Modeling material flows through plastic recycling chains

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LIST OF ABBREVIATIONS

ABS	Acrylonitrile Butadiene Styrene
APW	Agriculture plastic waste
ASR	Automotive shredder residue
BC	Building and construction cost
BFR	Brominated flame retardant
CAPEX	Capital expenditure
CBA	Cost benefit analysis
CDW	Construction and demolition waste
CFA	Cooling and freezing appliances
CPA	Circular plastic alliance
CR	Chemical recycling
C&I	Commercial and industrial
DfR	Design-for-Recycling
DRS	Deposit refund scheme
DSC	Differential scanning calorimetry
DSD	Duales system Deutschland
EC	European Commissions
EEE	Electrical and electronic equipment
EPR	Extended producer responsibility
ELV	End-of-life vehicle
EoL	End-of-life
EoL-RR	End-of-life recycling rate
EPS	Expanded polystyrene
EPMC	Engineering and project management costs
EU	European Union
EVOH	Ethylene vinyl alcohol
E-waste	Electronic waste
FTS	Fischer Tropsch synthesis
FTIR	Fourier-transform infrared spectroscopy
FU	Functional unit
HDPE	High-density polyethylene
IC	Installation costs (of equipment)
Kt	Kilotonne
LCC	Life cycle costing
LDPE	Low-density polyethylene
LHA	Large household appliances
LLDPE	Linear low-density polyethylene
MFA	Material flow analysis
MIOA	Multivariate input-output analysis
MPO	Mixed polyolefin
MR	Mechanical recycling
MRF	Material recovery facility
Mt	Million tonnes
MTO	Methanol-to-olefin
NIAS	Non-intentionally added substances
NIR	Near-infrared

NTCP	Nationaal Testcentrum Circulaire Plastics
PA	Polyamides
PC	Polycarbonate
PCR plastic	Post consumer recycled plastic
PCFP	Post-consumer flexible packaging
PE	Polyethylene
PET	Polyethylene Terephthalate
PIR plastic	Post industrial recycled plastic
PMMA	Polymethyl Methacrylate
PO	Polyolefins
PoC	Price of equipment
POM	Placed on the market
POP	Persistent organic pollutant
PP	Polypropylene
PPWD	Packaging and Packaging Waste Directive
PPWR	Packaging and Packaging Waste Regulations
PRO	Producer responsibility organization
PS	Polystyrene
PTTs	Pots, trays, and tubes
PUR	Polyurethane
PVC	Polyvinyl Chloride
P2P	Plastic-to-plastic
P2C	Plastic-to-chemicals
P2F	Plastic-to-fuel
QRP	Quality recycling process
RBR	Recyclability benefit rate
RC	Recycled content
RoHS	Restriction of Hazardous Substances
RVC	Robotic vacuum cleaner
SBR	Solvent-based recycling
SHA	Small household appliances
SUPD	Single-use Plastic Directive
TC	Transfer coefficient
TD	Triangular Distribution
TEA	Techno economic assessment
TRL	Technological readiness level
T1	Tier 1
T2	Tier 2
OPEX	Operational expenditure or operational costs
OVAM	Openbare Vlaamse Afvalstoffenmaatschappij
VLAREMA	Vlaams Reglement betreffende het duurzaam beheer van Materiaalkringlopen en Afvalstoffen
WFD	Waste Framework Directive
WEEE	Waste electronic and electrical equipment
WEEP	Waste electronic and electrical plastic
XRF	X-ray fluorescence
XRT	X-ray transmission

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SUMMARY

The unprecedented growth of plastic demand in our daily lives is inevitable because of its broad applications, such as for food and beverage packaging, home piping and insulation, lightweight automotive or electronic parts, and horticulture products (agriculture film or gardening pots). The broad application of plastic owes to its excellent performance, such as high moldability, light weight, resistance against chemical and mechanical degradation, low production cost, and excellent barriers against aspects like oxygen, water vapor, microorganisms, and carbon dioxide. Nevertheless, when becoming waste, plastic is often managed in unsustainable ways causing adverse environmental impacts. Plastic recycling rates are still low due to a lack of adequate waste management infrastructure such as separate waste collection at source (e.g., at households, schools, offices, restaurants, etc.), material sorting, and recycling. The low plastic recycling rates can also be caused by the fact that recycling plastic is challenging because of the complex material structures (e.g., different polymer grades, multimaterial products, contamination during the use phase, etc.), and mix of (Non) Intentionally Added Substances (NIAS), including inherited hazardous legacy chemicals (e.g., flame retardants), which affects recycled plastic quality. In addition, plastic recycling operations are not always economically feasible and self-sustaining, because recycled plastic has a low selling price due to its inferior quality compared to virgin plastic. However, in Europe, ambitious targets have been set out by the European Commission alongside voluntary pledges made by the European plastic industry, which aim to increase plastic recycling into high-value secondary materials suitable for broader end market applications. Thus, it is evident that there is an urgent need to improve the *status quo* of plastic waste management, starting by identifying the main bottlenecks, exploring technological options, and evaluating the economic and environmental aspects of the recycling chains.

In this PhD research, the potential improvements within plastic recycling chains in Europe are investigated by building and applying material flow analysis (MFA) models in combination with several circularity indicators, cost benefit analysis and life cycle assessment.

First, Chapter 1 of this PhD research provides a scientific literature review on the current state-of-the-art plastic waste management system (*status quo*) in Europe, including a few key (provisional) regulations and pledges made by the European Commission and the plastic industry.

In Chapter 2 of this research, a MFA model is developed and used to investigate the current and (potential) future performance of plastic waste management in Europe. The main objective of the MFA model in Chapter 2 is to measure the contribution of various plastic recycling technologies to increase plastic circularity and achieve recycling targets in Europe. This research considers improvements within plastic recycling chains (e.g., improved mechanical recycling (MR) yield), as well as new emerging recycling technologies, including chemical recycling (CR) and solvent-based recycling (SBR). The analyses are done by tracing the flows of the ten most used plastic types in Europe in five key waste generating sectors in Europe. The ten polymers considered in this analysis are (Linear) Low-Density Polyethylene (LLDPE), High-Density Polyethylene (HDPE), Polypropylene (PP), Polyethylene Terephthalate (PET), Polystyrene (PS), Expanded Polystyrene (EPS), Polyvinyl Chloride (PVC), Acrylonitrile Butadiene Styrene (ABS), Polyurethane (PUR), and Polyamide (PA). The five key plastic waste generating sectors are packaging, building and construction, automotive, electronic, and agriculture sector. For the purposes of measuring potential improvements in the European plastic circularity, one baseline scenario (*status quo* in 2018, as the benchmark) and six future scenarios in 2030 are considered. One of the future scenarios (in 2030) only considers the improvement of source separation of plastic, sorting, and mechanical recycling (MR), while the remaining scenarios consider the combination of MR, CR, and SBR to deal with plastic waste in Europe. One of the future scenarios also investigates the impact of processing so-called 'missing plastic', i.e., plastic waste generated but not accounted for in statistics, towards achieving plastic recycling targets in Europe. Next, five evaluation indicators, namely end-of-life recycling rate (EoL-RR), plastic-to-plastic rate (P2P), plastic-to-chemical rate (P2C), and plastic-to-fuel rate (P2F), are selected to interpret and compare the MFA modeling results. Moreover, an uncertainty propagation analysis is carried out to measure the modeling output uncertainties (i.e., the mass balance and evaluation indicators. Lastly, the potential plastic recycled content is estimated by quantifying the ratio between the uptake of recycled plastic produced from MR, CR, and SBR (per sector) over the total projected plastic demand (per sector) in Europe in 2030. Given the modeling parameters and considered scenarios in the MFA model, it is evident that new emerging plastic recycling technologies such as CR and SBR positively contribute to plastic circularity and to achieve recycling targets in Europe. In the most positive scenario, the highest EoL-RR is achieved at 80%, in which 61% is P2P and 19% is P2C. In all scenarios, the P2F ranges from 3–6%. The mass balance accounting of the uptake of recycled plastic (per sector) over the projected plastic demand in 2030 suggests that closed-loop recycling, processing the 'missing plastic', and emerging plastic recycling technologies (CR and SBR) will be necessary to achieve the recycled content targets.

Next to building and developing a prospective MFA at the European level (as the system boundary), the MFA model is also applied to assess potential improvements in plastic recycling from two key sectors (as case studies) in Europe, namely the electronic sector and packaging sector. In Chapter 3, the MFA model is applied to quantify the amount of recycled plastic production in Belgium and The Netherlands from three selected electrical and electronic equipment (EEE) within the small household appliances (SHA) group, namely vacuum cleaners, coffee machines, and electric shavers. The three products are selected because of their high plastic content in EEE. For the chosen EEE, a multivariate input-output analysis (MIOA) model is developed and applied to better estimate the amount of waste electrical and electronic equipment (WEEE), which is later combined with MFA to estimate the potential recycled plastic production from WEEE. The MIOA model is chosen because it considers the dynamic interconnection between EEE placed on the market (sales), stock flows (i.e., EEE accumulation in the market), and product lifespan distribution. In addition to combining MIOA and MFA, a sensitivity analysis is also carried out to identify the main bottlenecks and potential improvements within the WEEE recycling chains. Through mass balance accounting and analyzing WEEE composition, the following facts were studied: i) the amount of recycled plastic released from the selected WEEE recycling, ii) recycled content availability for the selected (W)EEE, and iii) the inherited hazardous legacy chemicals content from the selected (W)EEE. The model results indicate that improving the selective collection of WEEE and pre-processing steps to recover plastic from WEEE (dismantling, shredding, and material sorting) are key to reaching the EEE recycling target. Moreover, the model predicts that the inherited hazardous legacy chemicals from WEEE can still be found in 2030 (up to 5% by weight), albeit the restriction of hazardous substances has been put into force since 2006. This can be explained by observing the lifespan distribution of the selected EEE products and MIOA results, which suggest that a considerable amount of (W)EEE (up to 10% by weight) purchased from 1990 – 2006 could still be discarded by 2030.

The next case studies presented in this research investigate the recycling performance (from technical and economic perspectives) of the so-called difficult-to-recycled plastic packaging formats: flexible packaging waste. Specifically, this PhD research investigates potential improvements in flexible packaging waste management from households (e.g., individual houses) and non-household sectors (e.g., stores, warehouses, factories, etc.). Chapter 4 of this PhD investigates the recycling performance (from technical and economic perspectives) of an improved mechanical recycling process for flexible packaging waste from households. For this purpose the recycling performance of an improved mechanical recycling process developed (and proposed) by the industry called Quality Recycling Process (QRP) is investigated, and compared with a conventional mechanical recycling process for flexible packaging waste in Europe. The developed MFA starts from two sorted bales (e.g., bales rich in PE film or polyolefin film) after household waste sorting at a German material recovery facility (MRF), which can be recycled either through QRP or a conventional mechanical recycling process. The MFA model is used to trace the flows of materials such as flexible packaging waste and non-flexible packaging waste (e.g., rigid packaging or residue). The MFA results assessed by applying four evaluation indicators related to quantity (process yield and net recovery indicators) and quality (polymer grade and transparency grade). Next, the economic performance of QRP and conventional mechanical recycling is estimated by combining the MFA results (mass balances) and economic modeling parameters such as utility consumption (e.g., electricity, fuel, water, etc.), capital investment, and regranulate sales. Moreover, sensitivity analyses are carried out to assess potential variations of the selected modeling parameters toward the model results (i.e., evaluation indicators and economic balance). This research indicates that QRP for flexible packaging waste from households performs similarly to conventional mechanical recycling, yet QRP produces a higher regranulate quality. From an economic perspective, QRP improves the economic balance of flexible packaging waste recycling, and QRP can be important in increasing plastic circularity for flexible packaging.

Next to household waste, the economic feasibility study of collecting and mechanical recycling for flexible packaging waste from non-household sector is investigated in Chapter 5. The economic feasibility is assessed by building a cost benefit analysis (CBA) model starting with a logistic simulation of selective collection, followed by a MFA of mechanical recycling, and its associated economic aspects (i.e., capital investment, annual costs and reganulate sales). Particularly, this research considers urban areas of the City of Ghent in Belgium, and its twelve neighboring municipalities as a case study (system boundary). The CBA of flexible packaging from non-household sectors starts by characterizing the composition and estimating the waste quantity generated per business activity (also called NACE sector in Europe) in the considered urban areas in this study. The waste quantity is estimated by real sampling in 2018 in the City of Ghent, combined with extrapolation with Orbis databases that provides data on the number of companies per region. The waste compositions are characterized during real sampling campaigns between December 2021 – February 2022. The waste sampling campaigns cover five key non-household sectors, such as Wholesale (e.g., NACE G.46), Retail (e.g., NACE G.47), Construction (e.g., NACE F.41), Logistics (e.g., NACE H.49), and '*other*' sectors (e.g., NACE C. 10, NACE C.18, etc.). Once the waste composition and quantity have been estimated, the annual costs of selective collection costs in three different collection frequencies (weekly, fortnightly, and monthly) are estimated using the OptiFlow© software. Furthermore, the material flows and economic balance of mechanical recycling of non-household flexible packaging waste are investigated, including the estimated revenue from regranulates sales. Lastly, the carbon footprint associated with collecting and mechanical recycling of non-household flexible packaging waste is estimated, and compared with the baseline scenario (i.e., virgin PE granulate production with incineration as EoL treatment). This research suggests that collecting and mechanical recycling of non-household flexible packaging waste strategy is contingent upon waste collection frequencies, achieving minimum waste quantity, and maintaining high-quality feedstock (waste quality to be processed through mechanical recycling).

In chapter 6 of this PhD research, a preliminary assessment of recycled content availability for flexible packaging (household and non-household) in Europe is conducted, assuming 100% closed-loop recycling from flexible packaging waste back into new flexible packaging. The new proposed Packaging and Packaging Waste Regulations (PPWR) consists of mandatory minimum recycled content targets for flexible packaging, i.e., setting the ambition at 35% recycled content target for non-contact-sensitive and 10% recycled content target for contact-sensitive flexible packaging in 2030. For the purpose of assessing the feasibility to meet the recycled content target, a MFA model is developed to trace the fate of flexible packaging waste throughout the end-of-life treatment and regranulates production in Europe in 2030. The MFA model is built assuming that the in 2030 the flexible packaging design will be improved (e.g., from multi- to mono-material flexible packaging), more selective collection for flexible packaging (e.g., more P+MD collection in Europe), sorting techniques will perform better (e.g., 'smart' sorting techniques using digital watermarks or artificial intelligence), and better mechanical recycling and pyrolysis yield (e.g., advanced mechanical recycling like QRP or catalytic pyrolysis). Furthermore, for the purpose of modeling future end-of-life treatment, five scenarios are developed and investigated, consisting only mechanical recycling for flexible packaging as well as mechanical recycling and pyrolysis as complementary techniques to reach the recycled content target set out in Europe. Furthermore, the capital investment with achieving the recycled content targets are estimated, which is developed based on the MFA model results combined with economic factors (in €/tonne) found in literature. The MFA results suggest that the recycled content targets can be achieved by using mechanical recycling and pyrolysis as complementary techniques to deal with flexible packaging waste. In the most positive scenarios, €7.7 – 8.8 billion of capital investment would be needed to build mechanical recycling and pyrolysis infrastructure, including pretreatment and hydrotreatment for pyrolysis. The MFA results also indicate a trade-off between achieving higher-quality of regranulates to meet 10% recycled content target for flexible packaging (assuming pyrolysis would become a more dominant technique to achieve the target), and annual regranulates production (i.e., quantity of secondary materials). As results, the overall end-of-life recycling rate for flexible packaging could drop when the contact-sensitive recycled content targets are achieved, assuming that pyrolysis would become a more dominant technology to produce higher-quality regranulates suitable for contact-sensitive applications in Europe such as for example for food packaging.

Finally, in the last Chapter 7, the general conclusion of this PhD research is presented, including the future perspective on improving the developed MFA model and plastic circularity. The importance of expanding the system boundaries to other regions (e.g., other countries within or beyond Europe), product categories (e.g., other EEE), and sectors (e.g., automotive sector) are highlighted. The applied MFA model can be improved by developing more scenarios (e.g., implementing new designs or business models like repair or reuse) and more detailed material composition characterization. In addition, the quantity-based MFA models presented in this PhD research can be improved by linking in more quality aspects of the regranulates, which becomes an important step considering a wide range of technical requirements for end market applications. Finally, the remaining challenges and opportunities for further research around the sustainability aspects of plastic are discussed.

Overall, this PhD research shows that plastic circularity needs to improve in Europe. MFA can serve as a tool to assess the status quo and measure potential improvements within the European plastic recycling chain by developing potential circularity scenarios. MFA can provide insights into how significant the improvements could be (compared to the baseline scenario) and potential extra capital investment and costs. Finally, the MFA presented in this PhD research can be used to formulate plastic recycling strategies (e.g., minimum recycled content targets) based on science-based projections as well as to monitor the attainment of the plastic recycling targets (e.g., plastic recycling rates) in Europe.

SAMENVATTING

De ongekende toename van de behoefte aan plastics in ons dagelijks leven is onvermijdelijk door het uitgebreide gebruik van kunststoffen in verschillende domeinen, waaronder verpakkingen van voedingsmiddelen en dranken, leidingen en isolatie voor huishoudelijk gebruik, automobiel- en elektronische componenten, maar ook tuinbouwproducten zoals landbouwfolie of tuinpotten.

Het uitgebreide gebruik van polymeren wordt toegeschreven aan hun opmerkelijke functionaliteit, waaronder opmerkelijke buigzaamheid, licht gewicht, weerstand tegen chemische en mechanische degradatie, lage productiekosten en uitstekende barrière-eigenschappen voor zuurstof, waterdamp, micro-organismen en kooldioxide. Toch worden plastics na hun gebruik vaak op een niet-duurzame manier behandeld, wat nadelige gevolgen heeft voor het milieu. Recyclagecijfers voor plastics zijn nog steeds laag door een gebrek aan adequate infrastructuur voor afvalbeheer, zoals afvalcollectie aan de bron (bv. bij huishoudens, scholen, kantoren, restaurants, enz.), sortering en recyclage. De lage recyclagepercentages van kunststoffen kunnen ook te wijten zijn aan het feit dat het recycleren van kunststoffen een uitdaging is omwille van de complexe materiaalstructuren (bv. verschillende polymeerkwaliteiten, producten die bestaan uit verschillende materialen, vervuiling tijdens de gebruiksfase, enz.) en de aanwezigheid van niet-intentioneel toegevoegde stoffen (*non intentionally added substances*, NIAS), waaronder historisch toegevoegde gevaarlijke chemische stoffen (bv. vlamvertragers), die de kwaliteit van gerecycleerde kunststoffen beïnvloedt. Bovendien zijn bepaalde recyclageprocessen niet altijd economisch haalbaar, omdat gerecycleerd plastics een relatief lage verkoopprijs hebben door de inferieure kwaliteit in vergelijking met nieuwe plastics. In Europa heeft de Europese Commissie echter ambitieuze doelen vooropgesteld, samen met niet-bindende doelstellingen van de Europese kunststofindustrie, om meer plastics te recycleren tot hoogwaardige secundaire materialen die geschikt zijn voor bredere eindmarkttoepassingen. Het is dus duidelijk dat de status quo van het beheer van kunststofafval dringend moet worden verbeterd, te beginnen met het identificeren van de belangrijkste knelpunten, het onderzoeken van technologische opties en het evalueren van de economische en milieuaspecten van de recyclageketens.

In dit doctoraatsonderzoek worden de potentiële verbeteringen binnen plasticrecyclageketens in Europa onderzocht door het bouwen en toepassen van materiaalstroomanalyse (MFA) in combinatie met verschillende circulariteitsindicatoren, kosten-batenanalyses en levenscyclusanalyses.

Allereerst wordt in hoofdstuk 1 van dit doctoraatsonderzoek een wetenschappelijke literatuurstudie gegeven over het huidige systeem voor plastic afval (*status quo*) in Europa, inclusief enkele belangrijke (voorlopige) regelgevingen en toezeggingen van de Europese Commissie en de kunststofindustrie.

In hoofdstuk 2 van dit onderzoek wordt een MFA-model ontwikkeld en gebruikt om de huidige en (potentiële) toekomstige prestaties van het recyclagesysteem in Europa te onderzoeken. Het hoofddoel van het MFA-model in hoofdstuk 2 is het meten van de bijdrage van verschillende kunststofrecyclagetechnologieën aan het vergroten van de circulariteit en het behalen van de recyclagedoelstellingen in Europa. Dit onderzoek houdt rekening met verbeteringen in plasticrecyclage (bv. verbeterd rendement van mechanische recyclage (MR)), maar ook met nieuwe opkomende recyclagetechnologieën, waaronder chemische recyclage (CR) en solvent-gebaseerde recyclage (SBR). De analyses zijn gedaan door de stromen van de tien meest gebruikte plastics in Europa te traceren in vijf belangrijke afvalproducerende sectoren in Europa. De tien polymeren die in deze analyse worden meegenomen zijn (lineair) lagedichtheidpolyethyleen (LLDPE), hogedichtheidpolyethyleen (HDPE), polypropyleen (PP), polyethyleentereftalaat (PET), polystyreen (PS), geëxpandeerd polystyreen (EPS), polyvinylchloride (PVC), acrylnitril-butadieen-styreen (ABS), polyurethaan (PUR) en polyamide (PA). De vijf belangrijkste kunststofafvalproducerende sectoren zijn de verpakkingssector, de bouwsector, de automobiel sector, de elektronicasector en de landbouwsector. Om de potentiële verbeteringen in de Europese kunststofcirculariteit te meten, worden één basisscenario (*status-quo* in 2018, als benchmark) en zes toekomstscenario's in 2030 bestudeerd. Een van de toekomstscenario's (in 2030) beschouwt alleen de verbetering van bronscheiding van plastics, sortering en mechanische recyclage (MR), terwijl de overige scenario's de combinatie van MR, CR en SBR beschouwen om kunststofafval in Europa te behandelen. Een van de toekomstscenario's onderzoekt ook de impact van de verwerking van zogenaamde 'ontbrekende kunststoffen', d.w.z. kunststofafval dat wordt geproduceerd maar niet in de statistieken wordt opgenomen, op het behalen van de doelstellingen voor plasticrecyclage in Europa. Vervolgens worden vijf evaluatie-indicatoren geselecteerd, namelijk het recyclagepercentage aan het einde van de levensduur (EoL-RR), het percentage plastic naar plastic (P2P), het percentage plastics naar chemicaliën (P2C) en het percentage plastics naar brandstof (P2F), om de resultaten van de MFA-modellering te interpreteren en te vergelijken. Bovendien wordt een onzekerheidsanalyse uitgevoerd om de onzekerheden in de modeluitvoer te meten (d.w.z. de massabalans en evaluatie-indicatoren). Tot slot wordt de potentiële hoeveelheid aan gerecycleerde plastics geschat door de verhouding te kwantificeren tussen de opname van gerecycleerde plastics geproduceerd uit MR, CR en SBR (per sector) en de totale verwachte vraag naar plastic (per sector) in Europa in 2030. Gezien de modelparameters en de beschouwde scenario's in het MFA-model is het duidelijk dat nieuwe opkomende recyclagetechnologieën zoals CR en SBR positief bijdragen aan de circulariteit van plastics en aan het behalen van de recyclagedoelstellingen in Europa. In het meest positieve scenario wordt de hoogste EoL-RR bereikt van 80%, waarin 61% P2P en 19% P2C is. In alle scenario's varieert de P2F van 3 tot 6%. De massabalans van de opname van gerecycleerde plastics (per sector) ten opzichte van de verwachte vraag naar plastics in 2030 suggereert dat “closed loop” recyclage, verwerking van de “ontbrekende plastics” en opkomende recyclagetechnologieën (CR en SBR) nodig zullen zijn om de streefcijfers voor de gerecycleerde content te halen.

Naast het bouwen en ontwikkelen van een prospectieve MFA op Europees niveau (als de systeemgrens), wordt het MFA-model ook toegepast om potentiële verbeteringen in plasticrecyclage van twee belangrijke sectoren (als casestudies) in Europa te beoordelen, namelijk de elektronicasector en de verpakkingssector. In hoofdstuk 3 wordt het MFA-model toegepast om de hoeveelheid aan gerecycleerde plastics in België en Nederland te kwantificeren van drie geselecteerde elektrische en elektronische apparaten (EEE) binnen de groep kleine huishoudelijke apparaten (SHA), namelijk stofzuigers, koffiezetapparaten en elektrische scheerapparaten. De drie producten zijn geselecteerd vanwege hun hoge gehalte aan plastics in EEE. Voor de gekozen EEA wordt een multivariate input-outputanalyse (MIOA)-model ontwikkeld en toegepast om de hoeveelheid afgedankte elektrische en elektronische apparatuur (*waste electrical and electronic equipment*, WEEE) beter in te schatten, dat later gecombineerd wordt met MFA om de potentiële productie van gerecycleerde plastics uit WEEE in te schatten. Het MIOA-model is gekozen omdat het rekening houdt met de dynamische relatie tussen EEE die op de markt wordt gebracht (verkoop), voorraadstromen (d.w.z. de accumulatie van EEE op de markt) en de distributie over de levensduur van producten. Naast het combineren van MIOA en MFA wordt er ook een gevoeligheidsanalyse uitgevoerd om de belangrijkste knelpunten en mogelijke verbeteringen binnen de WEEE-recyclageketen te identificeren. Door middel van massabalansberekeningen en het analyseren van de samenstelling van WEEE werden de volgende feiten bestudeerd: i) de hoeveelheid gerecycleerde plastics die vrijkomt uit de geselecteerde WEEE-recyclage, ii) de beschikbaarheid van gerecycleerde plastics voor de geselecteerde (W)EEE, en iii) het inherente gehalte aan gevaarlijke oude chemische stoffen uit de geselecteerde (W)EEE. De modelresultaten geven aan dat het verbeteren van de selectieve inzameling van WEEE en de voorbewerkingsstappen om kunststof terug te winnen uit WEEE (ontmantelen, shredden en sorteren van materialen) essentieel zijn om het recyclagedoelstellingen voor EEE te halen. Bovendien voorspelt het model dat de historische gevaarlijke oude chemische stoffen uit WEEE in 2030 nog steeds kunnen worden aangetroffen (tot 5 gewichtsprocent), hoewel de beperking van gevaarlijke stoffen sinds 2006 van kracht is. Dit kan worden verklaard door te kijken naar de levensduurverdeling van de geselecteerde EEE-producten en de MIOA-resultaten, die suggereren dat een aanzienlijke hoeveelheid (W)EEE (tot 10% in gewicht) die tussen 1990 en 2006 is gekocht, in 2030 nog steeds kan worden weggegooid.

De volgende casestudies in dit onderzoek onderzoeken de recyclageprestaties (vanuit technisch en economisch perspectief) van de zogenaamde moeilijk te recycleren plastic verpakkingen: flexibel verpakkingsafval. Specifiek onderzoekt dit promotieonderzoek potentiële verbeteringen in het beheer van flexibele verpakkingsafval van huishoudens (bijv. individuele huishoudens) en niet-huishoudelijke sectoren (bijv. winkels, magazijnen, fabrieken, enz.). Hoofdstuk 4 van dit proefschrift onderzoekt de recyclageprestaties (vanuit technisch en economisch perspectief) van een verbeterd mechanisch recyclageproces voor flexibel verpakkingsafval van huishoudens. Hiervoor worden de recyclageprestaties onderzocht van een verbeterd mechanisch recyclageproces, *Quality Recyclage Process* (QRP) genaamd, ontwikkeld (en voorgesteld) door de industrie, en vergeleken met een conventioneel mechanisch recyclageproces voor flexibel verpakkingsafval in Europa. De ontwikkelde MFA vertrekt van twee gesorteerde balen (bv. balen rijk aan PE-folie of polyolefinen-folie) na het sorteren van huishoudelijk afval in een Duitse sorteerinstallatie (*material recovery facility,* MRF), die kunnen worden gerecycleerd via QRP of een conventioneel mechanisch recyclageproces. Het MFA-model wordt gebruikt om de materiaalstromen te traceren, zoals flexibel verpakkingsafval en niet-flexibel verpakkingsafval (bijv. harde verpakkingen of residu). De MFA-resultaten worden beoordeeld aan de hand van vier evaluatie-indicatoren met betrekking tot kwantiteit (procesrendement en netto terugwinningsindicatoren) en kwaliteit (polymeerkwaliteit en transparantieniveau). Vervolgens worden de economische prestaties van QRP en conventionele mechanische recyclage geschat door de MFA-resultaten (massabalansen) te combineren met economische modelparameters zoals het verbruik van nutsvoorzieningen (bijv. elektriciteit, brandstof, water, enz.), kapitaalinvesteringen en de verkoop van regranulaat. Bovendien worden gevoeligheidsanalyses uitgevoerd om mogelijke variaties van de geselecteerde modelparameters in de modelresultaten te beoordelen (d.w.z. evaluatie-indicatoren en economische balansen). Dit onderzoek geeft aan dat QRP voor flexibel verpakkingsafval uit huishoudens vergelijkbaar presteert met conventionele mechanische recyclage, maar dat QRP een hogere kwaliteit regranulaat oplevert. Vanuit een economisch perspectief verbetert QRP de economische balans van de recyclage van flexibel verpakkingsafval, en QRP kan belangrijk zijn bij het vergroten van de circulariteit voor flexibele verpakkingen.

Naast huishoudelijk afval wordt in hoofdstuk 5 de economische haalbaarheid onderzocht van de inzameling en mechanische recyclage van flexibel verpakkingsafval uit de niet-huishoudelijke sector. De economische haalbaarheid wordt beoordeeld door het bouwen van een model voor kosten-batenanalyse (*cost benefit analysis*, CBA), beginnend met een logistieke simulatie van selectieve inzameling, gevolgd door een MFA van mechanische recyclage, en de bijbehorende economische aspecten (d.w.z. kapitaalinvestering, jaarlijkse kosten en verkoop van reganulaat). In het bijzonder beschouwt dit onderzoek stedelijke gebieden van de stad Gent in België en haar twaalf naburige gemeenten als een casestudy (systeemgrens). De CBA van flexibele verpakkingen uit niet-huishoudelijke sectoren begint met het karakteriseren van de samenstelling en het schatten van de hoeveelheid afval die wordt geproduceerd per bedrijfsactiviteit (in Europa ook NACE-sector genoemd) in de stedelijke gebieden die in deze studie worden beschouwd. De afvalhoeveelheid wordt geschat door reële steekproeven in 2018 in de stad Gent, gecombineerd met extrapolatie met Orbis-databases die gegevens verschaffen over het aantal bedrijven per regio. De afvalsamenstellingen worden gekarakteriseerd tijdens analysecampagnes tussen december 2021 en februari 2022. De campagnes bestrijken vijf belangrijke niet-huishoudelijke sectoren, zoals Groothandel (bv. NACE G.46), Detailhandel (bv. NACE G.47), Bouwnijverheid (bv. NACE F.41), Logistiek (bv. NACE H.49), en 'andere' sectoren (bv. NACE C. 10, NACE C.18, enz.). Zodra de afvalsamenstelling en -hoeveelheid zijn geschat, worden de jaarlijkse kosten van selectieve inzameling in drie verschillende inzamelfrequenties (wekelijks, tweewekelijks en maandelijks) geschat met behulp van de OptiFlow©-software. Verder worden de materiaalstromen en de economische balans van mechanische recyclage van niet-huishoudelijk flexibel verpakkingsafval onderzocht, inclusief de geschatte inkomsten uit de verkoop van regranulaat. Tot slot wordt de koolstofvoetafdruk van de inzameling en mechanische recyclage van flexibel verpakkingsafval van niet-huishoudens geschat en vergeleken met het basisscenario (d.w.z. productie van nieuw PE-granulaat met verbranding als EoL-behandeling). Dit onderzoek suggereert dat de strategie voor het inzamelen en mechanisch recycleren van flexibel niet-huishoudelijk verpakkingsafval afhankelijk is van de frequentie waarmee het afval wordt ingezameld, het bereiken van een minimale hoeveelheid afval en het handhaven van de kwaliteit van de grondstof (de kwaliteit van het afval dat mechanisch wordt gerecycled).

