc: human toxicity, cancer effects; HTnc: human toxicity, non-cancer effects; TY: technical yield; SE: sorting efficiency; SF: substitution factor; PP: polypropylene; PE: polyethylene; DM: distance to MRF; DR: distance to recycling; EE: electricity efficiency at MSWI; vPP: virgin PP price; ExP: price of extruder; vPE: virgin PE price; CO₂: carbon dioxide emissions at pyrolysis plant; St: steam; EC PYR: energy consumption at pyrolysis plant; O: oil; GS: gas share in energy consumption at pyrolysis plant; OP: oil price.

the technologies within the scenarios. The relatively few parameters representing most of the total variance indicate that the best possible quality of these data should be ensured, and that research efforts improving related technology aspects are important. In this study, these parameters relied on data from a review of 9-17 studies (see Tables D.1 and D.2). Data in the legend refer to the first three data points in Figure 3 for a specific impact category and scenario (see also Appendix I for additional results): for most impact categories the more contributing parameters were related to sorting efficiency, technical yield and market substitution factor.

As the market substitution factor may be challenging to estimate (see also Vadenbo et al., 2017), Figure 4 shows the effect on GWP and total costs when this parameter varies between 0 and 1. The value for the market substitution factor could determine whether the final results were positive or negative, most prominently for the financial assessment and for scenario aMR (steeper relationships). This clearly illustrates that the financial viability of recycling depends on the market acceptance of the recycled pellets, as all scenarios achieved net financial revenues in case of market substitution factors above 0.6-0.7. Market aspects such as image, risk of failure, poor characteristics (odours, touch, appearance), difficulty in guaranteeing capacity and demand, legislative barriers may significantly influence the acceptance of recycled pellets (RDC Environment, 2003). While some of these challenges may be addressed (poor image can be reversed and legislation may change), others may constrain the recycled plastic market. This demonstrates that both environmental and financial assessments should include the market substitution factor as a parameter distribution rather than a single value.

It is important to realise that linearity between selling prices and quality of recyclates is often assumed (as in this study, see Section 2.4.2). However, the linearity between the two variables may not apply or may be "interrupted" at some point, so that recycled pellets are sold with no revenues when the quality decreases below a certain threshold, as in the case of municipal subsidies for collection of recyclables that fall to zero in the case of large contamination of the collectables (Rigamonti et al., 2015). If this is the case also for raw material substitution, the economic expenditure required for scenario sMR could be even higher. However, as LCC and LCA must share the same methodological approach, and as no evidence could be found that such a relation exists for Danish plastic waste, an identical substitution factor was applied to both parts of the assessment.

Figure 4. Behaviour of Global Warming Potential (solid lines; left-hand axis) and total costs (dotted lines; right-hand axis) with decreasing degree of virgin plastic substitution.

As Figure 3 illustrates, financial results depended largely on the price of the substituted virgin plastic. In this study a price of 1200 €/tonne was chosen as a baseline, but it is important to remember that this value is closely linked to oil prices which are extremely volatile. Decreasing oil prices will decrease the price of virgin resins, and prices of recycled pellets would have to decrease accordingly in order to be competitive; however, the cost of recycling would remain constant. Changes in financial results when decreasing the price of virgin resins to 1000 €/t are substantial: +63% in scenario sMR (109 EUR/FU) and +47% in scenario aMR (-65 EUR/FU).

3.3.2 Importance of boundary conditions and background system

Results from scenario analyses are provided in Figure 5 for GWP and financial costs. The ranking of scenarios did not change when the energy provision changed to fossil sources: scenarios FR and aMR were affected only slightly by this choice (-15% and +8% of emissions respectively). In this situation, the use of thermal energy in the pyrolysis process (natural gas) became important, and the larger savings in scenario aMR provided by energy substitution from incineration were outbalanced by larger emissions from energy consumption for sorting and processing plastic waste. Emissions in scenario sMR were 84% smaller than in a renewable energy system, due to the large savings from recovery of energy replacing fossil fuels. In this situation, increased attention must be given to the choice of electricity and heat efficiencies, as electricity

efficiency was one of the main parameters in the baseline modelling (see Figure 3), and changing to a fossil-based electricity mix will increase the importance of this parameter. In Europe, electricity and heat recovery at waste incinerators typically range between 6-28% for electricity and 6-74% for heat (Alston and Arnold, 2011; Eriksson and Finnveden, 2009; Turconi et al., 2011).