In hoofdstuk 6 van dit doctoraatsonderzoek wordt een voorlopige beoordeling uitgevoerd van de beschikbaarheid van gerecycleerde content voor flexibele verpakkingen (huishoudelijk en niet-huishoudelijk) in Europa, uitgaande van 100% gesloten kringlooprecyclage van flexibel verpakkingsafval terug naar nieuwe flexibele verpakkingen. De nieuwe voorgestelde regelgeving voor verpakking en verpakkingsafval (*packaging and packaging waste regulations,* PPWR) omvat verplichte minimumdoelen voor gerecycleerde content voor flexibele verpakkingen, d.w.z. een ambitie van 35% gerecycleerde content voor niet-contactgevoelige en 10% gerecycleerde content voor contactgevoelige flexibele verpakkingen in 2030. Om te beoordelen of het haalbaar is om de doelstelling voor gerecycleerde content te halen, is een MFA-model ontwikkeld om het lot van flexibel verpakkingsafval tijdens de verwerking aan het einde van de levensduur te traceren en de productie in 2030 in Europa te regranuleren. Het MFA-model is opgebouwd in de veronderstelling dat het ontwerp van flexibele verpakkingen in 2030 verbeterd zal zijn (bijv. van multi- naar mono-materiaal flexibele verpakkingen), dat flexibele verpakkingen selectiever ingezameld zullen worden (bijv. meer P+MD-inzameling in Europa), dat sorteertechnieken beter zullen presteren (bijv. 'slimme' sorteertechnieken met behulp van digitale watermerken of kunstmatige intelligentie) en dat mechanische recyclage en pyrolyse een beter rendement zullen opleveren (bijv. geavanceerde mechanische recyclage zoals QRP of katalytische pyrolyse). Voor het modelleren van de toekomstige verwerking aan het einde van de levensduur worden vijf scenario's ontwikkeld en onderzocht, die alleen bestaan uit mechanische recyclage voor flexibele verpakkingen en mechanische recyclage en pyrolyse als aanvullende technieken om de doelstelling voor gerecycleerde content te halen die in Europa is vastgesteld. Verder wordt er een schatting gemaakt van de kapitaalinvestering die gepaard gaan met het behalen van de doelstellingen voor gerecycleerde content, die is ontwikkeld op basis van de resultaten van het MFA-model in combinatie met economische factoren (in €/ton) uit de literatuur. De resultaten van het MFA-model suggereren dat de streefcijfers voor gerecycleerde content kunnen worden gehaald door mechanische recyclage en pyrolyse als complementaire technieken te gebruiken om flexibel verpakkingsafval te verwerken. In de meest positieve scenario's zou er €7,7 – 8,8 miljard aan kapitaalinvesteringen nodig zijn om infrastructuur voor mechanische recyclage en pyrolyse te bouwen, inclusief voorbehandeling en hydrotreatment voor pyrolyse. De resultaten van de MFA wijzen ook op een wisselwerking tussen het bereiken van regranulaat met een hogere kwaliteit om de doelstelling van 10% gerecycleerde content voor flexibele verpakkingen te halen (ervan uitgaande dat pyrolyse een dominantere techniek wordt om de doelstelling te halen), en de jaarlijkse productie van regranulaat (d.w.z. de hoeveelheid secundaire materialen). Als gevolg hiervan zou het totale recyclagepercentage aan het einde van de levensduur voor flexibele verpakkingen kunnen dalen wanneer de streefcijfers voor contactgevoelige gerecycleerde content worden gehaald, ervan uitgaande dat pyrolyse een dominantere techniek zou worden om regranulaat van hogere kwaliteit te produceren dat geschikt is voor contactgevoelige toepassingen in Europa, zoals bijvoorbeeld voor voedselverpakkingen.

In het laatste hoofdstuk 7 wordt de algemene conclusie van dit doctoraatsonderzoek gepresenteerd, inclusief het toekomstperspectief op het verbeteren van het ontwikkelde MFA-model en plastic circulariteit. Het belang van het uitbreiden van de systeemgrenzen naar andere regio's (bijv. andere landen binnen of buiten Europa), productcategorieën (bijv. andere EEA) en sectoren (bijv. automobiel sector) wordt benadrukt. Het toegepaste MFA-model kan worden verbeterd door meer scenario's te ontwikkelen (bijv. het implementeren van nieuwe ontwerpen of bedrijfsmodellen zoals reparatie of hergebruik) en een meer gedetailleerde karakterisering van de materiaalsamenstelling. Daarnaast kunnen de kwantiteitsgebaseerde MFA-modellen die in dit doctoraatsonderzoek worden gepresenteerd, worden verbeterd door meer kwaliteitsaspecten van het regranulaat te koppelen. Tot slot worden de resterende uitdagingen en mogelijkheden voor verder onderzoek naar de duurzaamheidsaspecten van plastics besproken.

In het algemeen toont dit promotieonderzoek aan dat de circulariteit van plastics in Europa moet verbeteren. MFA kan dienen als hulpmiddel om de status quo te beoordelen en potentiële verbeteringen binnen de Europese plasticrecyclageketen te meten door potentiële circulariteitsscenario's te ontwikkelen. MFA kan inzicht geven in hoe significant de verbeteringen zouden kunnen zijn (vergeleken met het basisscenario) en potentiële extra kapitaalinvesteringen en kosten. Tot slot kan de MFA die in dit doctoraatsonderzoek wordt gepresenteerd, worden gebruikt om plasticrecyclagestrategieën te formuleren (bijv. doelstellingen voor minimale gerecycleerde content) op basis van wetenschappelijk onderbouwde prognoses en om het behalen van de plasticrecyclagedoelstellingen in Europa te monitoren.

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CHAPTER 1: GENERAL INTRODUCTION AND RESEARCH OBJECTIVES

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Chapter 1

Introduction and research objectives

1.1 PLASTIC IN THE ECONOMY AND OUR DAILY LIVES

Plastic is a term derived from the Latin "*plasticus*" or Greek "*plastikos*", which originally means 'to form' or 'to grow', for describing materials that can be formed and molded under heat and pressure. Plastic is used to describe a large family of different materials with different characteristics, properties, and uses. Polymers are the chemical compound that make up all modern plastics, which are large molecules consisting of a chain of repeating smaller molecules (monomers) combined through a polymerization process (Plastics Europe, 2018; OECD, 2022). The two categories of plastic are thermoplastic and thermoset plastic. The thermoplastic group is plastic that can be (re-)melted, heated, and shaped repeatedly. Examples of plastic within the thermoplastic family are High-Density Polyethylene (HDPE), Low-Density Polyethylene (LDPE) Polypropylene (PP), Polyethylene Terephthalate (PET), Polystyrene (PS), Expanded Polystyrene (EPS), Acrylonitrile Butadiene Styrene (ABS), Polyvinyl Chloride (PVC), Polyamides (PA), Polycarbonate, (PC), and Polymethyl Methacrylate (PMMA). The thermoset plastic group is plastic that cannot be (re-)melted, heated, and shaped because the polymers are cross-linked during the curing process to form irreversible chemical bonds. Examples of thermoset plastic are Polyurethane (PUR), polyester, vinyl ester, and phenolic resins (Plastics Europe, 2018).

The use of plastic in the modern economy is growing rapidly due to its unique properties of plastic such as high strength-to-weight ratio, high moldability, resistance to physical and chemical degradation, low production cost, and broad application range (Hsu et al., 2021; Lebreton and Andrady, 2019). Since mid-1950 the global plastic production began its unprecedented growth, which is expanding up to 230-fold compared to the present day. In 2019, global plastic use amounted to 460 million tonnes (Mt), which is estimated that 51% is produced in China, 19% is produced in North America (Mexico, Canada, and the United States), and 16% is produced in Europe. In the same year, it is estimated that 353 Mt of plastic waste was generated globally, of which 6% was effectively recycled while the remaining mass was mainly incinerated or landfilled (Plastics Europe, 2020; OECD, 2022).

The demand for plastic in Europe – European Union (EU) 27+Norway, Switzerland, and the United Kingdom–is estimated to be 50.7 Mt in 2019 (Figure 1.1). The packaging sector accounted for 40% of the total plastic demand in Europe, followed by the building and construction sector (20%), automotive sector (10%), electrical & electronic sector (6%), and agriculture sector (3%). The remaining 21% of plastic demand in 2019 was used in the textile, medical, and household sectors (incl. furniture and toys) (Plastics Europe, 2020). In Europe, the use of thermoplastic materials accounted for around 80% of the total plastic demand (equals 41 Mt), in which polyolefin (PO) materials (PE and PP) are the biggest fraction followed by PVC, PET, (E)PS, ABS, PA, and PC (Plastics Europe 2020; 2022) (Figure 1.1).

The European plastic industry strives to achieve a circular and climate-neutral plastic economy, in which a system is designed so that plastic is produced, converted, consumed, and managed in a sustainable way. This ambition also means that establishing sustainable end-of-life (EoL) management systems and fostering the use of recycled plastic are imperative (Plastics Europe, 2022; Maury et al., 2022; Feber et al., 2020; OECD, 2022). Moreover, a few initiatives have been initiated to increase the plastic recycling rate, such as the Circular Plastic Alliance (CPA), which aims to boost recycled plastic production to 10 Mt by 2025 (European Commission, 2022b). A few regulations have also been put into force to enable sustainable plastic waste management, such as recycling rate targets, recycled content targets, and limiting plastic waste to be landfilled in Europe (European Commission 2022a; 2018a; 2018b). More information about the EoL management systems and regulatory framework for managing plastic waste is further elaborated in the following sections.

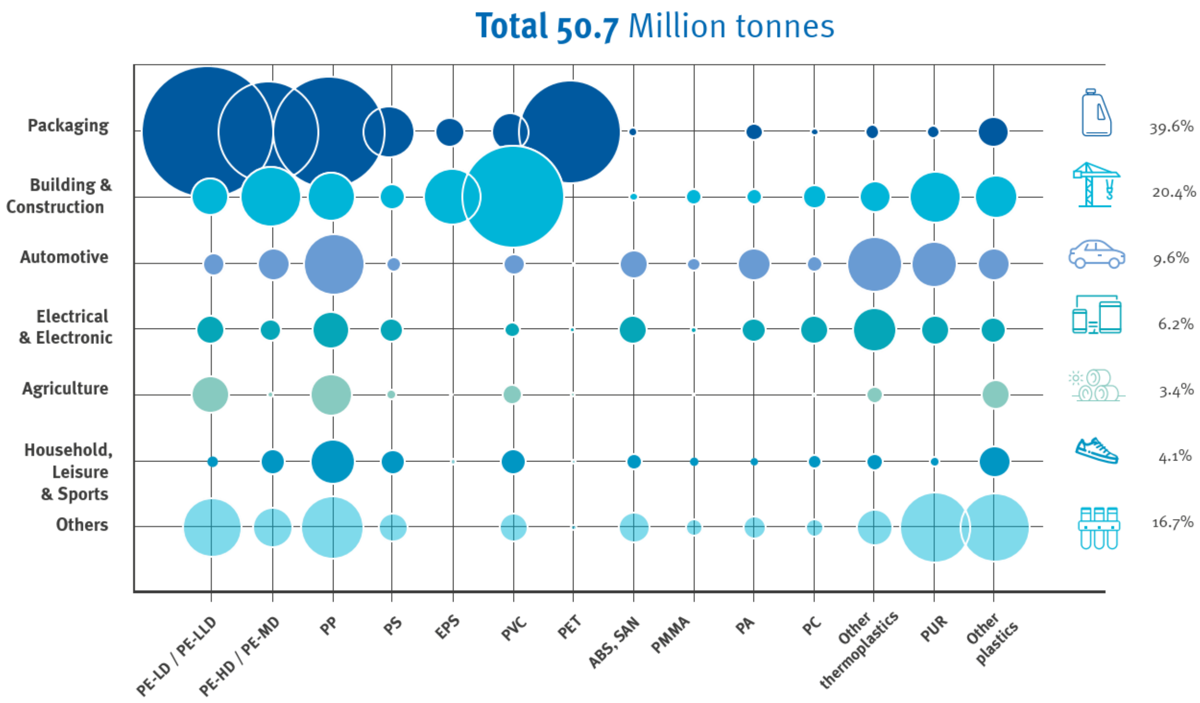

Figure 1.1 Total plastic demand by sector and polymer type in Europe in 2019. *Source*: Plastics Europe (2020).

1.2 END OF LIFE MANAGEMENT OF PLASTIC WASTE IN EUROPE

According to Plastics Europe (2020), Europe generated 29.1 Mt of plastic waste in 2019. Around 19.6 Mt (67%) of the generated waste was estimated to be landfilled or incinerated, and only 9.4 Mt (33%) was sent to recycling facilities in Europe in the same year. Ultimately, the recycled plastic production in Europe is estimated to be around 4 Mt, mainly produced via mechanical recycling (MR). Of the 29.1 Mt plastic waste generated, 17.4 Mt comes from the packaging sector, followed by electrical and electronics sector (1.8 Mt), building and construction sector (1.8 Mt), agriculture sector (1.5 Mt), automotive sector (1.5 Mt). The other sectors, such as textile, household, and medical generated 5.1 Mt plastic waste in 2019 (Plastics Europe, 2020; 2019a). Moreover, some studies also indicate a considerable amount of plastic waste leaks into the environment as macro- and micro-plastics (Ryberg et al., 2019; Peano et al., 2020; Boucher et al., 2020).

Optimizing end-of-life (EoL) waste management systems is essential to realize a circular economy for plastic. A typical plastic waste treatment in Europe starts with a selective collection of waste at source (also known as *source separation*) followed by material recovery or *sorting* (of the source separated waste) and *recycling* (of the correctly sorted waste) into new secondary materials to be used again in the economy (Plastics Europe, 2019a; 2020; 2021; 2022) (Figure 1.2). The source separation of plastic waste relies on adequate waste sorting by individuals (e.g., at school, offices, houses, etc.), which can be facilitated by well-established waste management infrastructures and providing relevant disposal instructions for each product (e.g., on-pack labeling) (Albizzati et al., 2023). Plastic waste fractions that are not source separated are found in the mixed residual waste stream, which is typically landfilled or incinerated (Figure 1.2). Extra sorting and cleaning steps can be done to (post-)sorting plastic from mixed residual streams. Nonetheless, these cannot always be performed in the most efficient way, which means that currently not all plastic waste is post-sorted from mixed residual streams for recycling (Plastics Europe, 2022; Brouwer et al., 2018; Picuno et al., 2021).

After collection, the source separated waste will be sent to material recovery facilities for sorting (Figure 1.2). A series of mechanical sorting units separate plastic from non-plastic materials. A few studies have indicated the importance of robust plastic sorting to improve feedstock quality for recycling and to produce high-quality secondary materials (Van Eygen et al., 2016; Parajuly et al., 2016; Roosen et al., 2022; Kleinhans et al., 2021a). Thereafter, the correctly sorted waste will be sent for recycling, in which a few recycling options can be selected (Figure 1.2). To date, mechanical recycling (MR) of plastic is the most ubiquitous option to convert plastic waste into secondary materials (Ragert et al., 2017; Plastics Europe, 2019a). However, in the near future, emerging recycling technologies such as chemical recycling (CR) of pyrolysis, gasification, chemical depolymerization (e.g., glycolysis, methanolysis, etc.), and solvent-based recycling (SBR) technologies (e.g., dissolution-precipitation) are expected to play a crucial role in achieving a circular economy for plastic. The outputs of the abovementioned recycling options are recycled plastic (e.g., from MR and SBR) as well as valuable base chemicals and fuels for secondary resources in the petrochemical industry (e.g., from CR) (Simon and Martin, 2019; Hann and Connock, 2020; Crippa et al., 2019) (Figure 1.2).

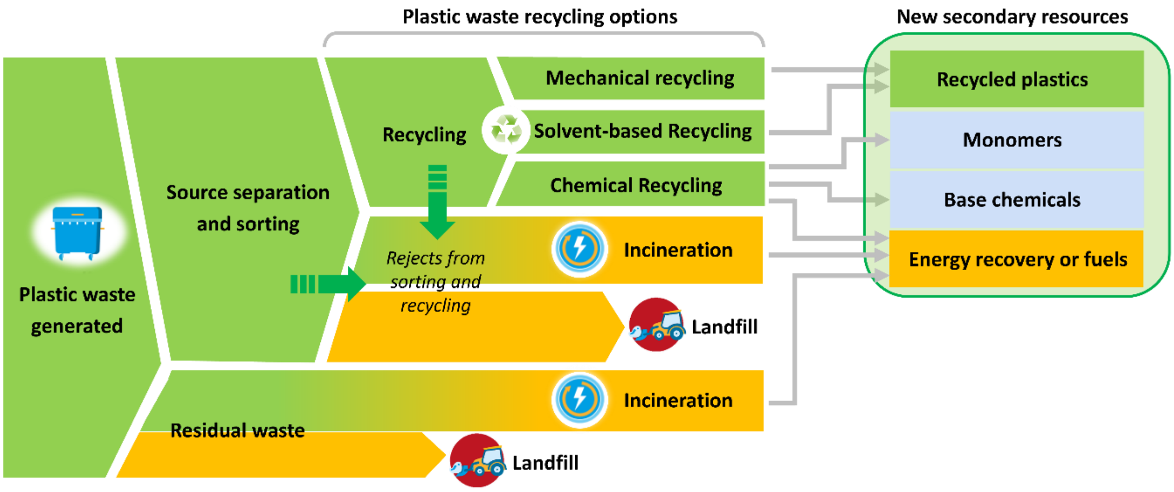

Figure 1.2 An illustrative end-of-life management of plastic waste in Europe. The thickness does not represent the mass. *Adapted from*: Plastics Europe (2019a).

1.2.1 Plastic waste collection

A separate waste collection at source (or *source separation*) typically increases plastic recycling rates. According to Plastics Europe (2022), plastic recycling rates are 13 times higher when the waste is sourced separated. A study by Cimpan et al. (2015) suggests that an established separate waste collection system diverts 56% of waste from the mixed residual stream toward material recovery and recycling. A study by WRAP (2009a) suggests that source separation of plastic waste increases the feedstock quality to be fed into the recycling facilities and, subsequently, the recycled plastic to meet end-market requirements. Albizzati et al. (2023) highlight the importance of source separation of plastic waste as an enabler for the high recovery of plastic waste as feedstock for further recycling chains.

Various source separation schemes for plastic waste can be found in Europe. For the waste electronic and electrical equipment (WEEE), authorized waste collectors use different channels to collect WEEE such as i) container parks, ii) reuse and repair centers, or iii) electronic shops through a take-back scheme (Recupel, 2018; 2013; Huisman et al., 2012). In the automotive sector, end-of-life vehicles (ELV) can be brought to an authorized dismantler to be (formally) deregistered and sent for recycling (Maury et al., 2022; ADEME, 2019; Baldassarre et al., 2022). Through these formal collection channels, the WEEE and ELV are treated according to specific guidelines to dismantle and de-pollute the waste before being sent for further sorting and recycling processes (Van Eygen et al., 2016; De Meester et al., 2019; Recupel, 2018; 2013; Huisman et al., 2012; Maury et al., 2022; Baldassarre et al., 2022;

ADEME, 2019; Aigner, 2020). Source separation of agriculture plastic waste (APW) and plastic in construction and demolition waste (CDW) is typically done under a specific agreement (take-back schemes) between the waste producers (e.g., farmers) and waste management companies. For example, waste management companies and farmers can come to an agreement to collect the APW (e.g., agriculture films) through a 'pick up' or 'bring' system (Agriculture Plastic Environment, 2021; Bauer, 2019). In the construction sector, the waste operators typically collect easy-to-identify objects such as window or door profiles, pipes, cables, and insulation from mixed CDW. The plastic-based items in mixed CDW are typically source separated at the collection sites in separate containers or bins for further recycling process (Bendix et al., 2021; Gardner, 2020).

In the packaging sector, plastic packaging waste can be collected through a deposit refund scheme (DRS), as demonstrated in PET bottle collection in Belgium Norway, Germany, Finland, Sweden (SYSTEMIQ, 2023; 2022; Eunomia, 2022; Ghosh et al., 2023). Another option for source separation of plastic packaging waste is a comingled collection scheme, whereby recyclable materials such as plastic packaging, metal cans, and drinking cartons are collected together. Examples of such comingled collection schemes are the P+MD system in Belgium and The Netherlands and lightweight packaging collection (yellow bags and bins) in Germany (Cimpan et al., 2016; 2015; Roosen et al., 2022; Kleinhans et al., 2021a; Picuno et al., 2021; KIDV, 2023). After collection of plastic waste, the separately collected plastic packaging at the source will be transferred to material recovery facilities (MRFs) for sorting and later on recycling (Kleinhans et al., 2021a; Cimpan et al., 2016).

1.2.2 Plastic waste sorting

The separately collected (source separation) waste is further sorted in dedicated sorting facilities, also known as material recovery facilities (MRFs). At the sorting stage, the mixed comingled waste fractions are sorted into different outputs based on the material characteristics. In order to make an efficient sorting of waste, manual sorting is often combined with state-of-the-art automated sorting units (Gundupalli et al., 2017; Kleinhans et al., 2021a; Cimpan et al., 2015). The automated sorting units are used to directly sort plastic from mixed waste streams based on different material properties. Examples of sorting equipment are overbelt magnet and eddy current separation that sort material from the mixed waste streams based on their magnetic susceptibility and electrical conductivity, respectively. The density-based and shape/size-based sorting is performed using windshifters, drum screens, and ballistic separators. Moreover, sensor-based sorting can also be used to sort plastic from mixed waste streams using a near-infrared technology (NIR). The NIR sorting is able to detect plastic from non-plastic materials as well as different plastic types (e.g., PET, PP, PE, PS, etc.) based on the unique reflected wavelength of specific plastic resins (Gundupalli et al., 2017; Cimpan et al., 2016; Kleinhans et al., 2021a). Several improvements using sensor-based technologies are also being developed such as to sort plastic packaging waste based on its original purposes (e.g., sorting food- vs. non-food packaging) using digital watermarks, chemical tracers, artificial intelligence, or robotic technologies (Soares et al., 2022; De Tandt et al., 2021; NTCP, 2023). In practice, all the abovementioned sorting technologies are used to recover plastic from different waste streams such as plastic packaging, plastic in WEEE, and plastic in ELVs. The sorting units are placed in specific orders and designs (i.e., sorting plant layouts) to recover the targeted plastic material effectively. Previous studies have shown potential sorting plant layouts in different waste streams such as Kleinhans et al. (2021a) for plastic packaging sorting, De Meester et al. (2019) in recovering plastic from WEEE, and Maury et al. (2022) in plastic recovery from ELVs.

The separately collected plastic packaging waste (in comingled stream with other recyclable materials) is typically sorted at MRFs to be separated based on the polymer types for recycling. In particular, sorting plants for packaging waste use combinations of the following sorting units: drum screens, windshifters, ballistic separation, magnetic separation, eddy current separation, and NIR machines. Moreover, manual sorting can be performed at the end of the plastic packaging sorting lines for quality control of the sorted waste streams (Kleinhans et al., 2021a; Cimpan et al., 2015; Picuno et al., 2021). Typical outputs of plastic packaging waste sorting are different bales for recycling such as PE Rigid bales (i.e., bale rich in HDPE bottles), PET bottle bales (i.e., bale rich in mixed PET bottles), PE Film bales (i.e., bale rich in PE films), MPO bales (i.e., bale rich in mixed rigid polyolefin items), etc. (Kleinhans et al., 2021a; Bashirgonbadi et al., 2022; Brouwer et al., 2018; Roosen et al., 2022).

In the electronic sector, sorting start with manually dismantling batteries, copper wires, glass, wood components, and components with high (precious) metal content from the WEEE by trained labors. The remaining (non-dismantled) WEEE parts are sent for further automated sorting processes using NIR, magnetic separation, eddy current, etc. to recover plastic from the mixed WEEE streams (Van Eygen et al., 2016; Menad, 2016). Similar to WEEE, the sorting of ELV starts with manual dismantling of ELV parts rich in plastic content such as the bumpers, dashboards, cushions, and 'under-the-hood' components. Later the manually dismantled part can be sent directly to recycling facilities, while the remaining ELVs parts are furthered process using automated sorting technologies (e.g., NIR) to separate and recover plastic for recycling (Maury et al., 2022; Aigner, 2020; Baldassarre et al., 2022). In agriculture and construction sectors, the farmers or construction workers typically do manual sorting of APW and plastic in CDW on the collection sites (in the agriculture fields or construction sites). After that the sorted plastic waste fractions are collected by waste management companies for recycling (Bauer, 2019; Scarascia-Mugnozza et al., 2012; Bendix et al., 2021; Gardner, 2020).

1.2.3 Plastic waste recycling

The separately collected and sorted plastic waste can be recycled using different recycling options such as MR, CR, or SBR (also called solvent-based purification), as shown in Figure 1.2 and Figure 1.3.

MR refers to mechanical reprocessing of plastic waste by means of shredding, washing, drying, and extrusion without breaking down the polymer chains (Ragaert et al., 2017; Crippa et al., 2019). Typical recycling equipment used in MR of plastic is cold washing, friction washer, hot washing, sink-float separation (density-based separation), dryers (mechanical or thermal), and plastic extrusion (Larrain et al., 2021; Faraca & Astrup et al., 2019; Horodytska et al., 2018; Brouwer et al., 2018; Ragaert et al., 2017). The output of MR is recycled plastic (as regranulate) that can be used directly in different applications such as film blowing or injection molding, typically with extra measures like blending with virgin material (Bashirgonbadi et al., 2022) (Figure 1.3).

Solvent-based recycling (SBR) is a process in which the polymer is dissolved using specific solvent (chemical agents), followed by removing additives through for example filtration or phase extraction. Later, the dissolved polymer is recovered by introducing an anti-solvent, in which the polymer will be precipitated and recovered (Crippa et al., 2019; Kol et al., 2021). The final product of SBR is a 'near-virgin' polymer that can be reformulated into different applications (Crippa et al., 2019), as shown in Figure 1.3.

Finally, CR refers to plastic recycling technologies that break down the polymer chains and convert them into oligomers/monomers, base chemicals, and hydrocarbons (Hann and

Connock, 2020; Arena and Ardolino, 2022; Manžuch et al. 2021) (Figure 1.3). CR is often used as an umbrella term that cover a broader set of technologies such as chemical depolymerization (also known as solvolysis such as glycolysis, methanolysis, etc.), pyrolysis, and gasification. *Chemical depolymerization*, mostly suitable for condensation polymers (e.g., polyester), is a process by which the polymer chain is broken down using chemical agents to produce shorter polymer chains such as oligomers or monomers. Next, the oligomers or monomers are recovered using distillation, precipitation, and/or crystallization techniques (Hann and Connock, 2020; Crippa et al., 2019; Manžuch et al., 2021; Vollmer et al., 2020). *Pyrolysis*, mainly suitable for addition polymers (e.g., polyolefin), is a process applied to plastic waste to thermally break polymer chains in an oxygen-free environment and elevated temperatures of 350–500°. The main product of pyrolysis of plastic is pyrolysis oil, which is further distilled into naphtha. The naphtha is further processed at steam crackers to produce monomers, base chemicals, and hydrocarbons (as fuel) (Kusenberg et al., 2022a; 2022b; 2022c; 2022d; 2022e; Ragaert et al., 2017; Vollmer et al., 2020). *Gasification* is a plastic recycling option involving partially oxidizing plastic waste at elevated temperatures of 700–1,500° to produce syngas (a mix of hydrogen and carbon monoxide). The syngas can be processed via Fischer Tropsch Synthesis (FTS) or Methanol-to-olefin (MTO) process to produce monomers and base chemicals (Hann and Connock, 2020; Crippa et al., 2019; SYSTEMIQ, 2022; Mastellone, 2019; Gholami et al., 2021). Thus, the output of CR plastic waste can be used as feedstock for polymerization (monomers or oligomers) or the petrochemical industry (base chemicals as secondary materials in a petrochemical refinery), as shown in Figure 1.3.

The abovementioned recycling options can be used depending on the quality of the sorted plastic streams (e.g., contamination level) or the targeted quality of the recycled plastic (Arena and Ardolino, 2022). Several studies have shown that MR faces several challenges in treating plastic waste (e.g., thermal-mechanical degradation). This results in the inadequate technical properties of final recycled plastic to meet the market demands (Ragaert et al., 2017; Demets et al., 2020; Huysveld et al., 2022; Bashirgonbadi et al., 2022; De Tandt et al., 2021). Several improvements can be introduced to tackle the challenges of MR plastic such as extensive (pre-)treatment processes (e.g., deinking and deodorization) and improved extrusion (e.g., double melt filtration with degassing), as shown by Roosen et al. (2021), Bashirgonbadi et al. (2022), and Kol et al. (2021). Furthermore, CR and SBR are expected to play a crucial role in European plastic recycling systems. These technologies are claimed to have a higher tolerance towards contaminations and can process complex waste streams (e.g., multi-material/multilayer packaging or mixed residual waste). Several studies also show that the final product of CR and SBR is of higher quality than recycled plastic from MR (Kusenberg et al., 2022b; Huysveld et al., 2022; De Tandt et al., 2021).

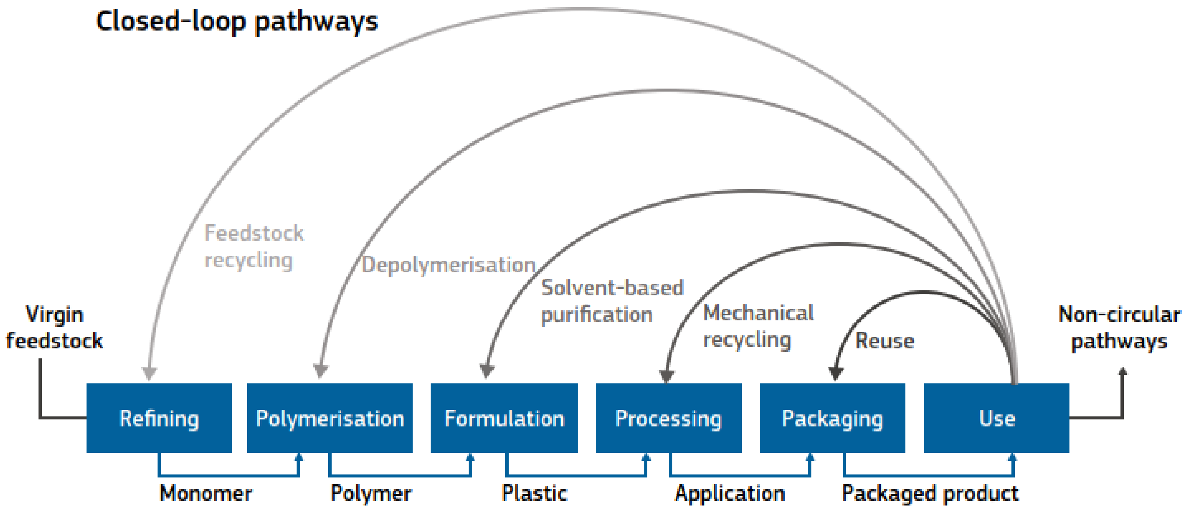

Figure 1.3 Overview of different recycling options for plastic in a circular economy. *Source*: Crippa et al. (2019).

1.3 EUROPEAN LEGISLATIONS AND PLEDGES TO INCREASE PLASTIC CIRCULARITY

To realize a circular economy for plastic, the European Commission (EC) has enacted several regulations, along with pledges made by the European plastic industry such as brand owners, manufacturers, and recyclers. The summary of the pledges and regulations related to plastic waste can be found in Table 1.1. The Waste Framework Directive (WFD) lays out the definition and hierarchy of waste management, such as 'waste', 'recycling', and 'recovery'. Article 3 of WFD implies that 'recycling' is *waste materials that are reprocessed into products, materials or substances whether for the original or other purposes*. However, reprocessing materials into fuels or as backfilling operations is not considered a 'recycling' under WFD (European Commission, 2018a; 2008). Furthermore, the amendments of *Council Directive 1999/31/EC on landfill of waste* and *Directive 2008/98/EC on waste* set out new targets in which Member States shall ensure all waste is suitable for re-use and recycling or other recovery processes (e.g., incineration with energy recovery) by 2030. Particularly, the amount of mixed (municipal) residual waste stream landfilled (by weight) is limited (or should be reduced) to 10% by 2035 (European Commission, 2018a; 2018b). The EC has also recently passed a tax for non-recycled plastic packaging waste of €800 per tonne starting from 2021 (European Commission, 2020a).

Specific to the packaging sector, the Single Use Plastic Directive (SUPD) and Packaging and Packaging Waste Directive (PPWD) have been introduced to enable a circular economy for plastic packaging (Table 1.1). The SUPD focuses on reducing the amount of single-use waste and establishing extended producer responsibility (EPR) schemes, in which the producers are obliged to properly manage the generated plastic packaging waste. Specifically, the SUPD bans certain single-use plastic items (e.g., plastic bags, plastic containers for food, etc.) from being placed on the European market. The SUPD also mentions specific targets to increase the collection of plastic bottles up to 90% by 2029, incorporating 25% recycled plastic in PET beverage bottles from 2025, and mandates 30% recycled content in all plastic beverage bottles by 2030 (European Commission, 2019a). Under PPWD, specific recycling targets are introduced, such as a 55% recycling target for plastic packaging by 2030 and a 65% recycling target for mixed (municipal) waste by 2035 (European Commission, 2018c). The new Packaging and Packaging Waste Regulations (PPWR) proposal to amend the current PPWD embeds new targets for plastic packaging around waste prevention, re-use and refill, and minimum recycled content targets (European Commission, 2022a). The PPWR mandates a packaging waste reduction of 5% by 2030 (e.g., through maximum allowable empty or unnecessary space in packaging), harmonized labeling scheme (incl. recycled content and disposal instructions), and a proportion of packaging to be reusable and/or refillable by 2030. Article 7 of the new PPWR proposal also sets out minimum recycled content targets per packaging item such as 10% for contact sensitive packaging (other than PET) and 35% for non-contact sensitive packaging (other than PET) by 2030. All packaging formats placed on the European market should be recyclable by 2030. This means that the 'Design-for-Recycling' (DfR) principles should be adopted for all packaging items entering the European market by 2030 (European Commission, 2022a). However, to attain recycled content targets, harmonized rules to calculate recycled content still needs to be developed. Such rules would allow a consistent monitoring and measuring system in Europe.

In the electronic sector, the WEEE Directive in 2012 mandates the integration of WEEE separate collection, sorting, and recycling systems. The WEEE Directive also calls for harmonization on WEEE categories and targets 85% of annual WEEE to be collected from 2019 onwards. Specific electronic components containing hazardous substances, such as plastic-containing brominated flame retardants (BFR) or mercury-containing components, should be removed and treated separately through authorized recycling facilities. Certain components containing high-value metals such as printed circuit boards, batteries, and electric cables must be selectively treated for material recovery and recycling (European Commission, 2012). Moreover, hazardous substances (e.g., BFR, mercury, etc.) are also restricted in new electronic products placed on the European market as stated in the Restriction of Hazardous Substances (RoHS) Directive. Safer substances must be used to substitute the restricted substances listed in Annex II of the RoHS Directive after the regulation was effectively put into force in 2006 (European Commission, 2011). With the legislation pushing towards sustainable use of plastic in new products placed on the market, the European electronic industry responded by pledging to use at least 25 – 50% of recycled plastic (by weight) in new electronic products by 2025 and 2030 (Sandoval, 2018; Philips, 2020; Whirlpool Corporation, 2018; Electrolux Group, 2020) (Table 1.1).

In the automotive sector, the ELV Directive was introduced in 2000 and a preliminary study to amend the directive was completed in 2020 (European Commission, 2000; Maury et al., 2022; Williams et al., 2020). The ELV Directive establishes a minimum requirement for reusing, recycling, and recovering materials from ELV. Article 7 or ELV Directive targets 85% of ELV to be 'reuse and recycling' and 95% of ELV to be 'reuse and recovery' starting from 2015. The Member States also must ensure new vehicles put on the European market do not use hazardous substances such as lead, mercury, cadmium, etc., starting from 2003 (European Commission, 2000). Moreover, several automotive producers also pledged to use recycled plastic in their new passenger cars. The voluntary pledges call for 20 – 25% of recycled plastic (by weight) in new cars by 2030 (Maury et al., 2022; Volvo, 2018; Carroll, 2021) (Table 1.1).

On the contrary, no European legislations directly address plastic waste recovery and recycling in APW and CDW yet. Plastic waste is typically found as a smaller fraction in CDW compared to other higher-density materials such as wood, glass, bricks, and concrete, i.e., estimated to be around 0.3 – 0.5% by weight in a mixed CDW. However, recycling plastic in CDW has high potential because of a relatively homogenous plastic composition found in several components such as window or door profiles (PVC), plumbing or piping system (HDPE), and insulation materials (EPS) (Hyvärinen et al., 2019; Bendix et al., 2021; Gardner, 2020).

Similarly, no European legislations directly target the recovery and recycling of APW. The waste is mainly managed voluntarily between the farmers or growers and waste management companies. In the agriculture sector, the APW can be found mainly as agriculture films (LDPE), bale wraps (HDPE), and horticulture products (PP pots or buckets). Also, APW is typically contaminated by organic waste and soil residues (Bauer, 2019; Agriculture Plastic Environment, 2021; Jansen et al., 2019; Vox et al., 2016).

Table 1.1 Summary of current and provision regulations or (voluntarily) pledges related to plastic waste treatment made by the European Commission and stakeholders within plastic value chain (e.g., brand owners, manufacturers, recyclers, etc.).

Current and provision regulation and pledge(s)	**Key objective(s) or target(s)**	**Target year**	**Sources**
Waste framework directive (WFD)	Definition of 'waste', 'recycling', and 'recovery'	-	European Commission, 2008; 2018a
	Hierarchy on (plastic) waste management: *prevention, preparing for reuse, recycling, recovery, disposal*	-	
	Preparing for re-use and recycling (60% by weight) of municipal solid waste	By 2030	
Landfill Directive	Restriction of landfilling for waste that is suitable for recycling or recovery by 10%	By 2035	European Commission, 2018b
Tax on plastic packaging waste	€800 per tonne of non-recycled plastic packaging waste	From 2021	European Commission, 2020a
Single Use Plastic Directive (SUPD)	Ban certain single-use plastic items	From 2019	European Commission, 2019a
	Introduction of extended producer responsibility (EPR) scheme		
	Collection target (90% by weight) for plastic bottle	By 2029	
	Minimum recycled content (25% by weight) in PET beverage bottles	By 2025	
	Minimum recycled content (30% by weight) in all plastic beverage bottles	By 2030	
Packaging and Packaging Waste Directive (PPWD)	Recycling target (55% by weight) for plastic packaging waste	By 2030	European Commission, 2018c
	Recycling target (70% by weight) for all packaging waste format	By 2030	
Packaging and Packaging Waste Regulation (PPWR) proposal	Design for Recycling (DfR) for all packaging formats placed on the European market	By 2030	European Commission, 2022a
	Reduction of packaging waste generated (5% by weight) compared to 2018 baseline	By 2030	

	Re-use and refill targets (e.g., 10% for alcoholic beverages)	By 2030	
	A minimum recycled content target for contact sensitive (10% by weight) and non-contact sensitive (35% by weight) packaging items (except for PET packaging format)	By 2030	
Waste Electrical and Electronic Equipment Directive (WEEED)	Formal and integrated separate WEEE collection point (channels) and recycling system	-	European Commission, 2018d
	Harmonized WEEE categories and separate collection target (85% by weight)	From 2019	
	Proper treatment of WEEE such as removal of all hazardous substances (plastic containing flame retardants) or a selective treatment for certain components (e.g., batteries, printed circuit boards, etc.)	-	
Restriction of Hazardous Substances (RoHS) Directive	Banning the use of hazardous substances in electronic products	From 2006	European Commission, 2011
End-of-life Vehicle (ELV) Directive	Minimum reuse and recycling targets of 85% by average weight per vehicle	From 2015	European Commission, 2000
	Minimum reuse and recovery targets of 95% by average weight per vehicle	From 2015	
	Restriction of hazardous substances in vehicles placed on the European market	From 2003	
Voluntarily Pledges	10 million tonnes of recycled plastic production by Circular Plastic Alliance (CPA)	By 2025	European Commission, 2022b; 2020b
	Minimum recycled content (25–30%) in new electronic products placed on the European market	By 2025 or 2030	Philips, 2020; Electrolux Group, 2020; Whirlpool Corporation, 2018; Sandoval, 2018
	Use of recycled content (20 – 25% by weight) in new cars placed on the European market	By 2030	Maury et al., 2022; Carroll, 2021; Volvo, 2018

1.4 ASSESSMENT TOOLS OF PLASTIC WASTE MANAGEMENT SYSTEMS

1.4.1 Material flow analysis

Several studies have investigated the performance of current plastic waste management systems by estimating the amount of plastic waste recycled effectively over the amount of plastic waste generated. MFA principles are often used to trace the fate of plastic throughout a defined system (e.g., global plastic flows, European plastic flows, or plastic flows at sorting/recycling plant level). The outputs of MFA are mass balances and compositional data, which are benchmarked against specific (evaluation) indicators for analyses and results comparison (Kleinhans et al., 2021a; Geyer et al., 2017; Ryberg et al., 2019; Kawecki et al., 2018; Eriksen et al., 2020; Antonopoulos et al., 2021).

MFA itself is defined as a systematic assessment of the flows and stocks of materials within a defined system, space and time. MFA connects the sources, pathways, and final sink of materials, which resulted to material balances comparing all inputs, stocks, and outputs of a process (Brunner & Rechberger, 2005). An example of MFA system can be found in Figure 1.4, which also consists of a series of terms to understand how MFA is performed. The *system* or *system boundary* is the actual spatial and temporal definition of the MFA investigation, which includes the flows of material entering and leaving the defined system. The terms *material* in MFA stands for both substances and goods. A substance is defined as an 'element' or 'compound' composed of uniform units (e.g., carbon dioxide), while goods are made up of substances or group of substances that have an economic value, which can be positive (such as resources) or negative (such as waste). Material can enter and leave a *process*, which refers to act of transportation, storage, or transformation of materials. Different processes within a defined system in MFA is connected by *flows* (or *fluxes*), which refers to process inputs and outputs. *Stocks* are defined as material accumulation (mass) within the analyzed system measured in mass unit, which can increase (material accumulation) when inputs are higher than outputs or decrease (material depletion) when outputs are higher than inputs. In MFA studies, typically a process is illustrated in box shape whereas a flow (flux) is illustrated in oval shape, as shown in Figure 1.4 (Brunner & Rechberger, 2005).

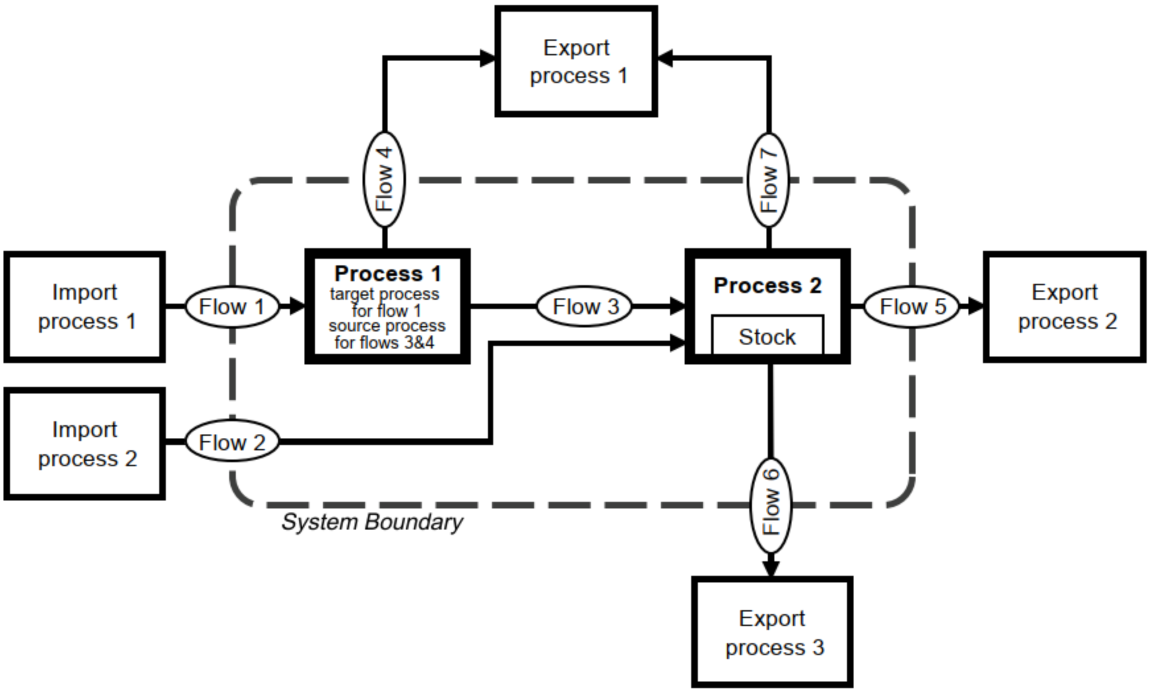

Figure 1.4 Example of material flow analysis system illustrating selected terms. *Source*: Brunner & Rechberger (2005).

MFA has been applied to investigate plastic waste management system at different levels (global, regional, country, or factory level). Geyer et al. (2017) investigate the fate of global plastic waste generation and treatments between 1950 – 2015. The results of the study suggest that the cumulative plastic waste generated (from 1950 – 2015) reached 6,300 Mt. Of this, only 9% (600 Mt) have been recycled, 12% (600 Mt) have been incinerated, and the rest (4,900 Mt; 79%) are accumulating in landfills or the environment. A study by Ryberg et al. (2019) indicates that a substantial amount of micro- and macro-plastic is lost to the environment. It is estimated that approximately 6 Mt of marco-plastic and 3 Mt were found in the environment in 2015 globally (annual figure from 2015 data; Ryberg et al., 2019). These studies conclude that a comprehensive assessment framework is needed to assess potential improvements within global plastic waste management systems.

The fate of plastic waste throughout European plastic waste management systems is investigated by Kawecki et al. (2018) and Eriksen et al. (2020), focusing on the flows of PET, PP, HDPE, LDPE, PVC, and (E)PS. The results of these studies suggest that ineffective (or non-existing) source separation of some plastic waste streams as one of the main reasons for low plastic recycling rates in Europe. Considerable material losses also occur during sorting and recycling of plastic waste. When looking at plastic waste sorting and recycling, certain polymer types (e.g., PET and HDPE) perform better than others (e.g., LDPE and PP films) because of their simpler composition and a more well-established infrastructure. According to Kawecki et al. (2018), PET and HDPE packaging performs better than LDPE and (E)PS packaging due to better collection rates and sorting yield. This can be explained by the fact that separate PET and HDPE packaging waste collection was implemented way before other packaging formats, as illustrated in the PMD collection system in Belgium and The Netherlands (Roosen et al., 2022; Brouwer et al., 2019). Through MFA, Kawecki et al. (2018) also indicate that the low plastic recycling rate from electronic and automotive sectors is caused by considerable waste export leaving the European market. Furthermore, Eriksen et al. (2020) argue that poor packaging designs can cause relatively low plastic recycling rate for certain packaging formats (e.g., multimateral packaging). Designing packaging in compliance with DfR guidelines (e.g., promoting monomaterial packaging) can substantially increase plastic packaging recycling rates. The MFA results of Eriksen et al. (2020) are consistent with Horodytska et al. (2019) study, which suggests that multimaterial multilayer film recycling is not happening at an industrial scale yet. The problem with multimaterial multilayer packaging film lies in the combination of noncompatible materials (i.e., polymers, aluminum, paper, etc.) to be correctly sorted and effectively recycled.

Moreover, several studies also indicate that a substantial amount of waste is 'missing' and not accounted for in statistical databases in Europe (SYSTEMIQ, 2022; Agora Industry, 2022; Material Economics, 2022). Underestimation of service lifetime of plastic applications, waste quantity in mixed (municipal) waste statistics, and undocumented flows (e.g., illegal plastic waste export) are regarded as potential reasons for the 'missing plastic' in Europe. One of the suggestions to trace the flows of the so-called 'missing plastic' in Europe is thus a better estimation of product lifetime combined with robust material flow modeling of plastic waste (Geyer et al., 2017; SYSTEMIQ, 2022). This is especially important for plastic-containing products with a relatively longer service lifetime, such as in electronic, automotive, or building and construction sectors. Kawecki et al. (2018) and Wang et al. (2013) have pointed out the importance of considering the lifetime distribution model to estimate the waste generation in electronic sector better.

The evaluation of plastic waste management performance using MFA can also be performed at a national level, as shown by Klotz et al. (2022) for plastic flows in Switzerland, Picuno et al. (2021) for plastic packaging in Germany, and Van Eygen et al. (2018) for plastic packaging flows in Austria. The findings of the three studies suggest that big efforts are still needed to reach the European recycling target (i.e., 55% by 2030). The plastic packaging recycling rate in Switzerland is estimated to be 17% in 2017 (Klotz et al., 2022), 26% in Germany in 2017 (Picuno et al., 2021), and 23% within the Austrian market in 2013 (Van Eygen et al., 2018). These findings are consistent with Antonopoulos et al. (2021) studies which suggest that significant improvements in the plastic packaging waste management systems in Europe are needed to reach the 55% recycling target set out in PPWD. The plastic packaging recycling rate in Europe is estimated to be 14–25% (Antonopoulos et al., 2021), and low collection rates and sorting yield are regarded as the main bottlenecks within the plastic packaging waste management systems in Europe (Picuno et al., 2021; Klotz et al., 2022; Van Eygen et al., 2018; Antonopoulos et al., 2021). Thus, it is evident that MFA can be used to assess the performance of plastic waste management at a national level and to underpin the potential improvements needed to reach recycling targets.

Several studies also use MFA principles to trace the fate of plastic waste in sorting or recycling plants (Kleinhans et al., 2021a; Cimpan et al., 2016; Larrain et al., 2021). Kleinhans et al. (2021a) examine the sorting behavior of up to 17 different plastic packaging formats throughout PMD and P+MD sorting facilities in Belgium. The granular modeling approach by Kleinhans et al. (2021a) illustrates how various packaging items behave throughout different sorting units and shows the limitation of the current state-of-the-art plastic packaging sorting process. For example, certain packaging items (e.g., PET trays) are missorted at the ballistic separator because they are light waste and compressed (and flattened) during waste collection. Moreover, the MFA results show that some packaging items (e.g., black bottles, multilayer multimaterial films, etc.) are not adequately sorted because NIR machines poorly detect them. Through MFA, Kleinhans et al. (2021a) also determine the expected bales' quality, which is crucial for further recycling. At the MR facilities, the study by Larrain et al. (2021) shows that the recycling yield of MR of plastic varies from 70–90% depending on the feedstock (bales) quality. A relatively lower recycling yield of MPO bales through MR (71%; Larrain et al., 2021) is mainly caused by higher impurities (e.g., non-PO matrials, organic waste, etc.). Larrain et al. (2021) also show that considerable mass losses in MR occur at sink-float separation and windshifter.

1.4.2 Multivariate input-output analysis

Several methods have been proposed to estimate plastic waste generation with a longer lifespan such as plastic in WEEE, ELV, CDW, etc. Generally, methods for quantifying such waste streams need to consider the disposal rate by the consumers based on socio-economic conditions (Wang, 2014). Multivariate input-output analysis (MIOA) is a commonly used method with multiple variables, which quantitatively describes the dynamics, magnitude, and interconnection of product sales, stocks, and lifespans (as illustrated in Figure 1.5). Generally, products flow into society (sales) and accumulated (stock) until the owner disposes of the items after a certain period (lifespan) hence flow out as waste (Wang et al., 2013). In such a sales-stock-lifespan model of MIOA, the relationship between these variables is taken into consideration when determining the outflow of the waste. Information on the stock age composition and disposal age composition can be extracted from each data point for any historical year owing to the lifespan distribution profile and statistical sales data (as illustrated in Figure 1.5). It means that the percentage and amounts of products that are accumulated in the market (stock) and flowing out as waste based on their *age* can be determined in this model (Figure 1.5).

The MIOA model is developed based on a more detailed calculation regarding the lifespan distribution using Weibull Distribution (Wang, 2014; Wang et al., 2013), acknowledging that the product annually accumulated in the market have different disposal (obsolete) rate depending on the consumers' behavior. In other words, the model considers the probability of products to be discarded depending on the obsolescence rates of products during the evaluation period. From such approaches, a more reliable waste generation can be quantified; thus potential plastic recycling can be estimated.

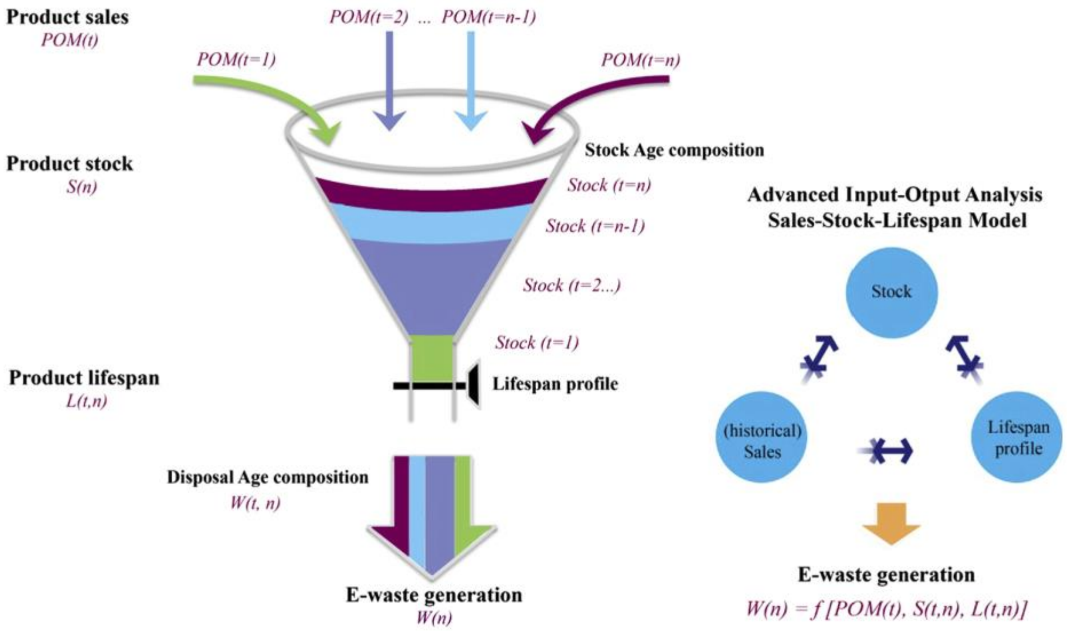

Figure 1.5 An illustrative diagram of multivariate input-output analysis applied in the sales-stock-lifespan model. Source: Wang et al. (2013)

1.4.3 Life cycle assessment

Life Cycle Assessment (LCA) is commonly used as a tool to calculate the potential environmental burdens of products or services over their life cycle stages, i.e., from material extraction to waste treatment (Rigamonti et al., 2020). Typically, an LCA study is developed and applied according to the international standards ISO 14040 and 14044 consisting of four steps, namely i) goal and scope definition, ii) inventory analysis, iii) impact assessment, and iv) interpretation (ISO, 2006), as shown in Figure 1.6.

The first aspect of goal and scope definition is determining the functional unit and system boundaries. A functional unit in an LCA study is used as the reference to evaluate the components within a single system or multiple systems (European Commission, 2010). The selected functional unit within an LCA study defines the quantitative and qualitative aspects of the function(s) or service(s) provided by the product being evaluated. After the functional unit is defined, a system boundary can be defined to decide which part of a system is included and which part can be excluded or cut-off (ISO, 2006). The system boundary of an LCA study can vary at different scales: a specific process within a company (process level), a process chain including all supporting utilities at the company level (“gate-to-gate”), a process chain including material extraction and processing to product manufacturing ("cradle-to-gate”), or a full life cycle stages from material extraction to end-of-life treatment ("cradle-to-grave") (European Commission, 2010).

The next stage of an LCA study is the construction of life cycle inventories, which involves the compilation of inputs and outputs throughout the life cycle stages defined in an LCA study (system boundary) (ISO, 2006). The life cycle inventories are then converted to environmental impact at the third phase of an LCA study, namely impact assessment. The collected data can be site specific, such as a real production process chain (Khoo, 2019) or modelled with software (Ügdüler et al., 2021). Alternatively, generic or averaged data can be used. For this purpose, a foreground and background system can be chosen within the selected list of unit operations. This former is defined as "those processes of the system that are directly affected by decisions analyzed in the study" or as "case-specific process", whilst the latter is defined as "those processes that are operated as part of the system but that are not under direct control or decisive influence of the producer of good" or as "market average processes" (European Commission, 2010). For background systems, databases can be used such as Ecoinvent (Ecoinvent, 2023), the Life Cycle Data Network (LCDN) (European Commission, 2023a), etc.

The interactions on (material or energy) inputs and outputs from the inventory phase are converted into environmental impact. The impact assessment phase evaluates the environmental impact using specific assessment methodologies (characterization models). For this purpose various assessment methodologies can be selected, such as ReCiPe2008 (Goedkoop et al., 2008), ReCiPe2016 (Huijsbergt et al., 2016), Environmental Footprint Methodology (European Commission, 2021), TRACI (Bare et al., 2012), etc. Finally, the interpretation phase of an LCA study covers several key elements such as i) identification of the significant issues based on the LCA results, ii) key conclusions, iii) limitations, and iv) recommendations derived from the LCA study (ISO, 2006). It is important to note that an LCA study is an iterative process by nature and the LCA results can be improved over time (European Commission, 2010).

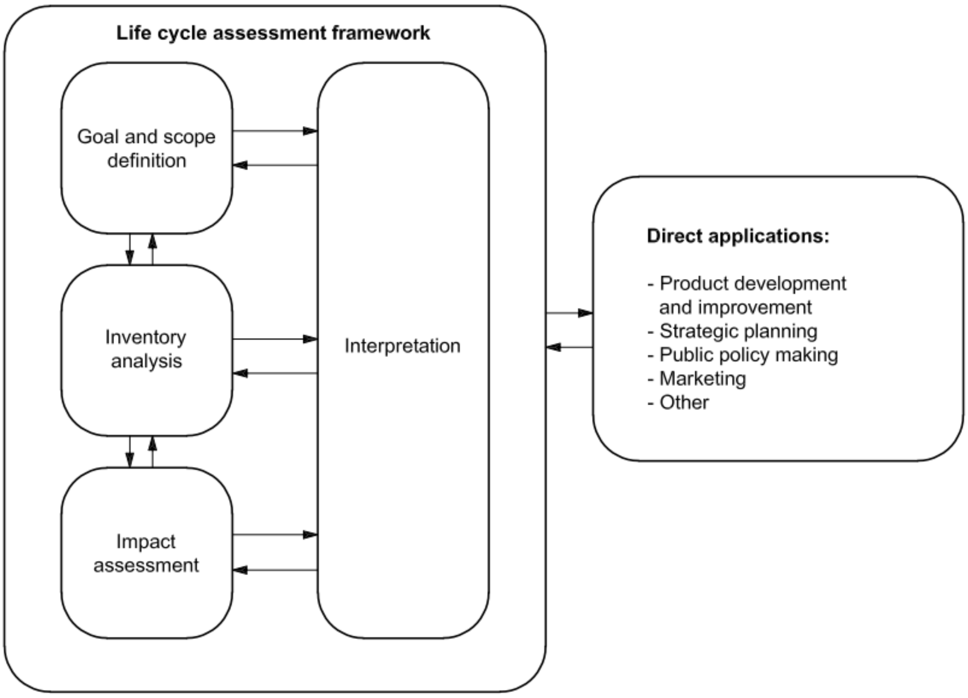

Figure 1.6 Four steps of a life cycle assessment by ISO 14040/44. Source: ISO (2006)

1.4.4 Economic assessment

Economic assessment within plastic waste management is widely used (De Meester, 2013). In the context of economic assessment, cost-benefit analysis (CBA) can be used, which is defined as an analytical tool for judging the economic advantages or disadvantages of an investment decision by assessing its costs and benefits in order to assess the welfare change attributable to it (European Commission, 2015). Another method that is typically used to assess the economic performance of plastic waste management is techno-economic assessment (TEA), which consists of four steps: i) market study to determine market prices and volumes, ii) development of process flow diagram, mass balance, and energy balance, iii) economic analysis, where investment criteria are used to assess profitability of the system, and iv) risk analysis to assess the uncertainty on the results (Thomassen et al., 2019). Finally, other methods, such as life cycle costing (LCC), can be used to assess the economic potential of emerging technologies or systems. The European Commission (2023b) defines LCC as an analytical tool that considers all the costs that will be incurred during the life time or the product, work, or service. The LCC method focuses on important cost aspects such as energy or labour, making conclusions on emerging technologies' economic profitability infeasible (Thomassen et al., 2019). For purposes of investigating economic improvements in plastic waste management, CBA can be used as a supporting decision-making tool to assess economic feasibility, which examines all aspects of the operational costs incurred and revenue generated from recycling systems. CBA is indeed, to some extent, developed based on LCC and TEA principles (Torkashvand et al., 2021).

1.4.5 Sustainability assessment of plastic waste management

The sustainability performance of waste management systems has been evaluated by combining the information on plastic flows (e.g., MFA or MIOA) with environmental, economic, and social impact assessment methods. Studies from Civancik-Uslu et al. (2021), Huysveld et al. (2022), Huysman et al. (2017), and Horodytska et al. (2020) compare the environmental footprints of different plastic waste recycling options. Cimpan et al. (2016) and Larrain et al. (2021) assess the economic balance (i.e., capital investment, annual costs, and revenue) of plastic packaging sorting and MR process. Taelman et al. (2020), Milios et al. (2018), and Sanjuan-Delmás et al. (2021) investigate the social impact of different waste management systems in several European cities (e.g., waste management in Ghent vs. Hamburg). In all the abovementioned studies, MFA is used as the primary modeling input to run environmental, economic, and social impact assessments, which determine the quality of impact assessment results. Schwarz et al. (2021), Huysveld et al. (2022), and Vollmer et al. (2020) studies also show that the environmental impact assessment of new emerging plastic recycling technologies (e.g., SBR and CR) relies on robust material flow modeling such as feedstock characterization and qualities. Caro et al. (2023) and Tabrizi et al. (2021) have proposed a mass balance accounting method (using MFA modeling alike) to trace plastic flows through complex plastic recycling chains to monitor the attainment of European plastic recycling targets, i.e., recycling rates, recycled content targets, etc.

MFA is also the backbone to develop and apply circularity indicators, which are used as one of the monitoring tools to make strategic decisions towards a circular economy. One example of a circularity indicator is the Recyclability Benefit Rate (RBR), which measures the potential environmental savings related to the recycling of a product over the environmental burdens of virgin production followed by disposal (Huysman et al., 2015; 2017). Specifically, the RBR indicator aims to improve resource efficiency in European product policies by using waste as a secondary resource (Ardente and Mathieux, 2014). Another circularity indicator, developed by Huysman et al. (2017), is called the Circular Economy Performance Indicator

(CPI), which measures the ratio of the actual obtained environmental benefit over the ideal environmental benefit taking into account quality factors of the secondary material, which is not fully addressed in RBR indicator. The CPI thus looks deeper at quality aspects in combination with a typical circularity indicator approach using MFA or LCA studies (Huysman et al., 2017). However, the definition and operational framework for quality aspects of recycling are still lacking to date (Tonini et al., 2022). Several studies have tried to make comprehensive circularity indicators by combining quality and monetary aspects such as the technical processability and properties (Demets et al., 2021; Chaon et al., 2020), functionality or circularity potential (Vadoudi et al., 2022; Vadenbo et al., 2016; Eriksen & Astrup, 2019; Eriksen et al., 2018) and technical and market substitutability (Roosen et al., 2023b; Huysveld et al., 2022; Vadenbo et al., 2016). A recent study by Roosen et al. (2023b) proposes an operational framework to measure the quality of recycling by taking into account three dimensions of recycling process, namely environmental impact (through LCA studies), substitutability rate (from technical and economic perspectives), and materials durability in the economy (i.e., in-use lifetime). Lastly, for multi-output processes, Broeren et al. (2022) and Caro et al. (2023) also offer their view on measuring circularity by tracing different products' yield and losses (at different stages) of various recycling technologies (e.g., plastic-to-plastic yield from pyrolysis; Broeren et al., 2022).

1.5 RESEARCH OBJECTIVES

It is evident that the European government and the plastic industry have ambitious targets to realize a circular economy for plastics. Current plastic waste management systems need substantial improvements to collect and sort more plastic waste for further recycling process. However, scientific-based evidence related to the extent to which waste management infrastructure could be improved and potential impact towards plastic circularity in Europe is scare. Moreover, the economic feasibility and environmental consequences of improving plastic waste management system also needs to be investigated.

To provide insights towards the potential technological, economical, and environmental improvements of potential future scenarios in the plastic waste management system in Europe, material flow analysis (MFA) has been used in this PhD research. Thus, this PhD research aims at providing scientific evidence of the contribution of technological advancements towards achieving a circular economy for plastic and the attainment of European recycling targets, such as recycling rates and recycled content targets. Particularly, the following chapters and objectives are addressed in this PhD research:

A general structure linked to this PhD research is presented in Figure 1.7. **Chapter 1** of this book highlights current state-of-the-art plastic waste management systems and challenges to increase plastic recycling rates. In addition, several key legislations and (voluntarily) pledges to realize sustainable plastic recycling chains in Europe are elaborated. Lastly, a few relevant studies attempting to evaluate the performance of plastic recycling chains at different levels (e.g., global, regional, national, or factory) are highlighted.