The scenario analysis on the location of the recycling process demonstrated that exporting plastic waste resulted in higher GHG impacts and total costs. The increase in GWP can be explained by three factors: transporting plastic waste for longer distances, non-renewable energy in the country-specific electricity mix, and smaller credits obtained from the substitution of energy due to lower energy recovery efficiencies. The main financial contributor was transportation cost which increased from 5%, 3% and 3% in scenarios sMR, aMR and FR to 26%, 17% and 28% in scenarios sMR-IT, aMR-IT and FR-IT and to 13%, 8% and 12% in scenarios sMR-DE, aMR-DE and FR-DE respectively. As also reported by Groot et al. (2014), the lightness and bulkiness of plastic makes cost-effective transport challenging. Nevertheless, the ranking of financial results resembles the baseline case: scenario sMR shows a net expenditure (109 EUR/FU in both scenarios sMR-IT and sMR-DE), scenario aMR provides consistent savings (-216 EUR/FU in scenario sMR-IT and -316 EUR/FU in scenario sMR-DE), and scenario FR is close to a financial zero (-3EUR/FU in scenario sMR-IT and -6 EUR/FU in scenario sMR-DE). However, if transportation costs are assumed associated to the exporting country (i.e Denmark), then depending on the choice of the recycling scenario the importing country may bear a cost or obtain revenues, while the exporting country may always bear a cost.

Figure 5. Global Warming Potential 100 (left-hand graph) and total costs (right-hand graph) results for scenario analysis on the provision of electricity (red cross symbol), the geographic location of the facility (black triangular symbol and grey circle symbol), and the substitution of WPC (grey square symbol).

The last system assumption investigated by scenario analysis evaluated the results for scenario sMR in case the recycled pellets were used in wood plastic composites (WPC) rather than recycling into plastic products. The results from the environmental assessment did not change, as the substituted material (virgin plastic pellet) did not change. However, the total costs increased from 65 EUR/FU in the baseline scenario to 229 EUR/FU in scenario sMR-WPC (+252%). This increase was caused by the lower prices that recycled pellets reach as raw material for the production of WPC (or other types of composites): the plastic is used as filler and its aesthetics/purity are less important compared with "pure" plastic application. This clearly illustrates that higher overall costs may be associated with recovering and recycling lower-quality plastic.

4 Conclusions

Drawing upon the comparison of three key arch-types of hard plastic waste recycling routes (two mechanical recycling scenario characterised by a simpler – sMR – and a more advanced – aMR – technology, respectively and one feedstock recycling scenario by pyrolysis – FR) we showed that quality-oriented recycling is preferable from both an environmental and financial assessment perspective. Scenario aMR provided the largest savings in the highest number of impact categories, including global warming potential (GWP). Scenario sMR typically ranked second, apart from the case of GWP where the pyrolysis alternative appeared more beneficial than a simple material recovery facility (MRF) treatment due to considerable emissions from incinerating a large amount of plastic rejects. Scenario aMR also provided the lowest total costs (negative costs), followed by FR and sMR scenarios (positive costs). The results demonstrated that if high quality of the recycled plastic is achieved, then both environmental savings and financial revenues are possible. Therefore, an advanced MRF treatment should be preferred for management of hard plastic waste from recycling centres. Typically, three to nine model parameters were sufficient to account for more than 90% of the total variance in the results: increased efforts should be put to model such parameters correctly, as the results will be largely dependent on them. The most important parameters in the highest number of impact categories were related to sorting efficiencies, technical yields and market substitution factors; these parameters should be addressed specifically (i.e. maximised) when increasing the environmental and financial savings is targeted. The ranking of scenarios was found to be robust with respect to assumptions on energy provision, location of recycling facilities and application of the recycled material (addressed by

scenario analysis). The resource quality was identified as a driving force towards economically and environmentally sustainable recycling, indicating that from a circular economy perspective "better recycling" may be preferable over "more recycling".

Acknowledgment

The Danish Environmental Protection Agency is highly appreciated for funding this study. The authors are also grateful to U. Arena (II University of Naples), S.A. Bassi and V. Bisinella (Technical University of Denmark), I. Puig (Fundació ENT) and A. Villanueva (Joint Research Centre) for useful comments on the study.

Funding

This work was supported by the Danish Environmental Protection Agency (Miljøstyrelsen).

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