Chapter 2 of this PhD research investigates the fate of plastic waste throughout plastic recycling chains in Europe in 2018 (i.e., *status quo* as a benchmark) using a MFA modeling. This analysis includes the ten most used plastic types in Europe from five key sectors: packaging, electronic, automotive, construction, and agriculture sectors. The plastic waste flows in the *status quo* scenario (2018) are used as a benchmark to compare prospective plastic waste flows in 2030 when the management infrastructure is improved (e.g., collection rate, sorting yield, etc.). Particularly, five potential future scenarios in 2030 are considered by projecting the plastic waste quantities, improvements in waste management infrastructure, and various recycling options (MR, CR, and/or SBR). Five evaluation indicators are selected to appropriately analyze and compare the MFA results in 2018 and 2030 scenarios, namely end-of-life recycling rates (EoL-RR), plastic-to-plastic rate (P2P), plastic-to-chemical rate (P2C), and plastic-to-fuel rate (P2F). Thus, *the first objective of this PhD presented in Chapter 2 is to investigate the current (status quo scenario in 2018) and potential future flows of plastic waste (in 2030) throughout plastic recycling chains at European level, including the impact of the emerging plastic recycling technologies on plastic circularity in 2030*.

Although technological advancements such as CR and SBR are quite promising to deal with plastic waste in Europe, each sector has its own unique challenges to recycle plastic waste (e.g., plastic recycling problems in electronic versus packaging sector). Therefore, the next chapters focus on investigating future plastic waste treatment in two specific sectors (as case studies): the electronic and packaging sectors. In **Chapter 3**, multivariate input-output analysis (MIOA) and MFA models are developed to gain better insights into the WEEE recycling chain. The MIOA is selected because this approach considers the dynamic interconnection between product sales (in the past years), stock flows (product accumulation in the economy), and electronic product lifetime distribution. Combining MIOA and MFA enables forecasting the amount of WEEE generation in 2030 (and its disposal age composition) and regranulates production from WEEE recycling. The WEEE recycling chains in The Netherlands and Belgium are selected as the system boundary and three electronic devices from small household appliances group are considered: vacuum cleaners, coffee machines, and electric shavers. This study includes a sensitivity analysis to identify the bottlenecks of plastic recycling from WEEE. The sensitivity analysis also provides information on potential improvements needed within WEEE recycling chains to reach European WEEE recycling targets. Finally, the MIOA model has the ability to determine the origin of the incoming waste in 2030, i.e., the disposal age composition. Through this modeling approach, the inherited hazardous chemicals (e.g., brominated flame retardants) can be identified, which is relevant information for WEEE recycling industries because a separate treatment of plastic-containing hazardous retardants is mandatory under WEEE Directive. Therefore, *the second objective of this PhD research is to present a better estimation of WEEE generation and regranulates production from WEEE recycling in 2030, allowing the identification of potential bottlenecks, technological advancements, and inherited legacy chemicals from WEEE recycling systems*.

Chapter 4 of this research focuses on flexible packaging waste recycling from households (e.g., LDPE film, PP pouches, etc.) because flexible packaging is considered a difficult-to-recycle waste with relatively low recycling rates in Europe. Most household waste management infrastructure is to date developed to process rigid plastics (e.g., PET bottles or HDPE bottles). When recycled, current regranulates from flexible packaging are often considered inferior to virgin plastics and are often used for open-loop applications such as park benches or horticulture products (e.g., garden pots, buckets, etc.). As a step to improve flexible packaging recycling rates and the use of regranulate from flexible packaging, an improved MR process is developed, such as *Quality Recycling Process* (QRP). This improved MR process introduces a more rigorous pre-treatment of separately collected and sorted flexible packaging waste, such as additional sorting using NIR, hot washing, double melt filter during extrusion (with degassing unit), and deodorization steps. As a result, QRP produces new valuable outputs (PE Film Natural and PP Film regranulates) suitable for more demanding applications such as shrink film, sealable pouches, standing pouches, etc. The recycling performance of QRP versus conventional MR process is investigated by developing and combining a MFA and cost-benefit analysis (CBA) model. Four evaluation indicators related to quantity (process yield and net recovery) and quality (polymer grade and transparency grade) are selected and applied to compare the recycling performance of QRP versus conventional MR. The MFA results also provide granular compositional analyses at the bales, flakes, and regranulate levels, which are compared with experimental compositional analyses from the actual flakes and regranulates of QRP for model validation. A sensitivity analysis is also carried out to identify the impact of potential variations of the MFA modeling inputs on the recycling performance. The MFA and CBA models are developed based on inputs from mechanical recyclers, machine builders, and literature. The CBA results indicate the estimated capital investment, annual costs, and revenue from QRP operation compared to conventional mechanical recycling. *Hence, the third objective of this PhD research is to investigate the recycling performance of flexible packaging waste via improved mechanical recycling (by using QRP as a case study) compared to conventional mechanical recycling*.

In **Chapter 5**, the economic viability of non-household flexible packaging waste recycling from urban areas is investigated and discussed. The non-household sector also generates a substantial amount of flexible packaging waste, yet waste management of non-household plastic waste receives considerably less attention than household waste. Currently, collecting and recycling non-household plastic waste is limited because of scarce information on the waste quantity, composition, and economic feasibility. To improve the status quo, a CBA model is developed and applied, consisting of a logistic simulation (i.e., non-household plastic waste collection), non-household plastic waste flows through mechanical recycling processes, and economic assessment of the mechanical recycling process. The urban areas of the City of Ghent–Belgium and its twelve neighboring municipalities are considered in the system boundary. Particularly, three different collection frequencies (weekly, fortnightly, monthly) and two different mechanical recycling plant layouts (basic and advanced configuration) to deal with the non-household plastic waste from the considered urban areas are investigated and discussed. The basic mechanical recycling plant consists of shredding, washing, drying, and extrusion steps, while the advanced mechanical recycling plant layout adds NIR separation and hot washing steps. The waste quantity and composition data are collected from real waste sampling in the City of Ghent–Belgium. The economic parameters are collected from relevant literature related to flexible packaging waste recycling chains. Furthermore, a sensitivity analysis on the residual content (i.e., contamination level at source separated waste) is carried out to assess the impact of poor source separation practices by individuals (at offices, stores, schools, restaurants, etc.) on the overall non-household plastic waste recycling performance. Lastly, the last part of this chapter compares the greenhouse gas (GHG) emission (in kg CO_2-eq) associated with producing regranulates from non-household plastic waste recycling (mechanical) versus current virgin PE granulate production and PE waste incineration (i.e., current linear economy as benchmark). Therefore, *the fourth objective of this PhD research is to investigate the economic feasibility of collecting and recycling the 'forgotten' non-household plastic film waste from urban areas complemented by its associated carbon footprint accounting*.

In **Chapter 6**, the preliminary assessment of recycled content availability for household and non-household flexible packaging is investigated. A MFA model is developed to trace the flows of flexible packaging waste throughout end-of-life treatment in Europe in 2030, assuming that the flexible packaging design will improve (e.g., more mono-material in the market), more selective collection of flexible packaging at the source (e.g., more separate collection like P+MD system in Belgium), sorting yield for mono- and multi-material flexible packaging will improve (e.g., by implementation of digital watermarks), and recycling yield will improve (e.g., advancement in catalytic pyrolysis or advanced mechanical recycling). Five scenarios are developed to identify impact of mechanical recycling and pyrolysis as complementary technologies to reach the (proposed) recycled content targets in Europe, namely: 35% recycled content for non-contact-sensitive and 10% recycled content for contact-sensitive flexible packaging in 2030 (European Commission, 2022a). In this study, the recycled content availability is defined as the ratio between the recycled plastic production (from mechanical recycling and/or pyrolysis) over the plastic demand for flexible packaging in 2030. Next to quantity-based modeling to estimate the quantity of recycled plastic production, the associated capital investment focusing on the mechanical recycling and pyrolysis infrastructure to reach the recycled content targets is estimated, which is based on the economic factors (in €/tonne input to mechanical recycling or pyrolysis) found in literature.. Hence, *the fifth objective of this PhD research is to investigate the feasibility of achieving recycled content targets for flexible packaging, as mandated (proposed) by the European Commission in 2030*.

Finally, **Chapter 7** discusses the general conclusion that can be drawn from the presented studies and future perspectives for further research. Particularly, this chapter highlights the importance of MFA in evaluating plastic waste recycling systems and combining

MFA with other assessment tools (e.g., quality of recycling aspects) for a more comprehensive sustainability and circular economy evaluation of plastic in Europe.

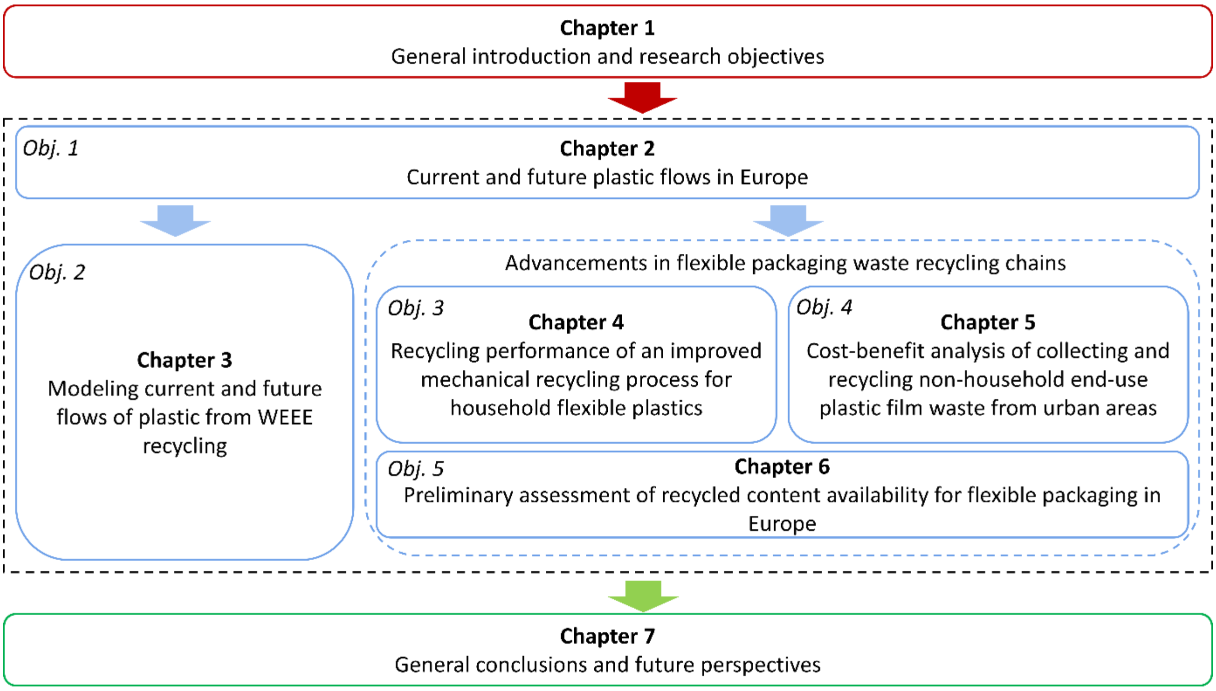

Figure 1.7 General structure of the research, organization of the chapters and their link to research objectives. Abbreviations: Mechanical Recycling (MR), Chemical Recycling (CR), Solvent-based Recycling (SBR), Objectives (Obj.)

CHAPTER 2: CURRENT AND FUTURE FLOWS OF PLASTIC WASTE IN EUROPE

Redrafted from:

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Chapter 2

Current and future flows of plastic waste in Europe

2.1 INTRODUCTION

According to Plastics Europe (Plastics Europe, 2020), Europe generated 29.1 Mt of plastic waste in 2019. Of the generated plastic waste in 2019, it is estimated that 19.6 Mt (i.e., 67%) was landfilled or incinerated and only 9.4 Mt (i.e., 33%) was sent to recycling facilities (Plastics Europe, 2020). Further, it is estimated that out of the plastic waste sent to recycling in 2019, only 4 Mt were effectively recycled, hence resulting in recycling rates of approx. 15–33%, depending on the calculation methods (Plastics Europe, 2020; Agora Industry, 2022). The 33% plastic recycling rate is calculated based on the quantity of plastic waste entering recycling facilities over the reported plastic waste generation (Plastics Europe, 2020). The 15% plastic recycling rate is calculated based on the quantity of recycled plastic production (after regranulation) over the total estimated plastic waste generated (i.e., reported plastic waste quantity plus the 'missing plastic') (Agora Industry, 2022). According to Material Economics (2022), the reported amount of plastic waste in Europe (i.e., 29.1 Mt) is deemed to be underestimated as substantial amounts are seemingly not accounted for (so-called 'missing plastic') in the statistical databases, e.g., municipal waste statistics. Studies from Agora Industry (2022) and SYSTEMIQ (2022) estimate that 7–15 Mt of plastic waste are 'missing' because of either underestimation of plastic in mixed (municipal) waste, underestimation of lifetime of plastic applications, or unidentified/undocumented flows (e.g., unauthorized waste treatment or exports of waste). Linked to the 'missing plastic', a few studies have also suggested that the flows of waste of electronic and electrical equipment (WEEE) and end-of-life vehicles (ELVs) are prone to illegal (unauthorized) end-of-life treatment because of the economic value within the waste streams (e.g., critical raw materials) (Williams et al., 2020; Baled et al., 2017). Moreover, one's waste might still be interesting for the others (Thapa et al., 2023). From this perspective, a considerable quantity of WEEE and ELV are not necessarily considered as 'waste' but as secondary products. This condition incentivizes shipments of WEEE and ELV to non-European countries when the cost of shipment is cheaper than domestic end-of-life treatment (Balde et al., 2017; Thapa et al., 2023). Thus, WEEE and ELV amongst other waste streams can be considered as part of the 'missing plastic'.

Reinforcing the efforts to improve the plastic circularity in Europe, the European Commission (EC) has enacted several regulations, along with (voluntarily) pledges made by stakeholders in the plastic value chain (e.g., by cars and electronic products manufacturers). For example, 55% of plastic packaging waste should be recycled by 2030 as stated in the Packaging and Packaging Waste Directive (PPWD) (European Commission, 2018c). Cars and electronic manufacturers also pledge to use 25–30% of recycled plastic in their new products by 2030 (Maury et al., 2022; Sandoval, 2018; Volvo, 2018). The Landfill Directive also limits municipal waste to be landfilled in 2035 by 10% (European Commission, 2018b). The complete list of relevant laws and pledges is available in Table 1.1. The regulations and pledges also aim to enhance the uptake of recycled plastic in new products (i.e., recycled content), which would increase the demand and potentially the price of recycled plastics (Maury et al., 2022, European Commission, 2022b). However several studies indicate that either the targets are not yet accomplished or significant improvements are still needed to achieve the targets. Lase et al. (2021) suggest that the recycled content targets in the electronic sector will be difficult to achieve in Belgium and The Netherlands due to inefficiencies in collection, sorting and recycling chains. Studies from Maury et al. (2022), Cardamone et al. (2022), and Williams et al. (2020) suggest that plastic from end-of-life vehicles (ELVs) are treated with less attention to polymer recovery and the 'reuse and recycling' target from ELV (i.e., 85%) stated in the End-of-Life Vehicles Directive (ELVD) is mainly achieved by recycling aluminum and metals from ELVs, leaving a substantial amount of plastic to be landfilled or incinerated. Similarly, a significant amount of plastics packaging waste is not separately collected, correctly sorted or recycled, while substantial improvements are needed to meet the 55% recycling target by 2030 stated in the PPWD (Picuno et al., 2021; Antonopoulos et al., 2021; Lopez-Aguilar et al., 2022; Van Eygen et al., 2018). It is thus clear that the circular economy for plastic needs an urgent boost.

The ways to improve plastic circularity and recycling rates in Europe are two-fold: implementation of plastic *production and use-oriented* solutions, and *end-of-life (EoL) treatment-oriented* solutions. *Production and use-oriented solutions* typically focus on improving products' design for easier EoL treatment (i.e., design-for-recycling principles), reducing material complexity (e.g., by changing from multi- to mono-material), reducing plastic use in a product (e.g., reduce packaging weight or unused space for packaging), and fostering new delivery business models (e.g., through promoting reuse) (OECD, 2022; SYSTEMIQ, 2022; Feber et al., 2020). On the other hand, the *EoL treatment-oriented* solutions focus on improving the existing waste management infrastructure and practices such as promoting separate collection, sorting per polymer group, and advancing recycling technologies (Ellen MacArthur Foundation, 2016; PRI, 2019).

Related to EoL solutions, today, mechanical recycling (MR) is still the most commercially used technique to recycle plastic (over 9.0 Mt processing capacity), while chemical recycling (CR) and solvent-based recycling (SBR) is treating only less than 0.2 Mt of plastic waste in Europe (Plastics Europe, 2019a). However, MR faces several challenges in treating plastic waste, such as thermal-mechanical degradation, the presence of legacy additives and chemicals, and inadequate technical properties of the final regranulates to meet the market demands (Ragaert et al., 2017; Simon and Martin, 2019; Eriksen et al., 2020). Also, potential degradation might occur by multiple rounds of recycling (Demori et al., 2015; Pérez et al., 2010; Schyns and Shaver, 2021; Arena and Ardolino, 2022). Several improvements can be implemented to tackle these MR challenges such as the implementation of advanced (pre-)treatment processes (e.g., deinking and deodorization), advanced washing (e.g., hot washing with detergents) and improved extrusion (e.g., double melt filtration) (Bashirgonbadi et al., 2022; Kol et al., 2021; Roosen et al., 2021; Demets et al., 2020). Nevertheless, even after elaborated sorting process, some plastic waste streams remain unsuitable for MR due to the heterogeneous composition (e.g., mixed of rubbers, thermosets, and thermoplastics), substantial level of hazardous substances (e.g., legacy chemicals from flame retardants), or multi-material structures (e.g., fiber-reinforced composites or multilayer packaging) (Cardamone et al., 2022; Arena and Ardolino, 2022).

On the other hand, several studies predict that CR technologies (i.e., pyrolysis, gasification, depolymerization) and SBR technologies (i.e., dissolution-precipitation, deinking, delamination) will play a big role in the future plastic waste treatment in Europe (Simon and Martin, 2019; Hann and Connock, 2020; Crippa et al., 2019; Manžuch et al., 2021). These technologies are claimed to have a higher tolerance in dealing with contaminated and complex waste streams, i.e., waste streams that are not recycled yet due to the limitation of current state-of-the-art MR (SYSTEMIQ, 2022; Cardamone, 2022; Arena and Ardolino, 2022; Vollmer et al., 2020; Solis and Silveira, 2020). Several plans to build CR plants have been announced such as gasification plant in Spain (treating non-recyclable mixed solid waste with 400,000 tonne/year capacity), pyrolysis plant in Spain and Belgium (treating mixed polyolefin and polystyrene with up to 65,000 tonne/year capacity), and chemical depolymerization plant in the United Kingdom, France, Belgium, and Spain (treating polyurethane; 2,000 tonne/year capacity and polystyrene; 15,000 tonne/year capacity) (Indaver, 2022; INEOS Styrolution, 2021; AIMPLAS, 2022). In this sense, CR and SBR technologies and are perceived as complementary to treat plastic waste streams that otherwise would have been landfilled or incinerated (Arena and Ardolino, 2022; Manžuch et al., 2021). From a life cycle perspective, diverting plastic waste streams from landfill, incineration, and export outside Europe (e.g., to African and Asian countries; Huisman et al., 2012; Jacobs et al., 2018) leads to environmental benefits by simultaneously avoiding such sub-optimal management practices and producing new secondary materials to replace production of virgin ones. Several studies indeed indicate better environmental performance of CR plastic waste compared to landfill and incineration (Arena and Ardolino, 2022; Vollmer et al., 2020; Demetrious and Crossin, 2019; Civancik-Uslu, 2021; Schwarz et al., 2021; Eschenbacher et al., 2022). However, while some studies have preliminarily investigated the environmental benefits (Arena and Ardolino, 2022; Vollmer et al., 2020; Civancik-Uslu, 2021; Jeswani et al., 2021) and technical feasibility (Kusenberg et al., 2022a; Kusenberg et al., 2022b; Larrain et al., 2020; Genuino et al., 2022) of some CR and SBR technologies, research on the performance and on the role and deployment of these technologies at industrial scale in the future Europe plastic waste management system is still scarce. Moreover, CR options such as pyrolysis and gasification produce not only monomers but also other base chemical products (i.e., benzene, toluene, xylene, wax, etc.) and fuels (i.e., hydrocarbons as synthesis gas or oil) (Kusenberg et al., 2022c; Kusenberg et al., 2022d). Nevertheless, such variety of outputs, while certainly contributing to plastic circularity, poses legal challenges as fuel- and energy-like outputs are not considered under 'recycling' in the Waste Framework Directive (WFD) (European Commission 2018a; European Commission, 2008).

In the context of the urgent need to increase the circularity of plastics, and to achieve (voluntary) targets or pledges, CR and SBR could play a pivotal role. However there is little quantitative evidence (and data available) on how big this contribution might be. Hence, study investigates the current and future flows of ten most used plastic waste throughout the plastic waste management systems (of five different sectors) in Europe. A prospective material flow analysis (MFA) model based on mass balance principles is developed and used. In prospective MFA, various scenarios are developed to evaluate the potential improvements within the plastic waste management system. The improvements can be associated with waste collection, sorting, and recycling phase, as well as the improvements in the production and use phase amongst others by varying the quantity and composition of the plastic waste generated in the future (Thomassen et al., 2022; Caro et al., 2023). For this purpose, six scenarios are developed and discussed: i) *status quo* scenario in 2018 (**S0**, as benchmark) and ii) five potential future scenarios in 2030 (**S1** – **S5**), including improving only collection, sorting, and MR as well as a combination of improved MR, CR, and SBR of plastic waste. One of the future scenarios also investigates the contribution of processing the so-called 'missing plastics' according to Material Economics (2022), Agora Industry (2022), and SYSTEMIQ (2022). The selection of suitable CR and SBR options in this study is determined by considering the capability of the CR and SBR options to treat plastic waste streams, including the type and composition of the streams as recently reported by the stakeholders to the EC.

For each scenario, a set of circularity indicators of plastic waste treatment are calculated based on MFA, namely: *EoL recycling rates* (EOL-RR), *plastic-to-plastic* (P2P), *plastic-to-chemicals* (P2C) and *plastic-to-fuels* (P2F) rates in order to assess the potential improvements when CR and SBR options are implemented at large scale. This study thus includes the amounts of materials produced such as polymers (i.e., recycled plastics from MR, CR, and SBR), base chemicals (e.g., wax, benzene, toluene, xylene from CR), and fuels (e.g., synthesis gas from CR). Lastly, the potential of recycled content availability in 2030 from different scenarios is quantified and discussed, which is based on the share of recycled plastic production (per sector) over plastic demand (per sector) in 2030.

2.2 MATERIALS AND METHODS

2.2.1 General Modeling approach

This study focuses on the ten most used polymers in the European Union (EU) 27+3 (Norway, Switzerland, and the United Kingdom) (Plastics Europe, 2019b) with high data availability in all life cycle stages from production to the EoL treatment (Eriksen et al., 2020; Kawecki et al., 2018) and considered as priority products within the plastic industry (Watkins et al., 2020). The ten polymers considered in scope within this study are (Linear) Low Density Polyethylene (LLDPE), High Density Polyethylene (HDPE), Polypropylene (PP), Poly(ethylene Terephthalate) (PET), Polystyrene (PS), Expanded Polystyrene (EPS), Poly(vinyl Chloride) (PVC), Acrylonitrile Butadiene Styrene (ABS), Polyurethane (PUR), and Polyamide (PA). These polymers are applied in different sectors with their specific use and EoL fate. The five sectors included in this study are: packaging, building and construction, automotive, electronic, and agriculture sector. Overall the selected polymers and sectors in this study cover 60% of the total reported plastic waste in 2018 in EU 27+3 (Plastics Europe, 2019a; Plastics Europe, 2019b). The other 40% of polymers that are not considered in this study (which is subjected for future research) include waste from household goods, textiles, and others (e.g., medical) (estimated to be 15–25%, based on Plastics Europe, 2019a) and some polymer types (e.g., Polycarbonate or Poly(methyl methacrylate), etc.) in packaging, electronic and automotive sectors (up to 35% of 'other polymers', based on Plastics Europe, 2019a and Plastics Europe, 2019b).

The MFA of the selected polymers is modelled by following four steps, following the methodology from previous studies (Antonopoulos et al., 2021; Eriksen et al., 2020; Kawecki et al., 2018). Firstly, the required inputs data for MFA model are gathered: (i) a process diagram of the current (and future) plastic waste management systems in EU 27+3, (ii) the respective transfer coefficients (TCs, in %) of each process, and (iii) the quantities of the selected polymers (in kilotonnes, kt). The TCs describe the partitioning of mass input(s) to output(s) for each process in the system. The MFA model quantifies the mass balance (in kt) throughout the defined system that is obtained by multiplying the mass input quantity with the TCs of each process in the system. Secondly, *status quo* scenario and five potential future scenarios are developed, representing the flow of the selected polymers in 2018 and potential flows in 2030, respectively. In order to model the mass flows in 2030, projections of waste quantities, improvements of the TCs, and recycling pathways (MR, CR, and/or SBR options) are implemented. Thirdly, the MFA results from the six different scenarios are assessed and compared by calculating four selected circularity indicators. Lastly, for each output (material flows and circularity indicators), the parametrical input uncertainties are propagated into output uncertainties. The uncertainty propagation with Monte Carlo simulation is performed and the standard deviation of the mass flow is calculated. The standard deviation is calculated assuming a Triangular Distribution (TD) of the dataset and the values are selected based on the relevant literature of plastic waste management in EU 27+3. Triangular Distribution is selected for this study, following Bisinella et al. (2016), mainly because i) statistical analysis and sampling of the selected parameters are not carried out, hence probability is assigned based on data variability found in literature, and ii) expert opinions are involved in determining TCs used in the model (i.e., preferred min, max, and mode values).

2.2.2 Defining the Scope of Recycling Technologies

After plastic waste is collected and sorted, MR, CR and SBR routes can be chosen. MR refers to mechanical reprocessing by means of shredding, washing, drying, and extrusion of polymers without breaking down the polymer chains. CR refers to a reprocessing technologies that break down the polymer chains and converts them into high added-value materials, such as oligomers, monomers, base chemicals, and hydrocarbons (solid, liquid, or gas) (Arena and Ardolino, 2022; Hann and Connock, 2020; Crippa et al., 2019; Manžuch et al., 2021). CR is an umbrella term that has been used to cover a broader set of technologies (Hann and Connock, 2020; Manžuch et al., 2021), such as thermal depolymerization (i.e., pyrolysis coupled with steam cracking or gasification coupled with Fischer-Tropsch Synthesis) and chemical depolymerization (i.e., glycolysis, methanolysis, etc.). SBR (also known as 'physical' or 'material' recycling) refers to material reprocessing by means of dissolving the polymer (or additives and pigments), in which the impurities is removed while the polymer is recovered through filtration or extraction phase (Crippa et al., 2019). A more detailed explanation of each technology at process level can be found in Appendix A–section 1.

2.2.3 Material flow analysis model development

2.2.3.1 Description of system boundaries and scenarios

This study focuses on Europe as EU27+3 as most of the datasets used in this study cover this region (Eriksen et al., 2020; Plastics Europe, 2019b, Kawecki et al., 2018; Watkins et al., 2020; Hestin et al., 2017). The diagram of the system boundaries can be found in Figure 2.1. The boundary comprises collection (i.e., source separation), sorting, and recycling, including the future potential plastic recycling using CR technologies in 2030. Figure 2.1 illustrates the waste management systems for plastic waste in the EU27+3 per sector. Detailed information on the waste management systems per sector can be found in Appendix A–section 2. There are three potential destinations of the plastic waste treatment: i) secondary materials to be used in the economy again, ii) waste streams that are sent for residual treatment (i.e., incineration or landfilling), and iii) waste export and/or informal waste treatments (Figure 2.1). As for the waste that is informally treated, the whereabouts of these flows are difficult to track. However, several studies suggest potential destination of these flows such as unreported recycling within EU 27+3, illegal export outside EU 27+3 (can be partially recycled), or leakage to the environment (Ryberg et al., 2019; Peano et al., 2020; Boucher et al., 2020). Note that in this study the legal waste export from EU 27+3 to other countries is merged together with 'informal waste treatments' due to limited data available to estimate exact fate of these flows, which has been pointed out by previous studies (Material Economics, 2022; Agora Industry, 2022; SYSTEMIQ, 2022). Plastic packaging waste export (including non-household waste) usually occurs after a certain degree of source separation (and sometimes partial sorting), i.e., 25% of the sorted bales are sent to countries outside EU27+3, as suggested by Antonopoulos et al. (2021). The plastic waste treatment of the exported waste at their final destinations (e.g., to Southeast Asia or African countries) is poorly reported, however it is a combination of recycling parts of it, with illegal dumping, unsanitary landfill, or open burning of residues (Tran, 2018; Wang, 2014; Liang et al., 2021; Petrlik et al., 2019; Chen et al., 2021; Lasaridi et al., 2018; Lase et al., 2021).

In this study, six scenarios are modelled (Table 2.1): one scenario as benchmark (i.e. the *status quo* in 2018) (**S0**), and five potential future scenarios in 2030 (**S1**–**S5**). The future scenarios take into account the feedstock type, composition, technology readiness level, and few improvements within the waste management systems (e.g., PP flex packaging waste could be separated from the mixed films streams; Lase et al., 2022). Table 2.1 also summarizes the supporting argumentations and assumptions of the five potential future scenarios in 2030, including information on the feedstock to CR and SBR options and their output(s). Moreover, it is assumed that the rate of waste export in 2030 will be significantly lower compared to the *status quo* scenario in 2018 because of two reasons. First, the implementation of CR and SBR is expected to allow more heterogenous waste streams to be reprocessed inside EU27+3 and second, stricter regulations of transboundary waste shipment (e.g., as mandated by UNEP Basel Convention; Lasaridi et al., 2018). It is important to note that indeed other scenarios might enroll in future work too, based on new developments and insights.

S1 illustrates the improvements of current state-of-the-art plastic waste management systems, following the trends of increased (source separation) collection rates and improved sorting and MR technologies (Maury et al., 2022; Lase et al., 2021; Antonopoulos et al., 2021). S1 assumes that only MR will be deployed to treat plastic waste. **S2** serves as 'explorative' projections, in which CR and SBR are assumed to outcompete MR (technologically and quantity wise) to deal with plastic waste. In S2, all sorted plastic waste, including the rejects from sorting and MR (i.e., 90–100% of mass, SYSTEMIQ, 2022) and mixed waste streams (i.e., 50–80% of mass, SYSTEMIQ, 2022) are assumed to be processed via CR. **S3** investigates the CR and SBR options as an alternative technology to MR option. In S3, it is assumed that CR and SBR options would take a small share of plastic waste stream that is already mechanically recycled (i.e., 1–20% mass), including rejects (i.e., 90–100% mass, SYSTEMIQ, 2022) and mixed waste streams (i.e., 50–80% mass, SYSTEMIQ, 2022). S3 assumes that MR still outcompetes CR and SBR (technologically and quantity wise) in processing sorted plastic waste. Also, it is assumed in S3 that CR and SBR will only process low quantities of sorted plastic in 2030 because they encounter several operational (and technical) issues to scale up the technologies at industrial scale (Jehano et al., 2022; Coates and Getzler, 2020; Tukker et al., 1999). Manžuch et al. (2021) and Kusenberg et al. (2022e) also indicates that significant improvements are needed to upgrade pyrolysis oil as well as feedstock quantity (and quality) for industrial steam crackers. Improvements are also still needed to scale up and optimize SBR technique (Jehano et al., 2022; Coates and Getzler, 2020; Tukker et al., 1999). Hence, S3 can also be perceived as 'sub-optimal' CR and SBR implementation, while MR is still chosen to be the main recycling technology. Furthermore, **S4** investigates CR as complementary technology to MR for waste streams that otherwise would be landfilled or incinerated. In S4, CR is assumed to process mixed polyolefin (PO) packaging (rigid and flexible) bales, mixed plastic packaging bales, 50–80% rejects, and 90–100% mixed waste streams. Notice that in the development of this scenario we strive to learn from, and to the extent possible align with, precedent studies that investigated the potential role of CR and SBR in the future in EU 27+3 (in S4, notably SYSTEMIQ, 2022; Arena and Ardolino, 2022; Hann and Connock, 2020; Manžuch et al., 2021). Finally, S5 is identical to S4 but accounts for the extra plastic mass (in kt) derived from the 'missing plastic', and explores the impact of processing 'missing plastic' on the overall performance of plastic waste treatment in EU 27+3. When the quantities of 'missing plastic' are normalized to the total plastic demand in Europe in 2019 and 2020 (Plastics Europe, 2020; Plastics Europe, 2019b), they account for 15–30% of the total plastic demand. A more detailed description of the explorative future scenarios and improvements per sector is reported in Appendix A–section 2.

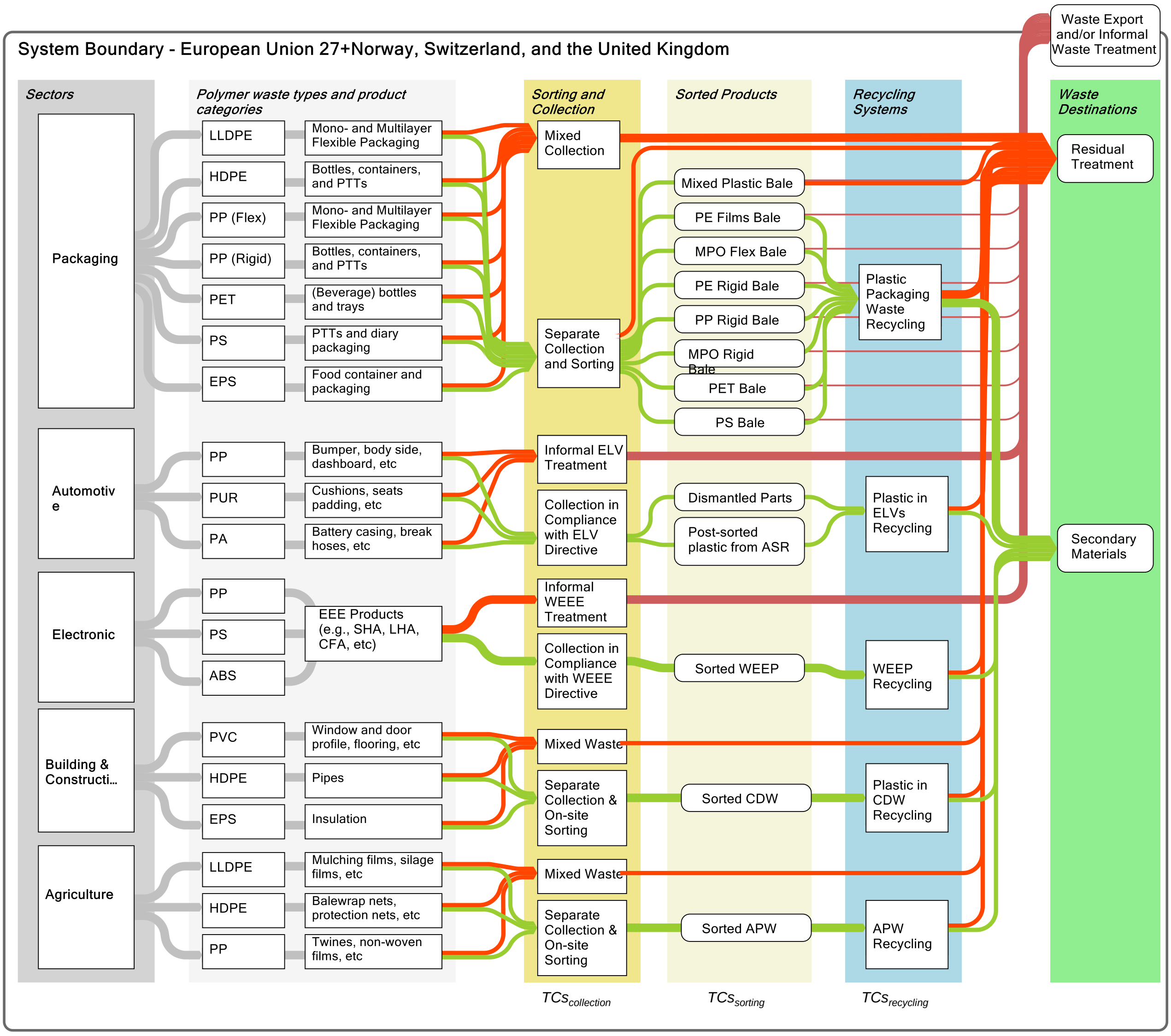


Figure 2.1 Conceptual diagram of the end-of-life treatment of the selected plastic waste from different sectors considered in this study. The thickness of the arrows does not represent mass/quantity. Abbreviations: ABS (Acrylonitrile Butadiene Styrene), APW (agriculture plastic waste), ASR (automotive shredder residue), CDW (construction and demolition waste), CFA (cooling and freezing appliance), EEE (electronic and electrical equipment), ELV (end-of-life vehicle), EPS (Expanded Polystyrene), HDPE (High Density Polyethylene), LLDPE (Linear Low Density Polyethylene), LHA (large household appliance), PA (Polyamide), PET (Polyethylene Terephthalate), PP (Polypropylene), PS (Polystyrene), PTTs (Pots, trays, and tubes), PUR (Polyurethane), PVC (Polyvinyl Chloride), SHA (small household appliance), TCs (transfer

coefficients), WEEE (waste electronic and electrical equipment), WEEP (waste electronic and electrical plastic).

Table 2.1 Overview of the developed scenarios for the MFA of plastic waste in European Union 27+Norway, Switzerland, and the United Kingdom. ABS (Acrylonitrile Butadiene Styrene), APW (agriculture plastic waste), CDW (construction and demolition waste), CR (Chemical recycling), ELV (end-of-life vehicle), MR (Mechanical recycling). PE (Polyethylene), PA (Polyamide), PET (Polyethylene Terephthalate), PP (Polypropylene), PS (Polystyrene), PUR (Polyurethane), PVC (Polyvinyl Chloride), SBR (solvent-based recycling), TCs (transfer coefficients), TD (Triangular distribution), WEEP (waste electronic and electrical plastic).

Scenarios	Supporting argumentation	Description	Input(s) for CR and SBR	Output(s) from CR and SBR
S0: *Status quo in 2018*	Benchmark (reference) scenario	Flows of plastic waste in 2018	Not applicable	Not applicable
S1: *Plastic waste treatment via MR in 2030*	Improvement in waste collection rate, sorting, and MR in 2030 towards breakthrough of (currently) known best practices in 2022 based on previous studies (Maury et al., 2022; Lase et al., 2021; Antonopoulos et al., 2021).	Improved TCs of waste collection rate, sorting, and MR yield in 2030 towards. The rejects (from sorting and MR) and mixed waste streams are sent to residual treatment	Not applicable	Not applicable
S2: *Plastic waste treatment via CR and SBR in 2030*	'Explorative' projections of plastic waste management in which CR and SBR options technologically outcompetes MR option	All sorted plastic are sent to CR or SBR, including 50–80% rejects from sorting and MR (assuming TD) and 90–100% mixed waste streams (assuming TD) in 2030	Dissolution-precipitation: ○ Sorted PVC and PS from CDW Chemical depolymerization: ○ Sorted PET bales (packaging sector) ○ Manually dismantled and post-sorted PA and PUR from ELVs Pyrolysis with Steam Cracking: ○ Sorted PE film, PP film, PE rigid, PP rigid, mixed PO (film and rigid), and mixed plastic film bales (packaging sector) ○ Manually dismantled and sorted PP from ELVs ○ Sorted PP, PS and ABS from WEEP ○ Sorted PE and PP from CDW and APW	Chemical depolymerization and dissolution-precipitation: ○ Polymer (and flakes for dissolution-precipitation) Pyrolysis with Steam Cracking: ○ Polymer ○ Base chemicals (e.g., wax, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis gas) Gasification with Fischer-Tropsch Synthesis: ○ Polymer ○ Base chemicals (e.g., tar, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis oil)

			Gasification with Fischer-Tropsch Synthesis: ○ Rejects from sorting and MR ○ Mixed waste streams	
S3: *Plastic waste treatment via MR and CR in 2030, in which MR option still technologically outcompetes CR and SBR options*	'Sub-optimal' CR and SBR implementation, while MR is still chosen as the main recycling option. Sub-optimal implementation is caused by operational (and technical) issues to scale up CR and SBR technologies to industrial scale (Jehano et al., 2022; Coates and Getzler, 2020; Tukker et al., 1999) and the need to optimize CR and SBR technologies (Manžuch et al., 2021; Kusenberg et al., 2022e).	Improved TCs of collection, sorting, and MR yield in 2030, while CR or SBR treats 1-20% mass (assuming TD) of sorted plastic waste from different sectors that is already sent to MR (in S1). Plastic waste in reject (50–80%, assuming TD) and mixed waste streams (90–100%, assuming TD) are also sent to CR in 2030	Dissolution-precipitation: ○ Sorted PVC and PS from CDW Chemical depolymerization: ○ Sorted PET bales (packaging sector) ○ Manually dismantled and sorted PA and PUR from ELVs Pyrolysis with Steam Cracking: ○ Sorted PE film, PP film, PE rigid, PP rigid, mixed PO (film and rigid), and mixed plastic film bales (packaging sector) ○ Manually dismantled and sorted PP from ELVs ○ Sorted PP, PS and ABS from WEEP ○ Sorted PE and PP from CDW and APW Gasification with Fischer-Tropsch synthesis: ○ Rejects from sorting and MR ○ Mixed waste streams	Chemical depolymerization and dissolution-precipitation: ○ Polymer (and flakes for dissolution-precipitation) Pyrolysis with Steam Cracking: ○ Polymer ○ Base chemicals (e.g., wax, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis gas) Gasification with Fischer-Tropsch Synthesis: ○ Polymer ○ Base chemicals (e.g., tar, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis oil)

S4: *Plastic waste treatment via MR and CR in 2030, in which CR options serve as complementary technology to treat waste streams that otherwise would be landfilled or incinerated*	CR as complementary technology to MR for waste streams that otherwise would be landfilled or incinerated such as mixed PO packaging (rigid and flexible) bales, mixed plastic packaging bales, rejects, and mixed waste streams. (SYSTEMIQ, 2022; Arena and Ardolino, 2022; Hann and Connock, 2020; Manžuch et al., 2021).	Improved TCs of collection, sorting, and MR yield in 2030, while CR treats mixed PO bales, mixed plastic bales, mixed waste (90–100%, assuming TD) and the reject streams from sorting and MR (50–80%, assuming TD) in 2030	Relevant to **S4** and **S5**: Chemical depolymerization: ○ Sorted PA from ELVs Pyrolysis: ○ Mixed Plastic bales and Mixed Polyolefin (MPO) bales Gasification: ○ Rejects from sorting and MR ○ Mixed waste streams	Relevant to **S4** and **S5**: Chemical depolymerization: ○ Polymer Pyrolysis with Steam Cracking: ○ Polymer ○ Base chemicals (e.g., wax, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis gas) Gasification with Fischer-Tropsch Synthesis: ○ Polymer ○ Base chemicals (e.g., tar, benzene, toluene, xylene, etc.) ○ Fuels (i.e., synthesis oil)
S5: *Plastic waste treatment via MR and CR in 2030, in which CR options serve as complementary technology to treat waste streams that otherwise landfill or incinerated, including the 'missing plastic' in 2030*		Identical to **S4**, with extra mass from accounting the 'missing plastic' (Plastics Europe, 2020; Plastics Europe, 2019b)		

2.2.3.2 Transfer Coefficients

For the purpose of calculating the uncertainty of the outputs, the TCs are assumed to have a Triangular Distribution (TD) to cover the diversity of the information from several studies. The full list of TCs for the listed scenarios in 2018 and 2030, together with the corresponding TD, can be found in Appendix A, Table A.1–A.5. Essentially, the study of Watkins et al. (2020) is used as the primary data points to model the flows of plastic in 2018. Additionally, a few studies are selected to be the key literature studies to compare or complement the TCs presented and used by Watkins et al. (2020). Table A.1–A.5 in Appendix A also provides information on the key literature that provide the TCs for the MFA modeling in the year 2018. Moreover, the approach to estimate TCs for the MFA model in 2030 as well as the TCs for CR and SBR are elaborated in the next sections.

2.2.3.2.1 Improved transfer coefficients in 2030 for collection, sorting, and mechanical recycling

To model the flows of plastic in 2030, it is assumed that the (source separation) collection rate, sorting yield, and MR yield will improve. For collection, improvements of the collection rates are extrapolated (and projected in 2030) using linear regression based on the past reported collection rate from several sources (Hestin et al., 2017; Global E-Waste Statistics Partnership, 2022; Eurostat, 2021; 2022b). From the linear regression calculations, the annual growth of the collection rates from 2018–2030 are obtained. For the packaging sector, the projection is based on Hestin et al. (2017) in 2012–2014. The annual growth of collection rate is estimated to be 4.0%. For the automotive sector, the collection rate is calculated as the share (or ratio) of the reported ELV recycling over ELV waste generated from Eurostat (2021) in 2010 – 2019. The annual growth of ELV collection rate is estimated to be 1.4%. Similarly, the collection rate of waste electronic and electrical equipment (WEEE) is calculated as the share (or ratio) of the reported WEEE generation (Global E-Waste Statistics Partnership, 2022) over the collected WEEE (Eurostat 2022b) in 2015 – 2019, with the annual growth of collection rate equal to 1.4%. For the building & construction and agriculture sector, it is assumed that the improvements of the collection rates are similar to the annual growth of the respective waste generation, i.e., annual growth of 0.8% for construction and demolition waste (CDW) and 1.0% for agriculture plastic waste (APW) (more in the next section).

The improvement of sorting and MR yields in 2030 are projected by assuming that the best practices of plastic waste sorting and recycling will be reached by 2035 (Antonopoulos et al., 2021). The assumption illustrates the optimization and widespread implementation of best available technologies in sorting and recycling different polymers across different sectors by 2035. The whole dataset in Table A.1–A.5 is used to calculate the uncertainty of the flows in 2030. Through this approach, the annual growth of sorting and recycling yields are calculated for different polymers across different sectors. More detailed information on the projections of collection rate, sorting, and recycling yields can be found in Appendix A, Figure A.10–A.22.

2.2.3.2.2 Transfer coefficients of chemical and solvent-based recycling

Pyrolysis, coupled with *Distillation, Hydrotreatment, and Steam Cracker*: The first steps of pyrolysis (after the plastic waste is separately collected and sorted) are shredding, cold washing (to remove contaminants like organic and inorganic residue; Larrain et al., 2021) and extrusion (using extrusive dehalogenation technique to remove substances like PVC and flame retardants; Kusenberg et al., 2022e). Afterwards, the plastic wastes are fed into the cracking and condensation reactor to produce pyrolysis oil that is distilled into naphtha and wax. The naphtha is fed into the steam crackers (with pyrolysis oil upgrading such as hydrotreatment) to produce monomers, which are used as a feedstock to recreate polymers again, and base chemicals. The TCs of the shredding, washing, and extrusion of MPO and Mixed Plastic bales are adopted from Lase et al. (2022) and Civancik-Uslu et al. (2021). The TCs from the cracking and condensation until (re)polymerization are obtained from literature (Civanvik-Uslu, 2021; Kusenberg et al., 2022a; Kusenberg et al., 2022b; Larrain et al., 2020; Jeswani et al., 2021; Genuino et al., 2022; Ghalomi et al., 2021; Zayoud et al., 2022; Kusenberg et al., 2022e).

Gasification, coupled with *Fischer-Tropsch Synthesis*: the processing of mixed waste and reject streams (from sorting and MR) via *gasification* starts with shredding the plastic waste into flakes followed by feeding them into gasification reactors to create mainly syngas with a small fraction of tar and char. The syngas is processed through Fischer-Tropsch Synthesis (FTS) to create monomers (incl. other base chemicals) that are used as feedstock for repolymerization processing. The TCs for converting plastic waste into syngas are obtained from literature (Mastellone, 2019; Lopez et al., 2018; Brems et al., 2015; Mastellone and Zaccariello, 2013; Arena, 2012). Lastly, the TCs for FTS and (re)polymerization are estimated from Zhao et al. (2021), Lee et al. (2008), and Jeswani et al. (2021).

Chemical depolymerization (i.e., glycolysis, methanolysis, aminolysis, etc.) is mainly implemented on sorted PET, PA, and PUR. The process starts with shredding and washing followed by depolymerization. The TCs for shredding and washing are estimated from Larrain et al. (2020) and Lase et al. (2022), while the TCs for depolymerization are obtained from Kol et al. (2021), Vollmer et al. (2020), Schwarz et al. (2021), Shen et al. (2010), and Nikje et al. (2011).

Solvent-based recycling (e.g., dissolution-precipitation) is employed to dissolve the polymer waste using a solvent, followed by the removal of additives through filtration or phase extraction to recover the dissolved polymer and the solvent (Crippa et al., 2019). The TCs for solvent-based purification are estimated from literature (Schwarz et al., 2021, Naviroj et al., 2019). More detailed information on the TCs for CR and SBR considered and used in this study can be found in Appendix A, Table A.6–A.10.

2.2.3.3 Waste categories and quantities

2.2.3.3.1 Waste quantity in 2018

Table A.11 shows the waste categories and quantities (in kilotonne, kt) used in this study, including some examples of the relevant products of the respective category. The estimation of waste quantities in 2018 is mainly based on Watkins et al. (2020). Within the packaging group, the share of mono- and multi-layer flexible packaging is estimated to be 80% and 20%, respectively (Lase et al., 2022). The share of bottles and pots, trays and tubes (PTTs) for PP rigid, HDPE, and PET is estimated from Hestin et al. (2017). The quantities of EPS foam are estimated from Hestin et al. (2017), i.e., 33% of the total PS in the packaging sector. In the automotive sector, the quantities of PA is estimated to be 12% of the total polymer used (Maury et al., 2022; European Commission, 2020b). Lastly, the quantities of PP and ABS used in the electronic sector are estimated from Lase et al. (2021) and European Commission (2020c).

2.2.3.3.1 Estimation of waste quantity in 2030 based on historical data extrapolation

The quantities of the selected polymers in 2030 are extrapolated using linear regression based on the historical waste generation found in statistical databases (Eurostat, 2021; 2022a; 2022b; 2022c), e.g., from 2010–2018 in packaging sector based on data availability for EU27+3 found in Eurostat (2022a), more in Appendix A, Figure A.23–A.27. Later, the information on the annual growth per sector is extracted and applied to estimate the quantities of waste in 2030 (see Table A.11). For the packaging, automotive, and electronic sector, the projections are based on the historical packaging waste, ELV, and WEEE generation based on Eurostat (2021; 2022a; 2022b). Regarding the projections of waste quantities for building & construction and agriculture sectors, historical waste quantity data by NACE activity (NACE F: Construction and NACE A: Agriculture, forestry, and fishing, respectively) are extracted from Eurostat (2022c). Overall the annual growth rates for packaging, ELV, WEEE, CDW, and APW are 1.4–1.8%, 1.3–1.6%, 1.1–1.2%, 0.8–0.9%, and 1.0–1.1%, respectively. Detailed results of the projections and annual growth can be found in Appendix A–section 7.

2.2.4 Circularity indicators

The summary of the four circularity indicators can be found in Table 2.2 (Equation 2.1–2.8). In Figure 2.2, a conceptual diagram of life cycle of plastic is presented to show the calculation point of each indicator. The *end-of-life recycling rate* (*EoL-RR*) (measured in %) is calculated as the ratio between the total mass (in kt) of polymer and base chemicals ($\mu_{polymer}$+ $\mu_{base\ chemicals}$) that is produced from the plastic waste treatments over the waste generated ($\mu_{waste\ generation}$) (in kt) (UNEP, 2011; Perio et al., 2018). On the numerator, only polymer and base chemicals are considered as recycled products to conform to the definition of 'recycling' by the WFD (European Commission 2018a; European Commission, 2008), which excludes materials (such as fuel) for energy usage. The *plastic-to-plastic rate* (P2P), *plastic-to-chemicals rate* (P2C), and *plastic-to-fuels rate* (P2F) (measured in %) are described as the share of total mass (in kt) of plastic waste generated ($\mu_{waste\ generation}$) that is converted into polymers ($\mu_{polymer}$), base chemicals ($\mu_{base\ chemicals}$), and fuels (μ_{fuel}), respectively (Broeren et al., 2022; Arena and Ardolino, 2022). On the denominator, in S0–S5, only the reported plastic waste ($\mu_{reported\ waste}$) is considered and in S5 the reported plastic waste plus 'missing plastic' ($\mu_{reported\ waste}$ + $\mu_{missing\ plastic}$) is considered (see Figure 2.2). In all developed scenarios, the assumed legal waste export for recycling is not counted in the EoL-RR and P2P rate calculations.

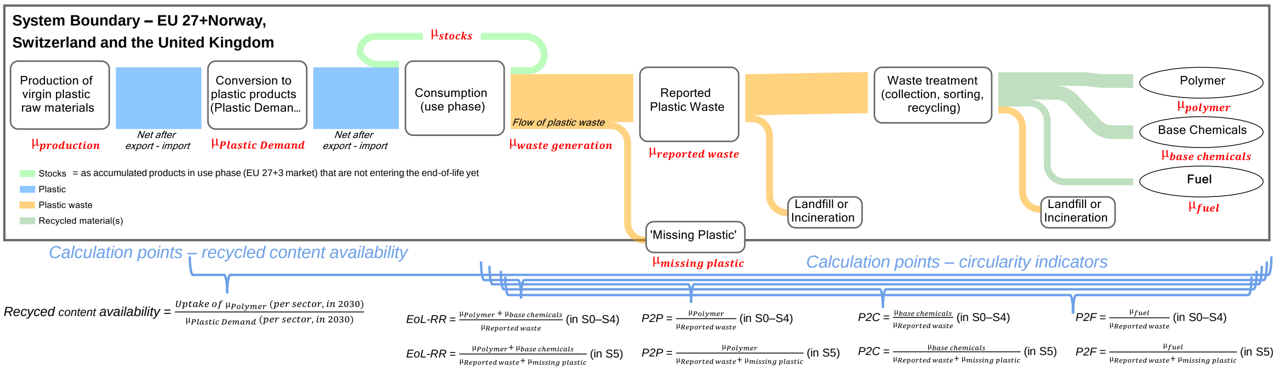


Figure 2.2 Conceptual diagram of life cycle of plastic (adapted from Plastics Europe) (Plastics Europe, 2019a; 2019b) to show the calculation points for each circularity indicator and recycled content availability. The thickness of the flows does not represent mass/quantity. Abbreviation: EoL-RR (End-of-life recycling rate), P2C (plastic-to-chemicals), P2F (plastic-to-fuel), P2P (plastic-to-plastic).

Table 2.2 Summary of circularity indicators of plastic waste treatment, their corresponding definitions and formulas applied in this study, which are also elaborated in previous studies (UNEP, 2011; Perio et al., 2018; Broeren et al., 2022; Arena and Ardolino, 2022).

Circularity indicators	Definition	Equation
End-of-life recycling rates (EoL-RR)	The total mass of plastic waste that is converted into secondary materials (polymers and base chemicals) over total plastic waste generation (i.e., reported plastic waste (in S0–S4) plus the 'missing plastic' (in S5) under the definition of '[1]recycling' from the European Commission (2018a; 2008), excluding plastic waste-to-energy (e.g., hydrocarbons)	$EoL - RR = \frac{\mu_{polymer} + \mu_{base\ chemicals}}{\mu_{reported\ waste}}$ (in S0–S4) (Equation 2.1) $EoL - RR = \frac{\mu_{polymer} + \mu_{base\ chemicals}}{\mu_{reported\ waste} + \mu_{missing\ plastic}}$ (in S5) (Equation 2.2)
Plastic-to-plastic rate (P2P)	The total of plastic waste that is converted into new polymer over the total plastic waste generation	$P2P = \frac{\mu_{polymer}}{\mu_{reported\ waste}}$ (in S0–S4) (Equation 2.3) $P2P = \frac{\mu_{polymer}}{\mu_{reported\ waste} + \mu_{missing\ plastic}}$ (in S5) (Equation 2.4)
Plastic-to-chemicals rate (P2C)	The total of plastic waste that is converted into base chemicals over the total plastic waste generation	$P2C = \frac{\mu_{base\ chemicals}}{\mu_{reported\ waste}}$ (in S0–S4) (Equation 2.5) $P2C = \frac{\mu_{base\ chemicals}}{\mu_{reported\ waste} + \mu_{missing\ plastic}}$ (in S5) (Equation 2.6)
Plastic-to-fuels rate (P2F)	The total of plastic waste that is converted into fuels for energy use over the total plastic waste generation	$P2F = \frac{\mu_{fuel}}{\mu_{reported\ waste}}$ (in S0 – S4) (Equation 2.7) $P2F = \frac{\mu_{fuel}}{\mu_{reported\ waste} + \mu_{missing\ plastic}}$ (in S5) (Equation 2.8)

[1]The definition of 'recycling' as stated in European Commission (2018a; 2008) reports are '*any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations*'. Hence, it (mainly) includes plastic waste recycling back into plastic from mechanical recycling in 2018. When chemical recycling is implemented in 2030, 'recycling' can include plastic waste recycling back into plastic or other materials for other purposes (e.g., base chemicals from pyrolysis for petrochemical industry such as cosmetics, fertilizers, pharmaceutical, etc.), excluding fuel or energy use.

2.2.5 Uncertainty analysis

Uncertainty analysis is carried out to quantify the uncertainty around the model results to capture the potential combined effects of variation of modelling parameters variability or the effects of modelling assumptions (Claverul et al., 2012). The uncertainty analysis is calculated because of the diversity of the modeling inputs that are taken from relevant literature related to the waste management practices in EU 27+3 (Table A1–A5 in Appendix A), rather than a single point estimate. Hence, the selected parameters are subjected to variability that can influence the model results. The uncertainty analysis is calculated assuming TD of the input parameters (i.e., the TCs). The uncertainty propagation method as suggested by Bisinella et al. (2016) is used in this research. For this, the Monte Carlo analysis with 1,000 iterations using RiskAMP add-in of Microsoft Excel® is used to randomly sample a value within each uncertainty distribution and calculate the standard deviation, which is shown relative to the likely value in %. For example, if the MFA result shows 4,090 kt of polymer production with the uncertainty of ± 353 kt, the result is presented as 4,090 kt ± 9% (i.e., 353 kt / 4,090 kt × 100%). For the circularity indicators, if the EoL-RR is estimated to be 24% with ± 2% uncertainty, it means that the likely values (i.e., 24%) can be deviated to 22% (min.) and 26% (max.). This approach is consistently applied throughout the MFA modelling in this study.

2.2.6 Estimation of recycled content availability in 2030

From the MFA model results, the potential use of recycled plastic in different markets and applications is investigated. However, it is challenging to project future market uptake of recycled plastic production because of i) different quality of recycled plastic, ii) a breadth of technical requirements of various applications, and ii) market saturation of some applications (Demets et al., 2020; Huysveld et al., 2022; Tonini et al., 2022). In this study, two assumptions are considered to quantify the potential recycled plastic (in kt) and recycled content (in %) availability in 2030. First, projecting the share of market uptake of recycled plastics in 2018 reported by Watkins et al. (2020) and European Commission (2020b) (details in Appendix A, Table A.12). Second, assuming 100% closed-loop recycling, i.e., no mass exchange between the sectors. The closed-loop recycling itself is defined as the use of recycled materials for the same market applications as that of its previous life cycle (UNEP, 2011), e.g., recycled plastics from packaging waste is used in the same sector. This is perceived as 'explorative' projection, in a sense that it does not take into account for example quality aspects yet (e.g., technical properties, processability, color, etc.) because of the difficulty to predict future market uptake (incl. potential market share of the intended applications) and the quality of recycling (Huysveld et al., 2022; Tonini et al., 2022; Hestin et al., 2017) at the time of writing the manuscript. Thus, it should be seen as a maximum uptake under optimal conditions and it is likely that the actual uptake will be lower.

The recycled content (in %) is quantified as the share of the uptake of recycled plastic over the projected plastic demand per sector in 2030 (Equation 9). For this purpose, the plastic demand in 2030 is projected using linear regression from Plastics Europe (data from 2014 to 2020) (Plastics Europe, 2019a; 2019b) and is elaborated in Appendix A, Figure A.28. It is important to note that the amount of plastic flowing from use phase to EoL phase ($\mu_{\text{waste generation}}$) in 2030 is not the same as the plastic demand in 2030 ($\mu_{\text{plastic demand}}$) because some plastic will remain in 'stock' (μ_{stocks}) depending on their lifespan distribution (Figure 2.2), as described by Lase et al. (2021).

$$Recycled\ content\ availability = \frac{Uptake\ of\ \mu_{polymer}\ (per\ sector,\ \ in\ 2030)}{\mu_{Plastic\ Demand}\ (per\ sector,\ \ in\ 2030)} \times 100\% \quad \text{(Equation 2.9)}$$

2.3 RESULTS AND DISCUSSIONS

2.3.1 Material flow analysis: *status quo* (in 2018) and future scenarios (in 2030)

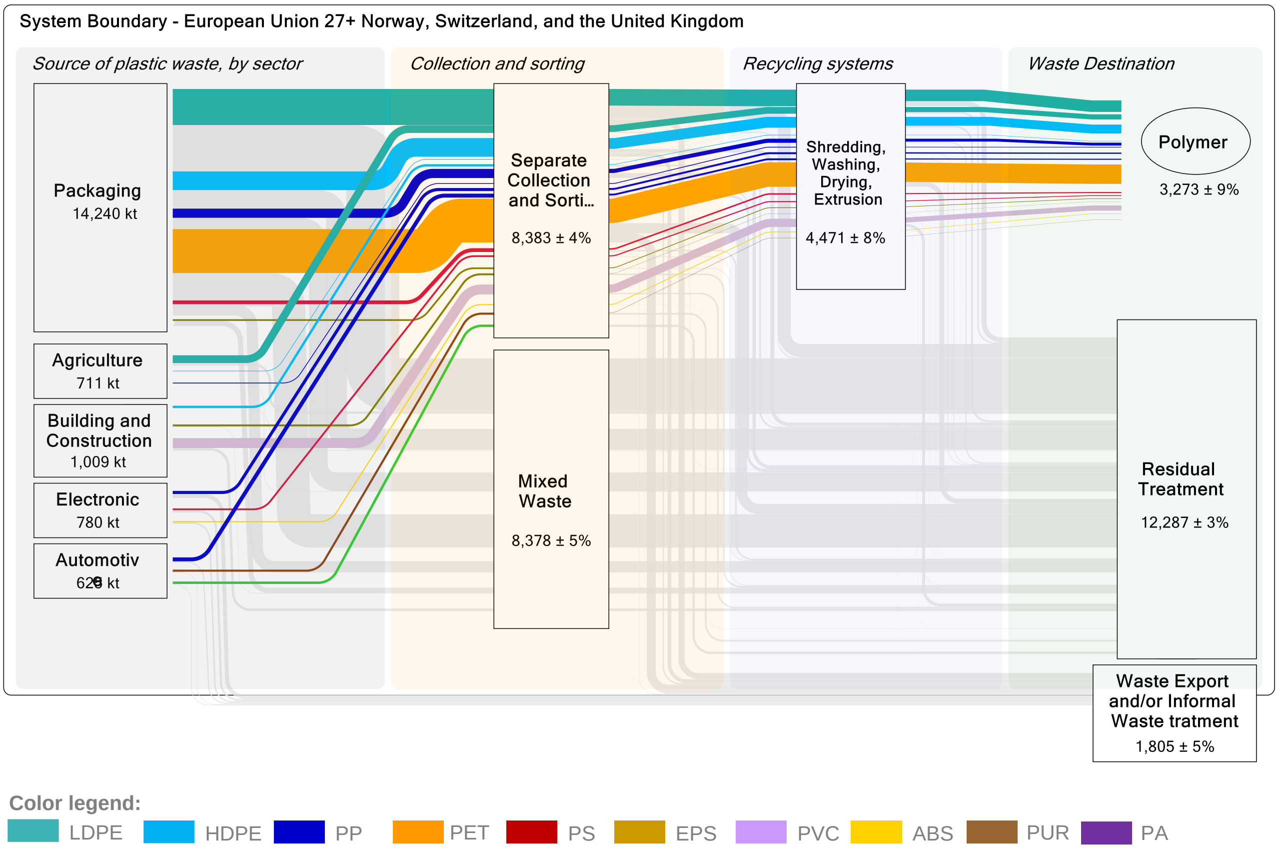
The MFA results of the ten polymers over the different sectors are shown in Figures 2.3A–3F for S0 – S5, respectively. Per sector, the Sankey diagrams can be found in Appendix A, Figures A.29-A.53, and the mass balances can be found in Table A.14.

In the *status quo* scenario (S0), it is estimated that 3,273 ± 9% kt of polymer (i.e., recycled plastic) is produced from plastic recycling systems in 2018, while 12,287 ± 3% kt plastic waste are sent for residual treatment and 1,805 ± 5% kt are sent to waste export and/or informal treatment (e.g., illegal export or unauthorized recycling by brokers or scraps dealers) (Figure 2.3A). According to the figures from Plastics Europe (Plastics Europe, 2019a; 2019b), it is estimated that 37% and 63% of the waste sent for residual treatment is landfilled and incinerated, respectively.

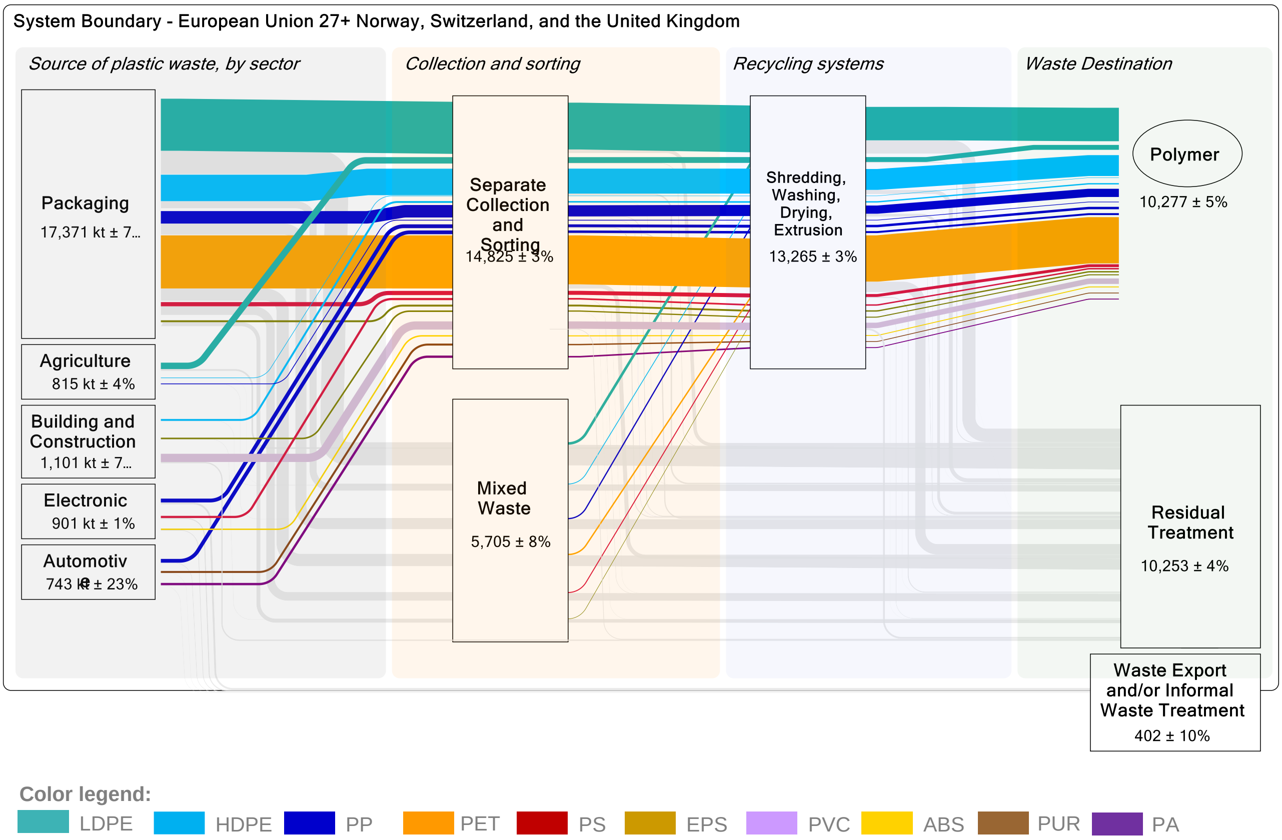
The results obtained for S1, assuming best practices of waste collection, sorting, and MR are widely applied in the whole EU 27+3, show that the recycled plastic production is expected to increase to 10,277 ± 5% kt (Figure 2.3B), which is approx. 3.0 times higher than what is estimated for S0. Still, a considerable amount of plastic waste is sent to residual treatment (i.e., 10,253 ± 5% kt) and a small share is still sent to waste export and/or informal treatment (402 ± 10% kt). From the results obtained for S2, which is the scenario in which CR and SBR have become dominant (i.e., CR and SBR technologically outcompetes MR), it can be observed that 7,903 ± 6% kt of recycled plastic are produced together with 7,247 ± 6% kt of base chemicals and 1,189 ± 4% kt of fuel (Figure 2.3C). The recycled plastic production in S2 (i.e., 7,903 ± 6%) is approx. 2.5 times higher than in S0. However, the recycled plastic production in S2 is slightly lower than in S1 (i.e., a reduction of approx. 23% of recycled plastic produced compared to S1), yet considerable amounts of base chemicals and fuels are produced in S2 as opposed to S1. Important to note is that these numbers are based on the Transfer Coefficients (TCs) retrieved from current data sources (see Table S7-S11), and, while potential improvements in the technology might still occur, it is nevertheless difficult to quantify technological learnings without an established history (as assumed for MR, sorting or collection rates).

In Figure 2.3D and 2.3E, the MFA results display that 12,262 ± 6% kt and 12,740 ± 6% of recycled plastic are estimated to be produced from S3 (the scenario in which little competition between CR, SBR, and MR occurs) and S4 (the scenario in which CR serves as complementary technologies to MR), respectively. This shows that the implementation of MR, CR and SBR in treating plastic waste delivers a higher quantity of recycled plastic compared to S1 (which considers only improved MR) and S2 (which considers only CR and SBR). In particular, the MFA results for S4 estimate that CR produces 3,110 ± 6% kt recycled plastic. This finding aligns with SYSTEMIQ (2022) and Caro et al. (2023) that estimate around 3,100 and 3,400 kt P2P production from CR and SBR, respectively. On the other hand, in S3 and S4, the amount of base chemicals production from plastic waste treatment is estimated to be 4,272 ± 6% kt and 3,951 ± 6% kt (i.e., a reduction of 41% and 45% compared to S2, respectively), while the fuel production from the same scenarios is estimated to be 683 ± 4% kt and 628 ± 4% kt, respectively. For S5 (the scenario in which extra mass from the 'missing plastic' is treated via CR and MR), the recycled plastic, base chemicals, and fuel production from plastic waste is estimated to be 18,536 ± 6% kt, 5,556 ± 6% kt, and 881 ± 4% kt, respectively (Figure 2.3F). The inclusion of the 'missing plastic' in future plastic waste recycling treatment, combined with complementarity between MR and CR, increases the total recycled plastic production by 5.5 times relative to S0. This would allow reaching the recycled content target (section 2.3.3).

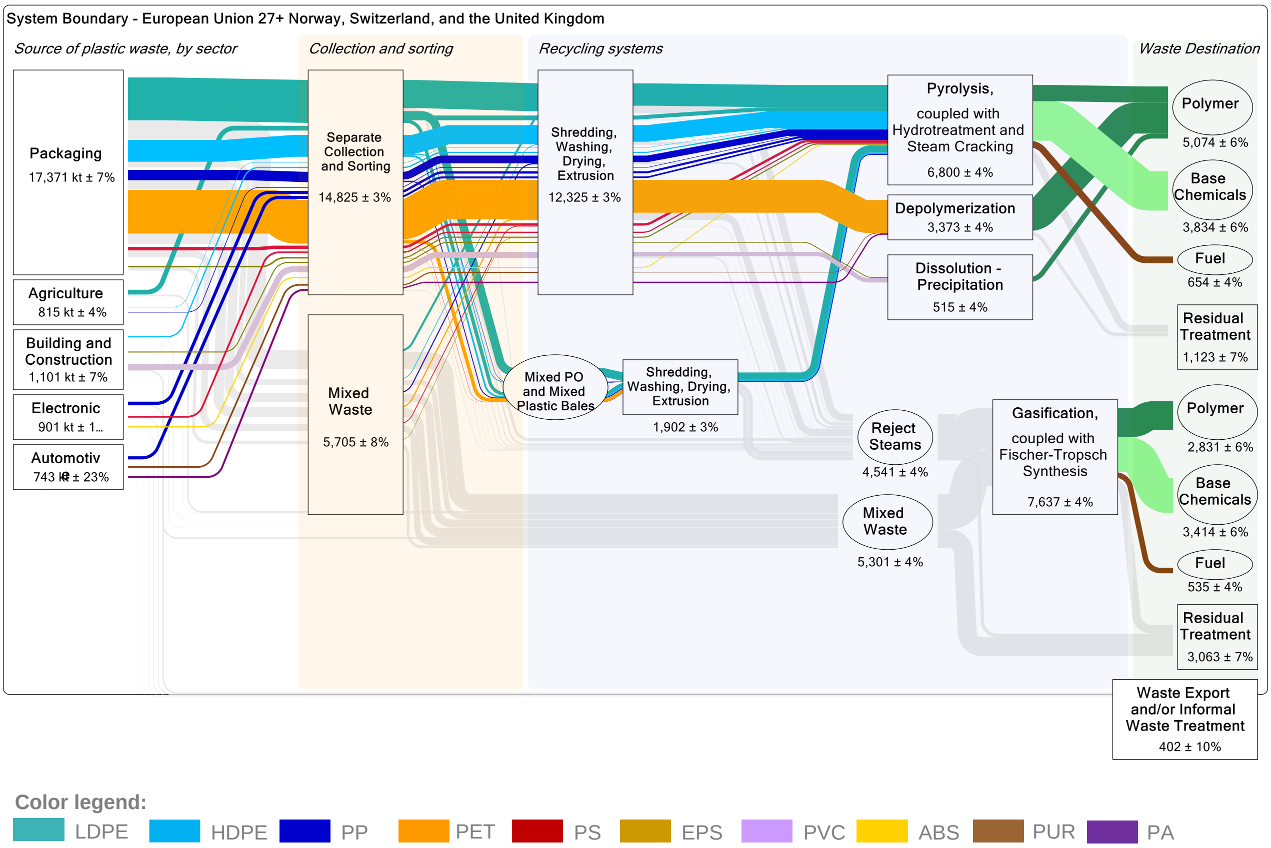
[**A**] Aggregated flows of plastic waste treatment in **S0**



[**B**] Aggregated flows of plastic waste treatment in **S1**



[**C**] Aggregated flows of plastic waste treatment in **S2**



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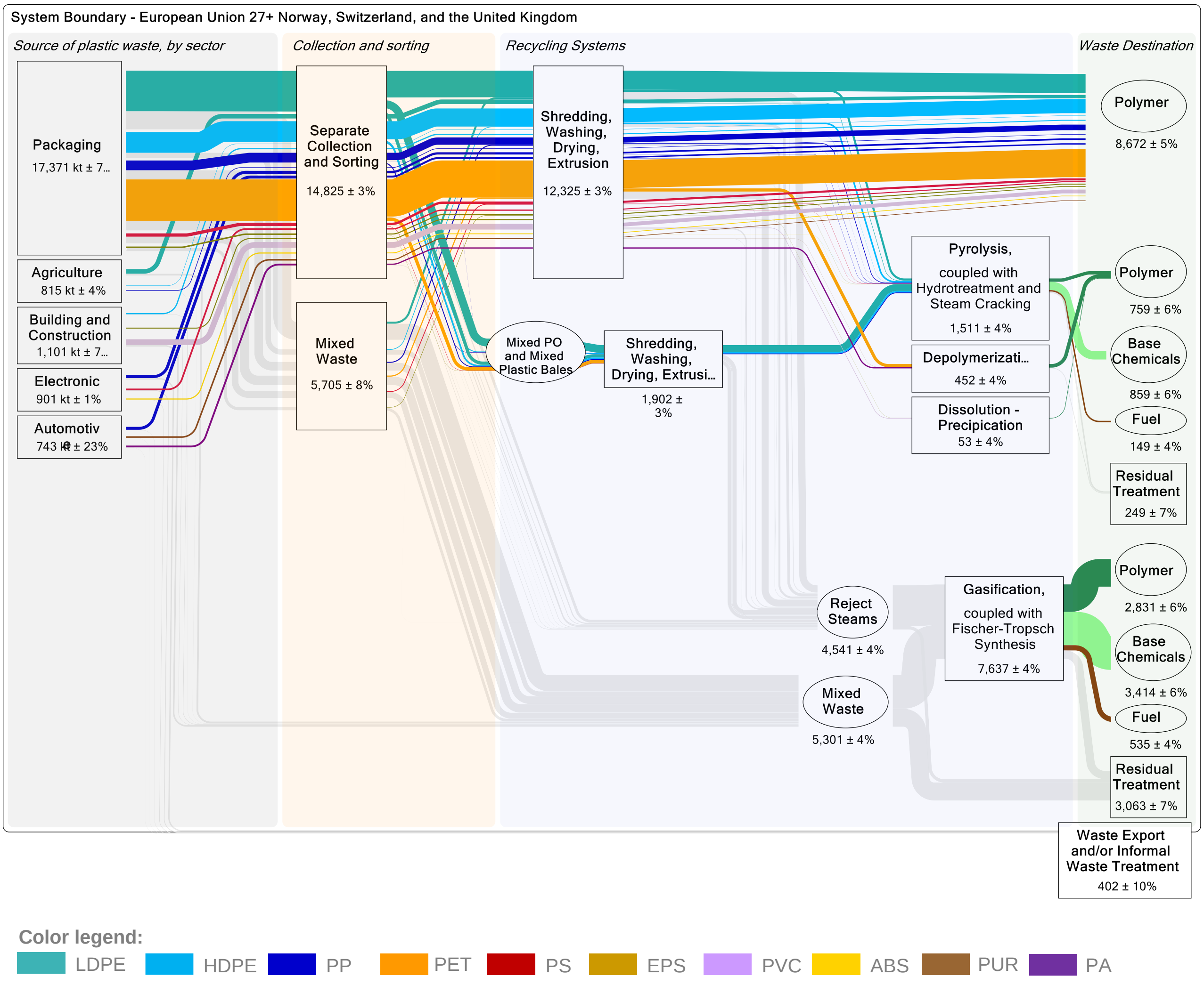
[**D**] Aggregated flows of plastic waste treatment in **S3**



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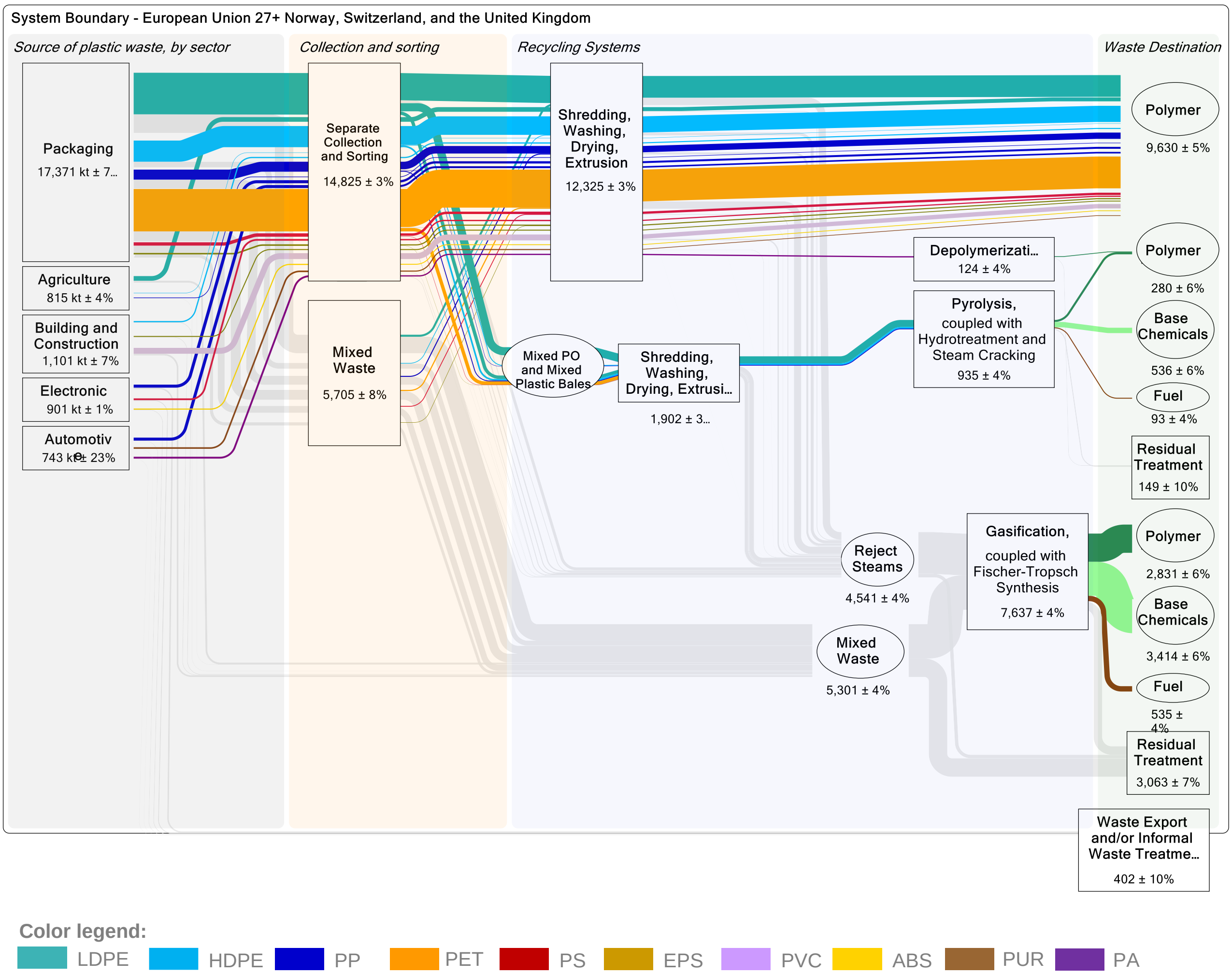
[**E**] Aggregated flows of plastic waste treatment in **S4**



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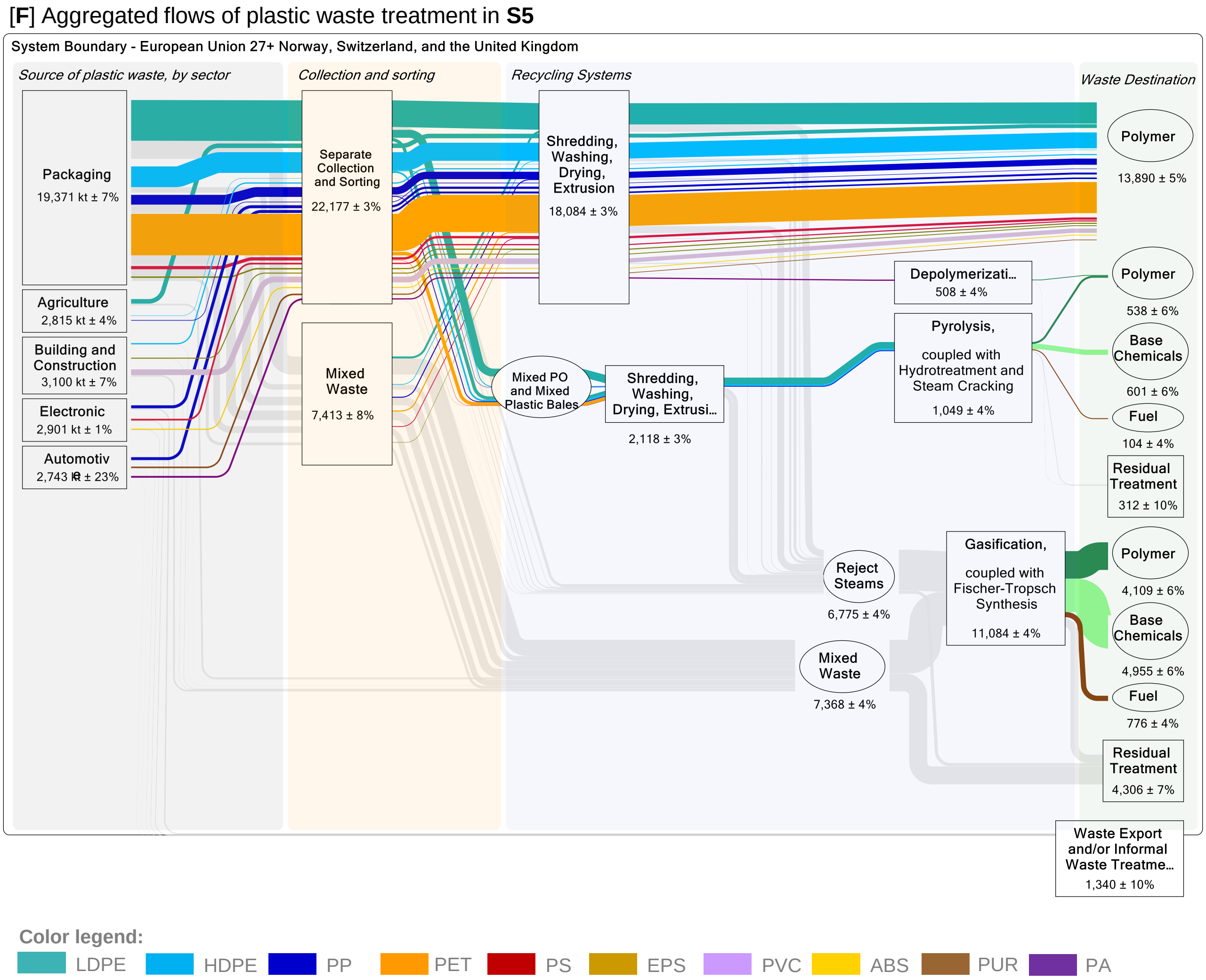


Figure 2.3 Material flow analysis of the selected polymers throughout the waste management systems in EU 27+3 in 2018 and 2030 of S0 (3A), S1 (3B), S2 (3C), S3 (3D), S4 (3E) and S5 (3F). Values are in rounded in kilotonne, including the calculated standard deviation (in %). Different colors represent different polymer types, while the grey color refers to waste export and/or informal waste treatment (e.g., illegal export or unauthorized recycling brokers/scraps dealers), mixed waste, reject (from sorting, washing, extrusion), and residual streams. The dark green, light green, and dark brown colors represent (mixed) polymer, base chemicals, and fuel productions from chemical recycling, respectively.

2.3.2 Circularity of plastic value chain: to what extent can chemical recycling contribute to improve plastic recycling EU?

Table 2.3 summarizes the results obtained for the circularity indicators (EoL-RR, P2P, P2C, and P2F rates) for scenarios S0 – S5. The EoL-RR of plastic waste in S0 is 18% ± 2% in which the only contributor is P2P from MR. Significant improvements in the EoL-RR can be observed in all future scenarios (S1 – S5). The overall EoL-RR in S1 (i.e., 49% ± 3%) is approx. 2.7 times higher than in S0 (i.e., 18% ± 2%). For the S2, the EoL-RR (i.e., 73% ± 3%) is approx. 4 times higher than in S0 and it is approx. 1.5 times higher than in S1. In S2, 38% ± 2% and 35% ± 2% of the overall EoL-RR come from P2P and P2C from CR and SBR (only P2P), respectively. However the P2P rate in S2 (38% ± 2%) is slightly lower than the P2P rate in S1 (49% ± 3%) (Table 2.3).

The results of circularity indicators for S3 and S4 estimate that the overall EoL-RR increases by approx. 4.5 times higher relative to S0 (Table 2.3). When comparing to S1, the EoL-RR of S3 and S4 (i.e., 78–80% ± 3%) is 1.6 times higher, while when comparing to S2, the EoL-RR of S3 and S4 is 1.1 times higher. It is estimated that the EoL-RR in S3 and S4 comes from P2P from MR (41–46% ± 3%), P2P from CR and SBR (15–17% ± 1%), and P2C from CR (19–20% ± 1%). When the 'missing plastic' is included in S5, the EoL-RR is identical to S4 however S5 produces more recycled plastics, base chemicals, and fuels as elaborated in the previous section (section 2.3.1). It is important to note that in this study the EoL-RR is calculated as the share of total recycled plastic (and base chemicals) over waste generated (incl. 'missing plastic' in S5) hence resulting the same EoL-RR in S4 and S5, but with different quantities involved. Furthermore, in all scenarios where CR options are implemented, the P2F rate is relatively low ranging from 3% (S3–S5) to 6% (S2).

In Appendix A–Table A.15 the plastic circularity per sector is presented. In *status quo* scenario (S0) the highest EoL-RR is achieved by agriculture sector (44% ± 5%), followed by building and construction (30% ± 2%), packaging (17% ± 2%), electronic (17% ± 2%), and automotive sectors (10% ± 1%), which is comparable with Maury et al. (2022) study. In all future scenarios (S1–S5), the EoL-RR will increase. In the most positive future scenarios (S4 and S5) the highest EoL-RR is achieved by building and construction and agriculture sectors (84% ± 5%), followed by packaging sector (81% ± 3%), automotive (72% ± 4%), and electronic sectors (60% ± 3%). CR and SBR implementation contribute in increased EoL-RR in future scenarios by adding around 15–18% ± 2% of P2C and 8–25% ± 2% of P2P from CR and SBR, while P2P from MR contributes 29–56% ± 3% (Table S16). Relative to S0, improved MR, CR and SBR implementation can increase the EoL-RR by roughly 2–5 times.

When focusing only on the P2P rate, the highest improvements can be observed in S4 and S5 (61% ± 3%). Moreover, the results suggest that the smallest improvement in P2P rate (compared to the S0 *status quo*, 18% ± 2% P2P rate) is observed for S2 (i.e., 38% ± 2%) where all the sorted plastic waste, including the rejects (from sorting and MR) and mixed waste streams, are sent to CR options. The result obtained for S2 (CR and SBR become more dominant than MR) is lower than what is obtained for S1 (i.e., 49% ± 3%) where all sorted plastic waste is treated via MR (and CR and SBR are assumed zero).

Table 2.3 Summary of the circularity indicators for all scenarios in 2018 (S0) and 2030 (S1–S5). Values are rounded, including the standard deviation (in %). Acronyms: CR (chemical recycling), EoL-RR (end-of-life recycling rate), MR (mechanical recycling), P2C (plastic-to-chemical), P2F (plastic-to-fuel), P2P (plastic-to-plastic), SBR (solvent-based recycling).

[1]Overall Circularity Indicators	S0	S1	S2	S3	S4	S5
P2P	18% ± 2%	49% ± 3%	38% ± 2%	58% ± 3%	61% ± 3%	61% ± 3%
P2P from MR	*18% ± 2%*	*49% ± 3%*	-	*41% ± 3%*	*46% ± 3%*	*46% ± 3%*
P2P from CR and SBR	-	-	*38% ± 2%*	*17% ± 1%*	*15% ± 1%*	*15% ± 1%*
P2C	-	-	35% ± 2%	20% ± 1%	19% ± 1%	19% ± 1%
P2F	-	-	6% ± 0%	3% ± 0%	3% ± 0%	3% ± 0%
[2]**EoL-RR**	**18% ± 2%**	**49% ± 3%**	**73% ± 3%**	**78% ± 3%**	**80% ± 3%**	**80% ± 3%**

[1] The 'overall' data points quantify the sum of mass quantities (in kt) from all sectors and aggregated calculations for the circularity indicators.
[2] EoL-RR considers only P2P and P2C because P2F recycling does not conform to the definition of 'recycling' in WFD (European Commission, 2018a; European Commission, 2008).

2.3.3 Recycled content availability in 2030

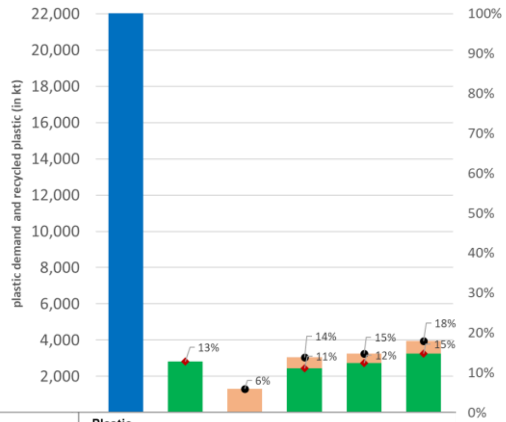
Figure 2.4 and Figure 2.5 summarize the potential recycled plastic (i.e., only P2P from MR, CR, and SBR) and recycled content (RC) availabilities in 2030 per sector. Figure 2.4 shows the results assuming that the share of recycled plastic uptake between sectors in 2018 is maintained in 2030 (i.e., closed-loop and open-loop recycling occur). For example, recycled plastic from plastic waste in packaging sector can be used for applications in packaging and other sectors too. In contrast, Figure 2.5 shows the results assuming 100% closed-loop recycling (i.e., the recycled plastic is used in the same sector where the waste is originated).

For example in Figure 2.4A, it is estimated that the packaging sector will demand 22,057 kt of plastic (blue bar) in 2030 and it is estimated that 2,817 kt of recycled plastic (green bar, in S1) will be produced, result in 13% RC availability (red dot, S1 in Figure 2.4A). In Figure 2.5A, assuming 100% closed-loop recycling based on best practices in MR, it is estimated that 8,528 kt of recycled plastic could be produced in S1 and will thus result in a potential RC availability of 39% in packaging sector. In Figure 2.4, the ‘others’ sector refer to household goods, furniture, textiles, sports equipment, etc.

From Figure 2.4 it can be observed for all future scenarios (S1–S5) a considerable amount of recycled plastic is diverted into 'others’ sector (3,670–7,141 kt) and make the RC availability for ‘others’ sector the highest among other sectors (from 39% in S2 to 77% in S5). The building and construction sector comes second as the biggest recycled plastic receiver (from 1,986 kt in S2 to 5,229 kt in S5), followed by the packaging sector (from 1,318 kt in S2 to 3,960 kt in S5). These result to RC availability ranges from 17% in S2 to 46% in S5 for building and construction sector, while the RC availability in packaging sector ranges from 6% in S2 to 18% in S5. The recycled plastic availabilities in the automotive (from 656 kt in S2 to 857 kt in S5), electronic (from 170 kt in S2 to 415 kt in S5), and agriculture sectors (from 107 kt in S2 to 936 kt in S5) are considerably lower than the packaging, construction and ‘others’ sector. These result in a considerably lower RC availability in automotive sector (6–15%) and electronic sector (3–11%). Again, note that all these results assume that the share of recycled plastic uptake between sectors in 2018 will be maintained in 2030.

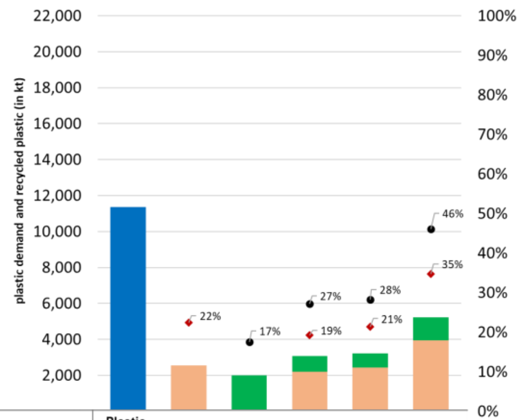
When 100% closed-loop recycling is assumed (Figure 2.5), the packaging sector is expected to receive the highest amount of recycled plastics (i.e., 7,698–11,430 kt), as expected, followed by the building and construction sector (i.e., 619–2,107 kt) since most of the recycled plastics are produced from these sectors (see Appendix A, Table A.14). The highest increase of RC availability can be observed in the packaging and electronic sectors (i.e., 3.5 times higher than assumed 2018 market uptake) and the lowest increase in the automotive sector (i.e., 1.5 times higher than assumed 2018 market uptake). As results, the RC availability increases to 35–52% in packaging sector and 5–38% in electronic sector. Still, in the automotive sector the RC availability increases to 5–26% (Figure 2.5). Instead, as expected, the potential RC availability in the construction sector is 2.5 lower than if current trends are maintained (open- and closed-loop, Figure 2.4), which reduces the RC availability to 5–19%.

[A] Packaging sector



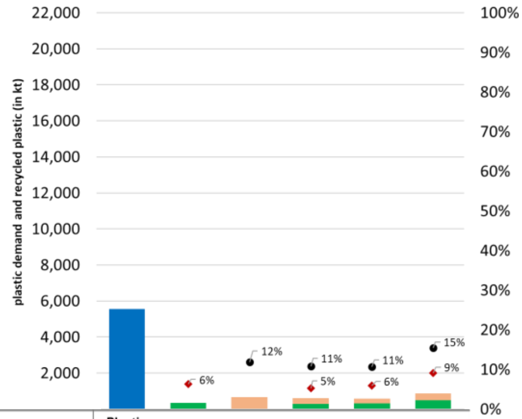
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	22,057	-	-	-	-	-
Recycled plastic from MR	-	2,817	-	2,453	2,728	3,258
Recycled plastic from CR	-	-	1,318	602	522	702

[B] Building and construction sector



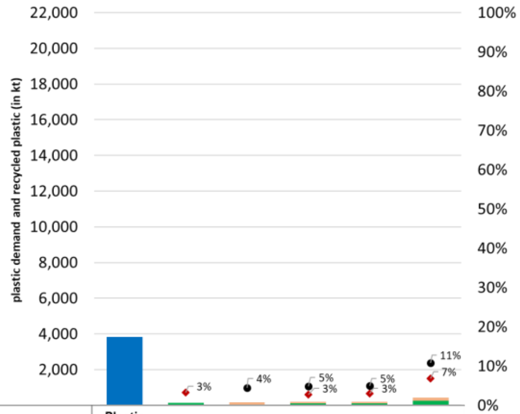
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	11,359	-	-	-	-	-
Recycled plastic from MR	-	2,546	-	2,191	2,434	3,942
Recycled plastic from CR	-	-	1,986	893	771	1,287

[C] Automotive sector



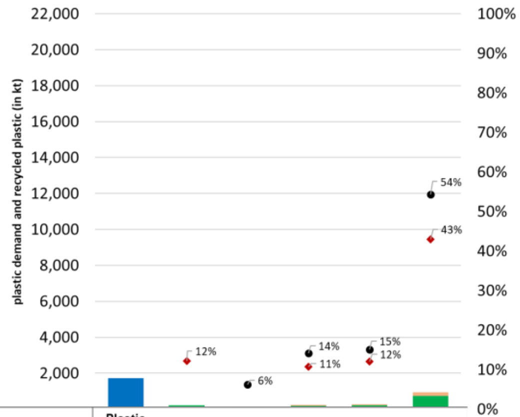
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	5,543	-	-	-	-	-
Recycled plastic from MR	-	350	-	296	326	503
Recycled plastic from CR	-	-	656	305	266	354

[D] Electronic sector



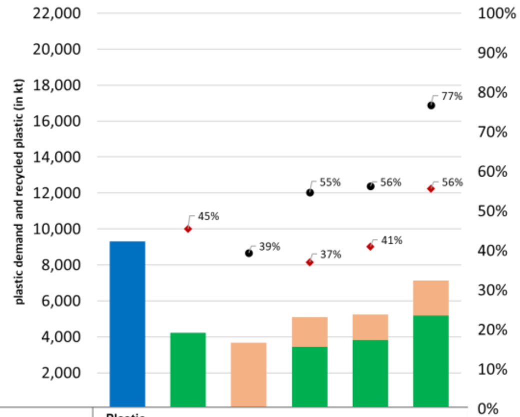
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	3,843	-	-	-	-	-
Recycled plastic from MR	-	126	-	104	115	264
Recycled plastic from CR	-	-	170	85	75	151

[E] Agriculture sector



Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	1,725	-	-	-	-	-
Recycled plastic from MR	-	210	-	185	207	742
Recycled plastic from CR	-	-	107	58	52	194

[F] 'Others' sector

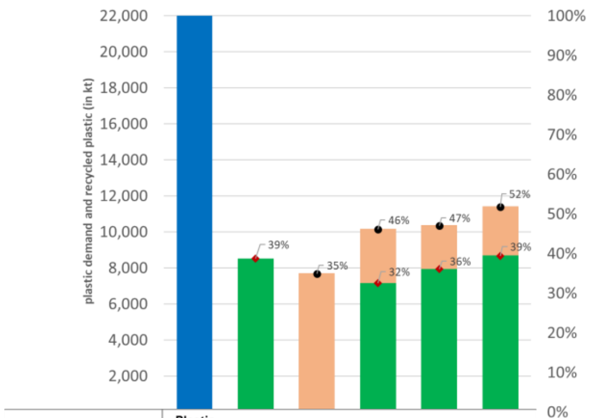


Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	9,308	-	-	-	-	-
Recycled plastic from MR	-	4,228	-	3,443	3,819	5,182
Recycled plastic from CR	-	-	3,670	1,647	1,421	1,959

Figure 2.4 Projected plastic demand (blue bar, in kt), recycled plastics production from mechanical recycling (green bar, in kt) and chemical recycling (orange bar, in kt), and recycled

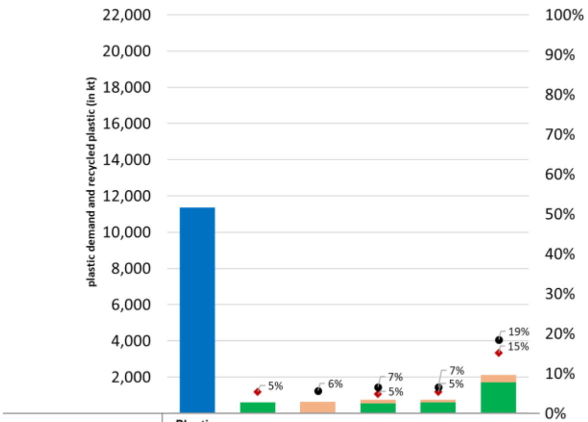
content (the dots, in %) assuming recycled plastic market uptake in 2018 based on Watkins et al. (2020) and European Commission (2020b). The red dot represents the potential recycled content achieved via mechanical recycling. The black dot represent potential recycled content achieved via chemical recycling or the sum of mechanical recycling and chemical recycling.

[**A**] Packaging sector



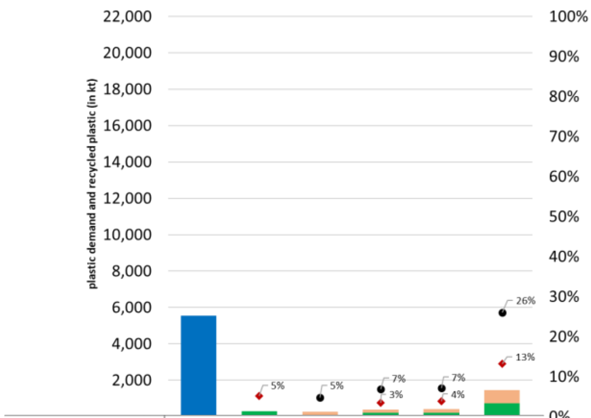
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	22,057	-	-	-	-	-
Recycled plastic from MR	-	8,528	-	3,015	2,413	2,737
Recycled plastic from CR	-	-	7,698	7,167	7,951	8,693

[**B**] Building and construction sector



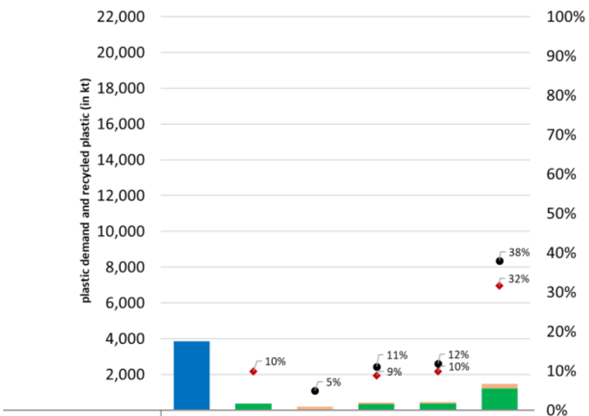
Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	11,359	-	-	-	-	-
Recycled plastic from MR	-	619	-	556	619	1,721
Recycled plastic from CR	-	-	647	190	137	386

[**C**] Automotive sector

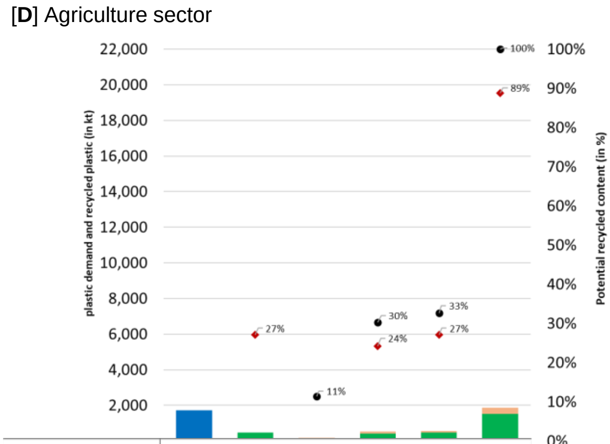


Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	5,543	-	-	-	-	-
Recycled plastic from MR	-	285	-	193	215	712
Recycled plastic from CR	-	-	267	191	184	732

[**D**] Electronic sector



Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	3,843	-	-	-	-	-
Recycled plastic from MR	-	377	-	338	377	1,214
Recycled plastic from CR	-	-	194	89	77	248



Mass Balances (in kt)	Plastic Demand	S1	S2	S3	S4	S5
Plastic demand	1,725	-	-	-	-	-
Recycled plastic from MR	-	468	-	418	468	1,530
Recycled plastic from CR	-	-	196	105	94	345

Figure 2.5 Projected plastic demand (blue bar, in kt), recycled plastics production from mechanical recycling (green bar, in kt) and chemical recycling (orange bar, in kt), and recycled content (the dots, in %) assuming 100% closed-loop recycling. The red dot represents the potential recycled content achieved via mechanical recycling. The black dot represent potential recycled content achieved via chemical recycling or the sum of mechanical recycling and chemical recycling

2.3.4 Interpretation and contextualization of the model results in relation to other studies

The MFA results of the *status quo* scenario (S0, 3,273 ± 9% kt) obtained in this study is comparable to the findings of Watkins et al. (2020) (i.e., 3,854 kt of recycled plastic in 2018). Plastics Europe (2020) instead estimated a higher amount of recycled plastic, i.e., around 4,900 kt in 2018, likely due to the higher number of polymer types considered. In S0 (*status quo* scenario), the total amount of waste export and/or informal waste treatments (1,805 ± 5% kt) is comparable to Plastics Europe (2020) and Eurostat (2022d) data, which indicate a total export of 1,900 kt and 1,593 kt in 2018, respectively. Several studies have shown that plastic waste can be legally or illegally exported to countries like Turkey, Malaysia, Vietnam, Ghana, Nigeria, Argentina, etc (Tran, 2018; Wang, 2014; Liang et al., 2021; Petrlik et al., 2019; Chen et al., 2021). Note that in this study the exported waste is neither counted as recycling (i.e., not contributing to the EoL-RR) nor considered in the calculation of recycled content availability in EU27+3. The exact fate of the currently shipped waste, evolution of export quantities, and final geographical and technical destination should be subjected to future research.

Looking at each future scenario in 2030 (S1–S5), including when the CR and SBR are implemented, it can be observed that the recycled plastic produced from only CR and SBR implementation (S2; 7,903±6% kt) is considerably lower than the other developed scenarios in this study (i.e., S1, S3, S4, S5). This also means that the highest recycled plastic production possible can be achieved only when MR, CR and SBR are implemented simultaneously, as can be observed in the other investigated future scenarios in 2030.

Focusing on the circularity indicators, it can be observed that modelling results of EoL-RR in 2018 (18% ± 2%) is lower than the one reported by Plastics Europe (2019a; 2019b) (i.e., 33%) but higher than the one reported by Material Economics (2022) and Agora Industry (2022), which is 15%. Important aspects to consider when comparing these numbers are i) recycling rate measurement points (as numerator in the EoL-RR formula) and ii) total considered EoL plastic waste (as denominator in the EoL-RR formula). The reported EoL-RR from Plastics Europe (2019a; 2019b) (i.e., 33% EoL-RR) calculates the share of sorted plastic waste that is sent to recycling facilities (9.6 Mt as the numerator) over the reported waste quantities (29.1 Mt as the denominator), which increases the EoL-RR. The reported EoL-RR by Plastics Europe (2019a; 2019b) further excludes the 'missing plastic' (i.e., 8–15 Mt) and, to a large extent, the losses from recycling, which inclusion will decrease further the EoL-RR. On the other hand, EoL-RR calculation by Material Economics (2022) and Agora Industry (2022) (i.e., 15% EoL-RR) consider the net recycled plastic production (6.7 Mt, after extrusion and losses from recycling) as the numerator and reported plastic waste generated plus the 'missing plastic' as the denominator (i.e., estimated to be 45 Mt). In this study the EoL-RR is calculated based on the share of total recycled plastic production (plus base chemicals when CR is implemented) over the total plastic waste generation (incl. the 'missing plastic' in S5) (see Table 2.2).

2.3.5 Potential contribution of chemical recycling to plastic recycling rates

Mass balance approach, as shown in this study, has been proposed to measure the contribution of CR and SBR (Broeren et al., 2022). For CR technologies, for example pyrolysis, the mass balance approach means accounting for the full process from breaking down the polymer chains into its basic building blocks (e.g., pyrolysis oil), purification steps (e.g., distillation and hydrotreatment), feeding into cracking process (e.g., steam crackers) to produce base chemicals (incl. olefins), and (re)polymerization (Tabrizi et al., 2021). Moreover, the mass balance approach can also be used as a tool by policy makers to monitor the yield of CR and SBR technologies and to formulate ambitious but realistic recycling targets (e.g., EoL-RR targets for plastic).

As CR options yield multiple products (i.e., monomers, chemicals, and hydrocarbon), it is important to clearly identify the potential quantities of each output to further distinguish recycled plastic production (i.e., plastic-to-plastic recycling) from the other outputs (e.g., plastic-to-fuel). This is also relevant to appropriately report the plastic recycling rate in Europe to monitoring the attainment of recycling targets, since the production of fuels (i.e., recycling products to be used as energy sources such as hydrocarbons) is not considered as 'recycling' under the Waste Framework Directive (WFD) (European Commission, 2018a; 2008). In this context, this study strives to differentiate between multiple outputs from CR (i.e., polymer, base chemicals, and fuel) in the endeavor to better illustrate the potential contributions of CR technologies to plastic circularity and recycling rates. Moreover, it is important to highlight that in all future scenarios in which CR options for plastic waste treatment are considered, the base chemicals production refers to producing valuable materials (e.g., wax, benzene, toluene, xylene, methane, propane, etc.) as suggested by previous studies (Civancik-Uslu, 2021; Kusenberg et al., 2022a; 2022b; 2022c; Ghalomi et al., 2021; Zayoud et al., 2022), which might be used as feedstock in the petrochemical industries. Moreover, the fuel is mainly hydrocarbons (gas and oil), which can also be used as energy input for CR processes (Civancik-Uslu, 2021; Larrain et al., 2020).

By observing the circularity indicators in all future scenarios (S1–S5), it can be noticed the most positive scenario with EoL-RR of 80% ± 3% is achieved when MR and CR are implemented simultaneously and complementarily, as opposed to only improving MR (S1, EoL-RR 49% ± 3%) or CR and SBR (S2, EoL-RR 73% ± 3%). The highest EoL-RR (80% ± 3%) is indeed achieved in S4 and S5. In these two scenarios, the EoL-RR reaches 80% ± 3% because of the contribution of improved MR (46% ± 3%) and complementary CR (34% ± 1%), where, out of the total obtained, 15% ± 1% is related to the P2P rate and 19% ± 1% to the P2C rate. These findings illustrate the importance of balancing the plastic waste streams into MR and CR options to reach the highest circularity potential possible, i.e. the two technologies need to be complementary and not competitive. It should also keep in mind that, in future scenarios, CR might be able to increase its P2P ratio (at the cost of P2C and P2F), for example by applying other pyrolysis conditions such as by adding catalysts, hydrocracking, etc.

(Kusenberg et al., 2022e, Kusenberg et al., 2022a). A study from Eschenbacher et al. (2022) suggests that the yield of olefins (i.e., C_2–C_4) from a mixed polyolefin waste can increase up to ~75% by introducing catalysts. Thus, for example, if the yield of naphtha can improve, the P2P rate can increase by up to ~65% (given the same yield from naphtha to monomers in the steam crackers).

By examining circularity indicators per sector, it can be observed that CR implementation contributes to reach recycling targets. In the packaging sector, the recycling targets stated by PPWD (i.e., 55% by 2030, European Commission, 2018a) cannot be achieved only by improving the current waste management treatments (i.e., collection, sorting, and MR) as the estimated EoL-RR in S1 is 49% ± 3%. The CR and SBR options will contribute to reach the recycling targets set by PPWD, as the EoL-RR is expected to increase to 73% ± 4%, 80% ± 3%, 81% ± 3%, and 81% ± 3% in S2, S3, S4, S5, respectively. The P2P and P2C from CR and SBR is estimated to add 15–38% ± 2% and 20–35% ± 2% to the EoL-RR in packaging sector (i.e., 73–81% ± 3%) in S2–S5, respectively. The contribution of CR and SBR options to increase the EoL-RR can also be noticed in the other sectors, e.g., significant improvements in the EoL-RR in the automotive sector are expected (from 38% ± 3% in S1 to 72% ± 4% in S4 and S5, Table A.15). Furthermore, the findings from this study (in Table A.16) can also be used as the basis to formulate recycling targets for plastic waste in the sector with no targets yet (e.g., in agriculture or construction sector).

In a similar way, the results can be used to perform plausibility-checks on stakeholders' pledges. For example, it can be observed that the recycled plastic produced from only CR and SBR of plastic (S2; 7,903 ± 6% kt) is not enough to meet the pledges made by CPA to reach 10,000 kt in 2030 (European Commission, 2022a). It is evident that such goal can only be achieved with an important contribution by MR, CR and SBR (as in scenario S3, S4, and S5)

2.3.6 Plausibility-checks on achievable recycled content targets

Mass balance approach has been proposed to monitor and determine recycled content of a product (Broeren et al., 2022; Tabrizi et al., 2021). For consumers, the mass balance approach means that brands and product manufacturers should ensure full transparency of the claimed recycled content (e.g., share of recycled plastic) of the total weight of a product. Policy makers can use mass balance approach to measure recycled content targets via, for example, transparent monitoring and certification systems (Tabrizi et al., 2021). Moreover, the presented MFA model (using mass balance principles) can support policy makers to formulate ambitious but realistic recycled content targets (for plastic-based items) by taking into account the quantities of recycled plastic produced annually. This MFA model can also be used to identify the bottlenecks towards meeting the targets.

Looking at the RC availability per sector, it can be observed that a significant amount of recycled plastic (i.e., 3,670–7,141 kt) might be used in the 'others' sector (i.e., household goods, textile, etc.) as open-loop recycling in 2030. In other words, if the current market uptake as of 2018 is maintained in 2030, the pledges or targets on RC in some sectors such as automotive, and electronic will not be achieved. For example, 25–30% RC target in electronic sector (Sandoval, 2018, Lase et al., 2021) will not be met as only 11% of RC would be available at the most positive scenario (i.e., CR is implemented and 'missing plastic' is accounted for, S5 in Figure 2.4D). Similarly, the RC targets in automotive sector (i.e., 20–25% in new passenger cars; Maury et al., 2022, Volvo, 2018) is not achieved as only 15% RC will be available in the most positive scenario (S5 in Figure 2.4C). For electronic and automotive sectors, the RC targets can only be achieved by the inclusion of CR options, processing the 'missing plastic', and closed-loop recycling (S5 in Figure 2.5C and 2.5D).

The findings on RC availability can be used to formulate targets for the each sector, e.g., packaging. It can be observed that around 35–52% of RC will be available for packaging sector in 2030, assuming closed-loop recycling (i.e., S1–S5 in Figure 2.3A). This finding aligns with 30% RC targets for PET beverage bottles stated by the Single Use Plastic Directive (European Commission, 2019) as well as study from Bashirgonbadi et al. (2022) that shows PP films can be made of 32 wt% recycled PP. Therefore, the findings of this research can also be used as basis to set RC targets for broader plastic packaging types such as flexible packaging, HDPE bottles, etc.

2.4 CONCLUSION

Current end-of-life recycling rate from 2018 data in Europe, based on mechanical recycling, is about 18% calculated from the amount of recycled plastic production over the reported plastic waste generation. The growth of plastic waste generation until 2030 is projected using historical data, while widespread implementation of production and use-oriented solutions such as waste reduction, re-use, re-fill, etc. are not yet considered in this study. In future, several scenarios can be deployed to improve the recycling rate. In first instance, stretching the possibilities of current commercially used mechanical recycling technologies can lead to an overall end-of-life recycling rate up to 49% in 2030. Results of this study show that the implementation of chemical and solvent-based recycling technologies bring positive impacts towards the end-of-life-recycling rate as plastic-to-plastic and plastic-to-chemicals recycling (from chemical recycling) will increase the rate up to 80%. In this most positive scenario (and potentially the most realistic one), chemical recycling becomes complementary (and not competitive) to improved mechanical recycling. In this scenario, plastic-to-plastic rate of 61% can be achieved (46% from mechanical recycling and 15% from chemical recycling), with an additional plastic-to-chemical rate of 19%. In all cases, plastic-to-fuel rates range from 3% to 6%, but it will likely be reduced in the future in favor for polymer and chemical production. Moreover, the findings from this research suggest that the recycled content targets are achieved when closed-loop, chemical and solvent-based recycling, and processing 'missing plastic' are all simultaneously accounted for in plastic waste treatment. Capturing and treating the 'missing plastic' can significantly increase the recycled plastic production and this contribution appears necessary to be able to reach the recycled content targets in some sectors (i.e., 25–30% recycled content targets in new electronic products in 2030). For policy makers, the approach (i.e., mass balance model) and findings of this paper can also be used to support proposals of realistically achievable recycled content targets and support which recycling technologies can play which role(s) in achieving the targets.

CHAPTER 3: MODELING CURRENT AND FUTURE FLOWS OF PLASTIC FROM WEEE RECYCLING

Redrafted from:

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Chapter 3

Modeling current and future flows of plastic from WEEE recycling

3.1 INTRODUCTION

The way we produce and use electrical and electronic equipment (EEE) nowadays has led to a considerable amounts of waste electrical and electronic equipment (WEEE) (Forti et al., 2020). The growth of WEEE is triggered by rapid technological advancements, falling prices, and shorter lifetime of EEEs (Tran et al., 2018; Wang et al., 2013). In 2019, the world generated 53.6 Mt of WEEE, of which the European market accounted for 22% (Forti et al., 2020). With the current production and consumption patterns of EEEs, it is expected that the quantities of WEEE will reach 115 Mt by 2050 (Parajuly et al., 2019; Baldé et al., 2017). Moreover, plastics have been widely used in EEEs and from the total collected post-consumer plastics waste in 2018, 6% was found in WEEE. However, the EEE industry only procures 2% of total recycled plastics (as regranulates) in the same evaluation year (PlasticsEurope, 2019b). The uptake of post-consumer regranulates in EEE thus still low and requires significant improvement.

Mechanical recycling remains the most ubiquitous WEEE management system in Europe (Parajuly et al., 2017; Ragaert et al., 2017). Recycling has also been perceived to be the solution to reduce the environmental impact of mining and primary material production (Vanegas et al., 2017). As a consequence, institutions such the European Commission have implemented the WEEE Directive in 2002, which was renewed in 2012 (Directive 2012/19/EU), with the aim to preserve, protect, and improve the quality of the resources. Particularly within plastics recycling, initiatives like the Circular Plastics Alliance (CPA) have declared their ambition to boost the market for regranulates up to 10 million ton by 2025 (European Commission, 2022b; 2018d; 2012). With legislation towards increased recycling of WEEE and also Plastics from WEEE – known as WEEP – pushing on the market, electronic producers have responded by pledging to use at least 25–30% of regranulates in EEEs by 2025 or 2030 (Philips, 2020; Electrolux Group, 2020; Whirlpool Corporation, 2018; Sandoval, 2018).

Stakeholders in the (W)EEE industry in Europe are mandated to establish an integrated WEEE collection, separation and recycling systems. This recycling chain should be able to deliver the secondary resources, including regranulates, to fulfil the pledges to use recycled material (i.e., recycled content targets). However, to date, we still have limited knowledge to predict the evolution in material compositions, stocks and flows of (W)EEE, which leads to an inefficient material recovery and recycling from WEEE (Thiébaud et al., 2017a; 2017b). Moreover, Thiébaud et al. (2017b) and Kawecki et al. (2018) have also highlighted that a better estimation of WEEE generation in the future becomes imperative to provide sufficient information about the future mass flows, required recycling capacities and technologies, and financial means to invest in recycling technologies.

Such estimations require high-quality data on product sales and lifespan distributions of EEE. The dynamic socio-economic conditions and the fact that EEE are often not disposed of immediately at the end-of-life, but rather end up in so-called 'stocks', cannot be easily modelled (Tran, 2018; Van Eygen et al., 2016; Wang et al., 2013; Thiébaud et al., 2017b). Current approaches to quantify WEEE generation, such as disposal related analysis, projection models, and factor models either have high uncertainty due to complex socio-economic situations or require huge amounts of data to run the model. These approaches are perceived to be inadequate in quantifying the flow of WEEE because the influence of socio-economic situations is not fully captured (Tran, 2018). Furthermore, one has to deal with the impact of technological advancement, changes in material composition, and potential legacy chemicals therein. Consequently, the current approaches to quantify WEEE fail to accurately assess if recycling targets or the establishment of sustainable end-markets for the uptake of regranulates are actually feasible. In this context, multivariate input-output analysis (MIOA) combined with material flow analysis (MFA) are perceived as suitable tools to answer the above mentioned questions because these approaches can better predict future WEEE generation (Wang et al., 2013; Wang, 2014). Additionally, these approaches may provide an outlook of the existing and future pathways of material valorization from WEEE, improvement potential on plastic recycling rate, and strategic planning towards better WEEE recycling processes (De Meester et al., 2019; Tran et al., 2018; Wang et al., 2013; Brunner and Rechberger, 2005).

A typical recycling chain consists of three main steps: separate collection (i.e., source separation), pre-processing, and end-processing (Vanegas et al., 2017; Parajuly et al., 2016;

Schluep et al., 2009). Collection plays a vital role in the overall recycling processes, as size and weight of equipment can be a determining factor for the effectiveness of WEEE collection. For example, small devices like mobile phones or electric shavers can easily be stored indoors for a relatively long period of time compared to larger sized electronic products (Wang, 2014). This type of information highlights the importance of understanding the lifespan distribution of EEEs, including their service lifetime and storage time when EEEs are hibernated in consumers' possession before being disposed of (Thiébaud et al., 2017b). After collection, pre-processing is crucial to determine the extent to which we can produce high-quality materials for recycling processes (Van Eygen et al., 2018; Parajuly et al., 2016). Furthermore, after a series of material size reduction, sorting and separation processes, the WEEP will be sent to the end-processing step that converts WEEP from flakes into granulates via extrusion (Cherrington & Makenji, 2019; Ragaert et al., 2017). Unfortunately, some materials, for instance WEEP, are more difficult to dismantle and separate at the very beginning of the recycling chain, which result in low overall recycling rates (Chancerel et al., 2009). As a result of the mentioned challenges, a significant amount of WEEP can still be found in the residual fraction and are thus destined for energy recovery by incineration or for heat production at cement kilns (De Meester et al., 2019; Jain & Sareen, 2006).

WEEP can also carry toxic elements or can be mixed with multiple polymer types that add to the complexity of WEEP recycling (Sahajwalla and Gaikwad, 2018; Parajuly et al., 2016; Habib et al., 2015). To overcome the problem of legacy chemicals present in WEEP, the Restriction of Hazardous Substances (RoHS) Directive was put into force in 2006, and continuously updated (RoHS Guide, 2020; European Commission, 2011). An example of hazardous substances, namely brominated flame retardants (BFRs), has been observed to cause an adverse impact on the recycling process, quality of regranulates, and human health at recycling facilities (Wagner and Schlummer, 2020; Delva et al., 2018; Sahajwalla and Gaikwad, 2018; Morf et al., 2005; Thuresson et al., 2005; Julander et al., 2005; Sjödin et al., 2001).

Given the current commitments to recycle WEEE and use regranulates, a method to assess the potential of the recycling chain to produce regranulates to be used as recycled content is needed. The assessment of the potential secondary resources in the future requires a better projection of future WEEE generation. Therefore, we combine MIOA and MFA in evaluating the current WEEP recycling processes and link this to the future amounts of expected WEEE generated and WEEP therein. The MIOA that is developed by Wang et al. (2013) is chosen because this approach considers the dynamic interconnection of product sales, stock, and lifespan. This modeling approach thus improves the prediction of future WEEE generation and its composition. The developed model can trace the fate of plastics and additives within the current and future formal WEEE management system and allows quantification of the annual regranulates production. This research consists of three parts. The first part aims to evaluate the current recycling rates of WEEP through MFA, starting when the consumers dispose their electronic products until the end of WEEP recycling process. Afterwards, sensitivity analysis is performed to identify the bottlenecks and potential scenarios to improve the WEEE recycling system. The second part of this research aims to estimate the amounts of WEEP that can be recycled by 2030. MIOA is used to predict the amount of WEEE generation based on historical product sales data and lifespan distribution models. Consequently, the result may reveal the resource potential of recycled plastics as well as the assessment of toxic elements from WEEE. The third part of this research aims to predict the future amount of recycled content, introducing scenarios with improvements in the WEEE recycling chain and changes in vacuum cleaners' material composition. As system boundary of this study, we focus on Belgium and The Netherlands and focus on three selected small household appliances (SHA) products being coffee machines, electric shavers, and vacuum cleaners.

3.2 MATERIALS AND METHODS

3.2.1 The considered EEEs: vacuum cleaners, coffee machines, and electric shavers

Three electronic products from the SHA category of EEE were selected for this research namely vacuum cleaners, coffee machines, and electric shavers waste because of their significant plastic concentration. Moreover, these three products have the highest plastics concentration among other EEEs and inherently different equipment in this category (European Committee of Domestic Equipment Manufacturers, 2017).

According to a study by Rames et al. (2019), cylinder vacuum cleaners are the prevalent type in Europe with a market share of 68% whilst upright vacuum cleaners account for 7%. On the other hand, the cordless and robotic vacuum cleaner (RVC) account for less than 10% of the current market in Europe. However, the cordless and RVC models differ from the cylinder vacuum cleaner in designs and material compositions. Notable differences in cordless and robotic models' material composition are the number of electronics, sensors, and batteries (Parajuly et al., 2016).

Related to coffee machines, Mudgal et al. (2011) estimated that the drip filter, pad coffee machines, and hard cap espresso coffee machines to have a market share of 38% and 22%, and 14% in 2020. The remaining portion of the market share of coffee machines in Europe belongs to the semi- and fully-automated coffee machines.

Between the electric shavers, the rotary shavers and foil shavers are the most used and popular models in the market. The significant difference between these models are the cutting systems (movement of the cutting blades) (Rietzler, M. et al., 2016, Löv & Fetene, 2012). Although the market data belongs to the European market condition, it is assumed that Belgium and The Netherlands have the same market condition in this research.

When determining the composition of plastics for selected EEEs, we choose the average plastic content of products with the highest market share in Europe. The cylinder and upright vacuum cleaners have the highest plastics content with 73% mixed plastics and 27% non-plastics such as copper, stainless steel, and aluminum. The mixed plastics composition is PP (34%), ABS (34%), Ethylene Vinyl Acetate (13%), PVC (11%), HDPE (8%), and Polyoxymethylene (1%) (Rames et al., 2019, Gallego-Schmid et al., 2016). For the coffee machine, drip filter coffee machines are made of 55% mixed plastics and 45% non-plastics. Of this 55% mixed plastics, 95% is PP and 6% is a combination of Polycarbonate (PC) and PVC. The remaining 45% of non-plastics are copper, glass, alloy steel, and aluminum (Mudgal et al., 2011). Lastly, the average plastic content of electric shavers is 51% and the remaining 49% of non-plastics content in the electric shaver. Especially for electronic shavers, the material composition remains similar regardless of the difference in models (Rietzler, M. et al., 2016; Löv and Fetene, 2012). These material compositions will be used in the material flow analysis to determine the fate of WEEP that enter the recycling facility and potentially recycled back into the economy.

3.2.2 Material flow analysis

3.2.2.1 Description of system boundaries and scenarios

This study targets the WEEP material flows within the vacuum cleaners, coffee machines, and electric shaver. Thus, other materials, such as metals and glass, are excluded when evaluating the performance of the current WEEE management system. The first part of this research focuses on evaluating the current WEEE management systems from disposal phase by the consumers until the end-of-life (EoL). The functional unit (FU) used in evaluating the current state of WEEE management systems in Belgium and The Netherlands corresponds to the treatment of one ton (1000 kg) of mixed WEEP and 200 kg of imported WEEE from vacuum cleaners, coffee machines, and electric shavers waste after being thrown away by the consumers. The second part of this research (more in section 3.2.3) focuses on forecasting the amount of regranulates production from WEEE recycling by combining MFA and MIOA. In forecasting the future regranulates production in 2030, the FU of 1 ton mixed WEEP is replaced by the amount of WEEP generated in 2030 [*WEEP(t,n)*] based on the MIOA (section 3.2.3).

3.2.2.2 Description of the system boundary

MFA refers to a systematic assessment of flows and stocks of every element, compounds, or goods within a defined system boundary (De Meester et al., 2019; Allesch, A. and Brunner P., 2017). In this research, the system boundary is limited to the processes and flows of the selected (W)EEE products and WEEP in Belgium and The Netherlands mainly because of the data availability (Figure 3.1). This WEEE management system is organized by different WEEE collectors and recyclers in Belgium and The Netherlands (Recupel, 2018; 2013, Huisman et al., 2012).

The lifecycle of plastics in EEEs begins at the production phase, where plastics are sourced from virgin materials and/or recycled products. Thereafter, EEEs are sold and placed on the market (POM) and EEEs begin to be used by the consumers (use phase). At this stage, EEEs are accumulated and disposed of depending on the individuals' behavior. The moment when consumers dispose of their EEEs marks the beginning of the WEEE disposal phase. From there on, there are two potential flows of WEEEs: (1) formal WEEE management and (2) informal WEEE management systems (Figure 3.1). The formal WEEE management systems are mostly driven by the introduction of national policies and environmental awareness, while informal systems are driven by the economic aspect of recyclable materials from WEEE (Wang, 2014). Both systems exist in Belgium and The Netherlands, and it is also the consumers' behaviour that mainly acts as the crucial determining factor of where WEEEs are going to be processed. WEEE may enter recycling facilities via several formal collection points. Once formally collected, WEEE will be sent for recycling, materials and components will be separated and further processed, and plastics can be recirculated back into the economy. On the other hand, WEEP will not go into such a system in the informal stream and plastics will not be necessarily recycled. The destinations for informally collected WEEE are incineration, landfill, or being exported outside our studied area. Some of the flows are even undocumented, hence called as unidentified stream in this research (Recupel, 2018; 2013; Huisman et al., 2012).

A detailed description of the processes, symbols, chosen values, and data sources used in Figure 3.1 is listed in Table 3.1. All collection, sorting and recycling efficiencies of these processes are indicated by η symbol and mass fractions are indicated by μ symbol. When consumers decide to dispose of their EEEs, waste can be *formally collected* (η_{Col}) via WEEE collectors, repair centers, or take-back schemes. The take-back scheme refers to authorized WEEE collectors in Belgium and The Netherlands, e.g., Recupel and Wecycle, respectively. In fact, the majority of WEEE (85%) is collected via take-back schemes. The remaining 15% of WEEE are collected from repair centers, and WEEE collectors, such as brokers or scraps dealers. Later, WEEE collectors report their collection data to the authorized WEEE collectors (Recupel, 2018; Huisman et al., 2012). At the repair centers, WEEE will be checked, and the repairable items will be sent for reuse purposes (μ_{RU}) (De Meester et al., 2019). However, in this research, we assumed that no products are reused because vacuum cleaners, coffee machines, and electric shavers are not typical products that get a second life. Moreover, it is also mentioned in many national WEEE reports that collectors or recyclers receive a fair amount of WEEE from other regions (National (W)EEE Register, 2019; Recupel, 2018; 2013; Baldé et al., 2017; 2016; European Environment Agency, 2012; Huisman et al., 2012; Lundgren, 2012). Therefore, an additional import stream is also introduced into the studied system.

Sorting and recycling then start from the quality check, clustering based on the WEEE category (in this case SHA), dismantling (depollution), shredding and material sorting by the recyclers, called *pre-processing* (η_{QCDS}) in this research. Prior to shredding and automated material sorting, trained labors dismantle copper wires, glass and wood housing, components with high (precious) metal content and batteries from many SHA on the moving conveyor belts, including for the selected EEEs. This process is crucial in the entire WEEE recycling chain because it determines the extent to which the materials are appropriately sent to the proper separation and end-processing step. It is also found that good practice of pre-processing treatment can upgrade valuable substances contained in WEEEs while eliminating non-valuable and hazardous substances (Van Eygen et al., 2016; Menad, 2016; Makenji and Savage, 2012; Chancerel et al., 2009). In the next step, the loose shredded mixed particles will go to *dry and wet separation* (η_S) and *compounding* (η_C). Typical examples of wet and dry separation technologies are near-infrared (NIR), sink-floats, magnetic and eddy current separations (Van Eygen et al., 2016; Menad, 2016; Makenji and Savage, 2012; Van Schaik and Reuter, 2012). Once the plastics material has been size reduced and sorted, it can be transformed into secondary materials using an extruder. Sometimes additives and virgin plastics are introduced in this process while certain contaminants can be removed with a melt filter (Makenji, K., and Savage, M., 2012; Ragaert et al., 2017). The effect of material degradation or decrease in quality of regranulates during EEE's lifecycle and recycling processes is neglected in this research. To our best knowledge this is a reasonable assumption for EEE plastics, especially at the lower (technical) limits of using regranulates (in EEE sector), which still allows for example to use certain colors on outer layers (Demori et al., 2015; Perez et al., 2010; Karahaliou & Tarantili, 2009; Brennan et al., 2002).

In contrary, the final destinations of the *informally collected* WEEE (η_{InfCol}) are *incineration* (μ_{INC}) or *export stream* (μ_{Ex}). Landfilling has been banned in Belgium and The Netherlands (Plastics Europe, 2018), but the possibility of miss sorting by the consumers still exist. According to Huisman et al. 2012, the portion of WEEE sent to landfill accounts for 9% of the total WEEE generated in 2011. Therefore, it is assumed that the same portion of landfilled WEEE is allocated to the incineration stream at the current market condition. Transboundary shipment is indispensable in global waste trading, thus an export stream is included in the quantification of WEEE mass flows. However, real-time data on the transboundary shipment of WEEE is often unavailable or low in quality (Tran et al., 2018; Baldé et al., 2017; Lundgren, 2012). Finally, some WEEE streams cannot be determined and undocumented due to the fact that illegal WEEE treatment does seem to exist in Belgium and The Netherlands. Hence, an *unidentified stream* is introduced into the system boundary, which was also suggested in several national WEEE reports in Belgium and The Netherlands (Recupel, 2018; 2013; Huisman et al., 2012).

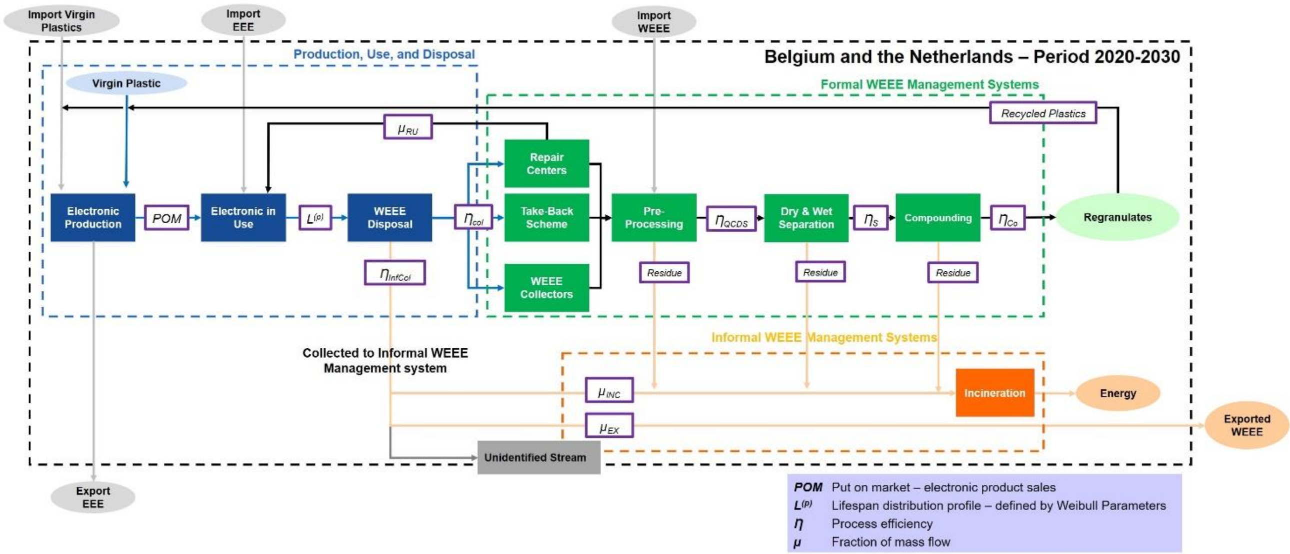


Figure 3.1 Overview of vacuum cleaners, coffee machines, and electric shavers lifecycle from the production to recycling phase in Belgium and The Netherlands used for material flow analysis. The flowsheet is based on previous studies and reports (De Meester et al., 2019; Recupel, 2018; 2013; Huisman et al., 2012).

Table 3.1 Parameter values, symbols, and data points used for mass balance calculation

Lifecycle of (W)EEE	Symbol	Value	Unit	Data Source
Production, use and disposal				
WEEE to recycling facilities	η_{Col}	49	(%)	(Recupel, 2018; National (W)EEE Register, 2019; Global E-Waste Statistics Partnership, 2020)
WEEE to informal systems	η_{InfCol}	51	(%)	
Post-disposal				
Imported WEEE	μ_{import}	20*	(%)	(Recupel, 2018; Huisman et al., 2012)
WEEE to reuse	μ_{Ru}	0	(%)	
Pre-processing	η_{QCDS}	48	(%)	(De Meester et al., 2019; Van Eygen et al., 2016)
Dry and wet separation	η_{S}	90	(%)	
Compounding	η_{C}	90	(%)	
WEEE to incineration	μ_{INC}	21	(%)	
WEEE to export	μ_{Ex}	10	(%)	
Unidentified Stream	-	20	(%)	

**Import WEEE equals to 20% of the total WEEE generated in each evaluation year (Recupel, 2018; Huisman et al., 2012)*

3.2.2.3 Description of mathematical framework used in material flow analysis

Based on Figure 3.1, mathematical equations are elaborated to calculate the mass balances. The equations are constructed to quantify annual regranulates production via formal WEEE management systems. Total regranulates production per year can be determined by multiplying the aggregated mixed WEEP (1000 kg) and imported WEEP (200 kg) from the selected EEEs in the evaluation year *n* [*W(n)*] with the collection (η_{Col}), pre-processing (η_{QCDS}), separation (η_{S}), and compounding (η_{Co}) efficiencies. The mathematical expression of annual regranulates production from the formal WEEE management system is shown in the Equation 3.1:

$$\boldsymbol{Regranulate\ production\ from\ WEEE\ recycling} = [(\boldsymbol{W(n)} * \eta_{\boldsymbol{Col}}) + (\boldsymbol{W(n)} * \mu_{\boldsymbol{import}})] * \eta_{\boldsymbol{QCDS}} * \eta_{\boldsymbol{S}} * \eta_{\boldsymbol{Co}} \qquad (\boldsymbol{tonnes\ /\ year})$$

Equation 3.1

However, 51% of the total WEEP generation are forwarded to informal WEEE management systems (η_{InfCol}). The total WEEP that goes to incineration equal to the sum of the recycling processes residue and miss sorted WEEP, as also indicated in the previous studies (De Meester et al., 2018; Vanegas et al., 2017). As for the portion of export WEEP, a fixed value is used, which was derived from the literature (Recupel, 2018; 2013; Huisman et al., 2012). The mathematical expression to quantify WEEP to export and incineration can be seen in Equation 3.2 and Equation 3.3, respectively:

$$WEEP\ to\ Export = W(n)\ x\ \mu_{EX} \qquad \text{(tonnes / year)} \qquad \text{Equation 3.2}$$

$$WEEP\ to\ Incineration = [W(n) * \mu_{INC}] + [W(n) * \eta_{Col} * (1 - \eta_{QCDS})] + [W(n) * \eta_{Col} * \eta_{QCDS} * (1 - \eta_{S})] + [W(n) * \eta_{Col} * \eta_{QCDS} * \eta_{S} * (1 - \eta_{Co})] + [(W(n) * \mu_{import}) * (1 - \eta_{QCDS})] + [(W(n) * \mu_{import}) * \eta_{QCDS} * (1 - \eta_{S})] + [(W(n) * \mu_{import}) * \eta_{QCDS} * \eta_{S} * (1 - \eta_{Co})] \qquad \text{(tonnes / year)} \qquad \text{Equation 3.3}$$

As already mentioned earlier, some of the WEEE streams cannot be determined; hence it is considered as unidentified stream that accounts for 20% of the total WEEE generated (Recupel, 2018; Huisman et al., 2012). This is the remaining portion of WEEE that is neither being sent for energy recovery nor exported. The mathematical expression to quantify the unidentified stream is as follows:

$$Unidentified\ Stream = W(n)\ x\ (\eta_{InfCol} - \mu_{INC} - \mu_{EX}) \qquad \text{(tonnes / year)} \qquad \text{Equation 3.4}$$

3.2.3 Multivariate Input-Output Analysis

In WEEE management systems, an accurate quantitative estimation of WEEE generation and composition is complicated. To address this challenge, multivariate input-output analysis (MIOA) developed by Wang et al. (2013) is applied. The MIOA considers socio-economic dynamics of electronic products flowing into the society (sales), then accumulate in the technosphere (stock), and eventually reaching the end-of-life after certain period of time (lifespan) as WEEE. The interconnections of sales, stock, and lifespan are modelled in MIOA, primarily by modeling the lifespan distribution model using Weibull distribution (Wang et al., 2013). For more information of MIOA, it is advised to read Wang et al. (2013) and Wang (2014).

3.2.3.1 Projection on WEEE generation in 2030

To predict the future WEEE generation, and its associated WEEP, MIOA is applied. Specifically, we apply a model developed by Wang et al. (2013) to measure the interconnection between the product sales and lifespan distribution of electronic products. For a better estimation, an approach that considers the dynamic nature of product obsolescence should be applied, contrary to the assumption of fixed, normally average, product lifespan (Parajuly et al., 2017). This approach generally describes the percentage and amount of EEEs that are accumulated in the market (stock) and flowing out as WEEE based on their age (owing to the year of product sales) in the evaluation year 2030. Through this approach, we acknowledge that electronic products that accumulate in the market have different obsolescence rate and have satisfied their service lifetime and storage time.

The result of this modeling, which is the *disposal age composition* in evaluation year *n* [*W(t,n)*], is combined with MFA to quantify the recycling rate and regranulates production from WEEE recycling process. The MIOA method incorporates *historical sales data* [*POM (t)*] and EEE *lifespan distribution profile* [($L^{(p)}(t,n)$)] computation in determining total outflow of WEEE in the evaluation year 2030.

3.2.3.2 Mathematical framework of multivariate input-output analysis

The term *disposal age composition of WEEE* [*W(t,n)*] describes the amount of WEEE from the past (year *t*) being disposed of in the evaluation year *n*, in which *n* is the evaluation year 2030 in this research. The total amount (in mass) of disposal age composition in the evaluation year 2030 can be quantified using Equation 3.5:

$$W(t,n) = \sum_{t=t_0}^{n} POM\ (t,n)\ x\ L^{(p)}(t,n) \qquad \text{(tonnes / year)} \qquad \text{Equation 3.5}$$

The amount of WEEE generation *W(t,n)* is the multiplication of vacuum cleaner, coffee machine, and electric shaver sales (POM) and lifespan distribution profiles [($L^{(p)}(t,n)$)]. Product sales data are obtained from the Eurostat database and National WEEE Report (Eurostat, 2022b; Recupel, 2018; National WEEE Register, 2019) and lifespan distribution was modelled using the Weibull distribution function. The Weibull distribution is chosen because it demonstrates the lifespan distribution of electronic products from being used until being thrown away by the consumers, including the 'hibernation' time. Moreover, the Weibull distribution profiles fit the disposal age compositions of WEEE that were surveyed by Wang (2014) in 2006 and 2007. The asymmetric profile of Weibull distribution also aligns with the fact that the characteristics of EEE lifespan can differ depending on the technological development or socio-economic conditions (Tran, 2018; Wang, 2014; Nordic Council, 2009). Detailed information related to the product sales data and its calculation is summarized in Appendix B–section 1.

The lifespan distribution [($L^{(p)}(t,n)$)] describes the probable obsolescence rate of EEEs in evaluation year *n* of the batch of products sold in historical year *t*. In other words, lifespan distribution denotes the probability (shown in percentage, %) of products being sold in the past (year *t*) that will become waste in evaluation year *n*. The mathematical equation of selected electronic products' obsolescence rate is presented in Equation 3.6, which is developed and validated by Wang et al. (2013) and Wang (2014):

$$L^{(p)}(t,n) = \frac{\alpha(t)}{\beta(t)^{\alpha(t)}}\ (n-t)^{\alpha(t)-1} e^{-\left(\frac{n-t}{\beta(t)}\right)^{\alpha(t)}} \qquad (\%) \qquad \text{Equation 3.6}$$

Here, the shape parameter $\alpha(t)$ and scale parameter $\beta(t)$ are used for the computation of lifespan distributions (Equation 3.6). The shape parameter, $\alpha(t)$, is the failure or obsolete rate of EEEs while the scale parameter, $\beta(t)$, also known as life characteristics, is where the bulk (67,5%) of the lifespan distribution lies (Murakami et al., 2010; Oguchi et al., 2010; Wang et al., 2013). As the lifespan of products changes through time, because of social and technical development of EEE, these parameters vary over time and have to be modelled corresponding to each historical sales year (Van Eygen, et al., 2016). In this research, the shape and scale parameter data are obtained from Eurostat database and previous research (Eurostat, 2022b; Forti et al., 2018; Wang, 2014). Detailed Weibull parameters data can be found in Appendix B–section 2.

Thereafter, total WEEP generated in 2030 can be quantified by multiplying the disposal age composition [*W(t,n)*] with the average composition of plastics of the selected products (see section 3.2.1). The mathematical expression to quantify total plastics for each the selected WEEE can be found in Equation 3.7.

$$WEEP\ (t,n) = \sum W(t,n) * Average\ Plastics\ Content\ (\%)\ (\text{tonnes / year}) \quad \text{Equation 3.7}$$

Finally, once the mass of WEEP generated in 2030 [*WEEP(t,n)*] has been calculated, the material flow analysis through the formal and informal WEEE management systems are investigated. The flow of WEEP in 2030 can be quantified by replacing the functional unit (1000 kg) of aggregated mixed plastics [*W(n)*] with the mass of WEEP [*WEEP(t,n)*] to Equation 3.1 – 3.4.

3.2.4 Sensitivity analysis and scenario analysis

Sensitivity analysis is conducted to identify the weight (influence) of a parameter in the case-specific model or study towards model results (Bisinella et al., 2016). In Chapter 3, a sensitivity analysis is performed on some of the selected parameters to identify which parameter has more influences on the model results and needs more attention in future research. Moreover, a sensitivity analysis in Chapter 3 is also carried out to identify the bottlenecks within the current WEEE management system. This approach is also applied to gain insights into a potential room for improvement of WEEE management systems in the future. For this purpose the selected parameters are varied from 0 – 100% one by one while maintaining the remaining parameters at a constant value. At every interval variation, the results of recycled plastics content (i.e., recycling rates) are recorded and plotted against the parameter changes. Five parameters are selected for the sensitivity analysis, which is separate collection rate (η_{Col}), pre-processing (η_{QCDS}), dry and wet separation (η_S), compounding (η_{Co}), and the WEEE import stream.

Lastly, a prediction of how changes in collection rate, pre-processing efficiency, and material composition impact recycling rate or regranulates availability was elaborated. In the coming years, three trends are expected to occur simultaneously: collection rates should increase, pre-processing efficiency might improve, and composition and designs might change. In this scenario analysis, a study case on how these changes impact regranulates production from vacuum cleaner waste recycling is elaborated. The change in collection rates is driven by the WEEE Directive while the material composition change because the market share of RVCs is expected to increase in the future (Rames et al., 2019; Parajuly et al., 2016). The RVCs use batteries and different electronic components, which reduce the share of plastics content to 55% (Parajuly et al., 2016). As for pre-processing technology, we assume that recyclers will keep improving their technologies. Therefore, three different scenarios were performed in which the parameters are varied simultaneously. Ultimately, the time in which the recycling target would be achieved can be revealed.

Three scenarios are developed namely *base, intermediate,* and *positive* scenarios. In the base scenario, the collection rate is expected to have an incremental increase of 1.4% annually (Global E-Waste Statistics Partnership, 2020). The market share of RVC is expected to increase by 1.5% annually (Rames et al. 2019). Lastly, in this scenario, we also assume that the efficiency of pre-processing technology will have an incremental increase of 1.4% annually similar to the collection rate. For the intermediate and positive scenarios, the incremental increase of the collection rates, pre-processing efficiency, and market share of RVCs are assumed to be slightly higher, 3% and 5% respectively. Detailed information about the incremental changes can be found in Appendix B–section 3.

3.3 RESULTS AND DISCUSSION

3.3.1 Material flow analysis of the current WEEE management system

The material flow of aggregated 1000 kg of mixed plastics waste (WEEP) throughout the current WEEE treatment in Belgium and the Netherland can be seen in Figure 3.2. The summary of mass balances within the investigated system can be found in Table 3.2. From this material flow diagram, it can be seen that the majority of plastics (i.e. 632 kg and equal to 63% of the total input) are sent for energy recovery (incineration) because of two reasons: firstly, a fair amount of WEEE is informally collected and secondly, significant losses occur due to inefficient pre-processing technology. Instead, the recycling rate of mixed WEEP to regranulates is only 22%, i.e. equal to 268 kg in Figure 3.2. The overall recycling rate of plastics is considered to be relatively low compared to the recycling rates of some other materials, for example, iron and aluminium with more than 85% recycling rates (De Meester et al., 2019; Van Eygen et al., 2016).

Moreover, from the FU of 1000 kg, a total of 632 kg of mixed WEEP ends up in incineration, of which 422 kg (67%) comes from the residues of the WEEE recycling chain. In comparison, only 210 kg (33%) comes directly from informally collected WEEE. These findings demonstrate the limitation of the current recycling technologies in producing high-quality regranulates from mixed WEEP, which are actually intended to be recycled content in high-end applications (e.g. closed-loop recycling).

When looking at the pre-processing step, plastics can be lost for several reasons. Dismantling and depollution is labor-intensive, in which trained labors manually dismantle hazardous substances and electronic components (e.g., copper cable and battery) from WEEE on moving conveyor belts (Makenji and Savage, 2012; Menad, 2016), which is a phase that is typically not dedicated to the plastic fraction (Buekens and Yang, 2014). Next to that, commonly used mechanical shredder based processing still underperforms related to plastics (Vanegas et al., 2017). This is partially caused by the processes themselves, for example by particle sizes issues in the sorting and separation technologies (Maisel et al., 2020), but also by design as plastics can be attached to other elements. Sorting and separation might even get worse in the future as new electronic designs tend to be smaller and thinner.

The size of particles after shredding can also affect the efficiency of plastics separation in further steps, and this needs to be optimized to minimize losses during and after shredding. For example, magnetic separation and eddy current separation work fine to remove metals at a size of 1 – 150 mm, but sensor-based technologies, like near-infrared spectrometer (NIR) or x-ray fluorescence (XRF), need particle sizes larger than 10 mm up to 120 mm to work optimally. If the particle sizes requirements are not met (e.g., too small or large), this can contribute to the decreases of sortability and increases of material losses. Additionally, fine fractions smaller than 10 mm is nowadays are often the materials being sent to landfilling or incineration, making integration between one pre-processing step to another becomes crucial (Maisel et al., 2020).

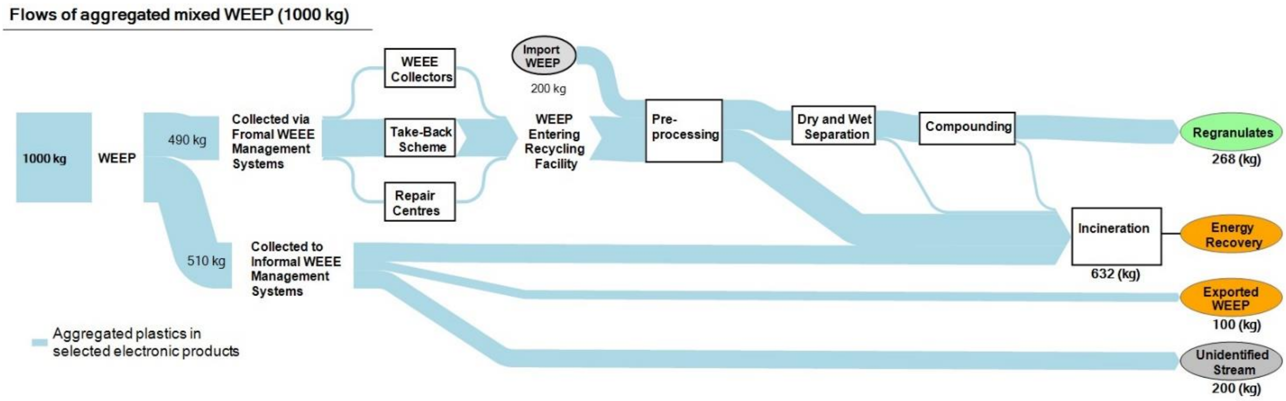


Figure 3.2 Flows of aggregated mixed WEEP (1000 kg) from vacuum cleaners, coffee machines, and electric shavers within the current WEEE management system in Belgium and The Netherlands.

Table 3.2 Effective (intermediate and final) outputs of 1000 kg of aggregated mixed WEEP from discarded vacuum cleaners, coffee machines, and electric shavers within current Belgium and The Netherlands WEEE management systems.

	Unit	Aggregated Mixed WEEP from Selected Products
Total aggregated mixed WEEP	kg	1000
WEEP Entering Recycling Chain		
Formally collected	kg	490
• via take-back scheme	kg	416
• via other WEEE collectors	kg	37
• via repair centers	kg	37
Imported WEEE	kg	200
Plastics at Recycling Facility		
Pre-processing	kg	690
• to separation process	kg	331
• to incineration (residue)	kg	359
Dry and wet separation	kg	331
• to compounding process	kg	298
• to incineration (residue)	kg	33
Compounding	kg	298
• to secondary materials	kg	268
• to incineration (residue)	kg	30
Outputs		
Regranulates	kg	268
To Incineration	kg	632
Exported	kg	100
Unidentified Stream	kg	200

3.3.2 Improving the current WEEE management systems: Belgium and The Netherlands case

The result of sensitivity analysis of the current formal WEEE management system is presented in Figure 3.3. The x-axis shows the parameter values of collection rates, recycling efficiencies, and import stream from 0% – 100% while the y-axis shows the WEEP recycling rate. The current state of the WEEE management system, with 49% collection rates, 48% pre-processing efficiency, 90% dry-wet separation and compounding efficiencies, and 20% import stream indicates that we can recycle 22% of WEEP back to granulates. If the flow of import stream is excluded in the calculation, the recycling rate drops from 22% to 19%, as shown by horizontal orange dashed line Figure 3.3. The graph can be interpreted in a way that the curves with the steeper slope hold a more significant impact on the improvement of WEEP recycling rate.

From Figure 3.3 we can see that the target to recycle 25% of plastics from WEEE can be achieved either by collecting 64% of the annual WEEE generated (green line), improving the overall efficiency of pre-processing technology to 64% (purple line), or importing more WEEE up to 40% of the annual WEEE generation in Belgium and The Netherlands (blue line). As for the further dry & wet separation and compounding processes, they already have a relatively high efficiency (90%) thus do not create much impact in improving the recycling rates (grey line).

The overall result of sensitivity analysis demonstrates that there is plenty of room for improvement to increase WEEP recycling by enabling more efficient collection and developing better pre-processing technologies. Consumers thus need to be educated about the separate collection of WEEE as well as the awareness to bring their used EEEs to the collection points. Pre-processing can still be improved a lot for better plastics recycling by, for example, promoting design for recycling of EEEs. While mechanical recycling remains the most ubiquitous pathway to recycle WEEP, design for recycling strategy should be implemented in a larger and massive scale in the future (De Meester et al., 2019; Vyncke et al., 2018; Rodrigo & Castells, 2003). Additionally, at the component level, designing EEEs with less complex material combinations will also greatly impact the improvement of the mechanical recycling process (Makenji and Savage, 2012). Later, the quality and technical properties of regranulates to be used in the EEEs should also be considered, which is not yet addressed in this research.

The sensitivity analysis can also be used to estimate the impact of regulations on WEEP recycling. One of the notable changes that were made in WEEE Directive 2012 is a new collection target, which should be achieved either by collecting 65% of average sales in three preceding years or 85% of the annual WEEE generation (European Commission, 2018d; 2012). Only by complying with the EU regulation to collect 85% of annual WEEE generated while keeping the other parameters constant, we can expect an increase in the plastics recycling rate from 19% to 33% in Belgium and The Netherlands, as shown by the green line on Figure 3.3.

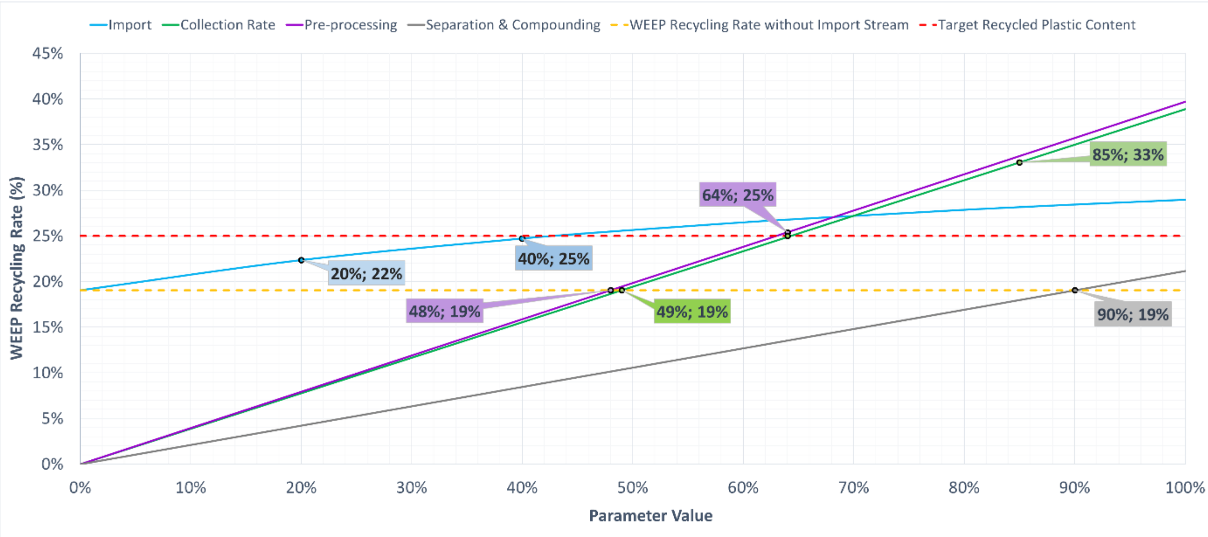


Figure 3.3 Sensitivity analysis of the effect of parameter changes on WEEP recycling rates from the current WEEE recycling chain. Each line was created by varying a single parameter (e.g., separate collection rate from 0% to 100%, shown in green line) while keeping the other parameters constant (i.e., pre-processing, dry and wet separation, compounding, and import). In the boxes, the number on the left denotes the value of the parameter (i.e., collection rates or recycling efficiencies) and the number on the right denotes the WEEP recycling rates.

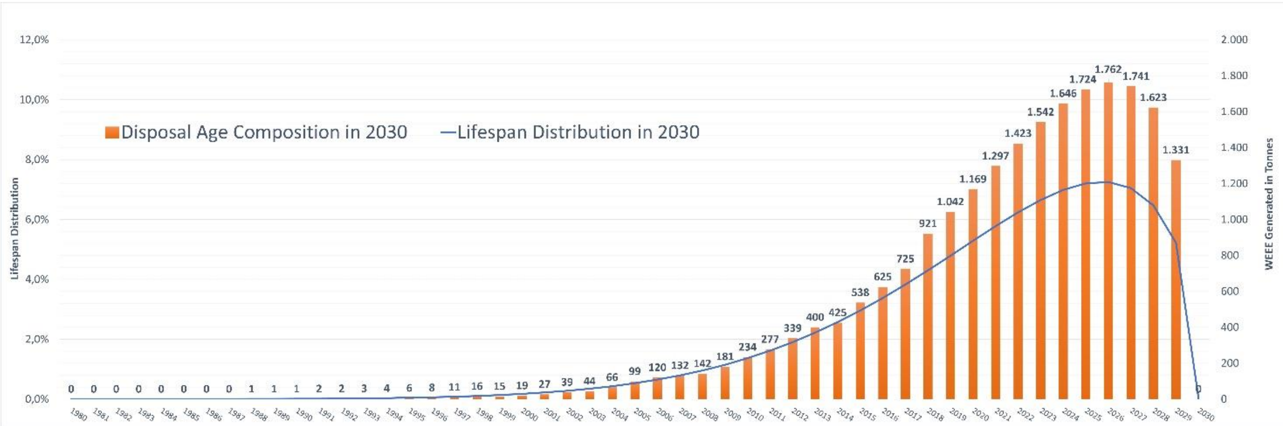
3.3.3 Predicting the future amount of WEEE generation and recycled plastic from WEEE: in case of *status quo* recycling chain in 2030

The result of lifespan distribution and disposal age composition of generated WEEE in 2030 are determined using MIOA. The results for vacuum cleaners (A), coffee machines (B), and electric shavers waste (C) in the evaluation year 2030 can be found in Figure 3.4. The summary of total selected WEEEs generation can be found in Table 3.3. From this mathematical modelling, we forecast that 21.722, 11.621, and 5.951 tonnes of WEEP in respectively vacuum cleaners, coffee machines, and electric shavers would be disposed of by the consumers in Belgium and The Netherlands in 2030. However, we can expect lower values of WEEP generated from the selected EEEs in 2030 because newer RVCs and coffee machines have lower plastics concentrations (Mudgal et al., 2011; Parajuly et al., 2016). Furthermore, Figure 3.4 also reveals the age composition for each selected WEEE in 2030. For example, the lifespan distribution curve (blue in Figure 3.4A) describes the probability of vacuum cleaners purchased in 2012, 2017, and 2022 to be disposed of in 2030 are 2%, 4%, and 6% respectively.

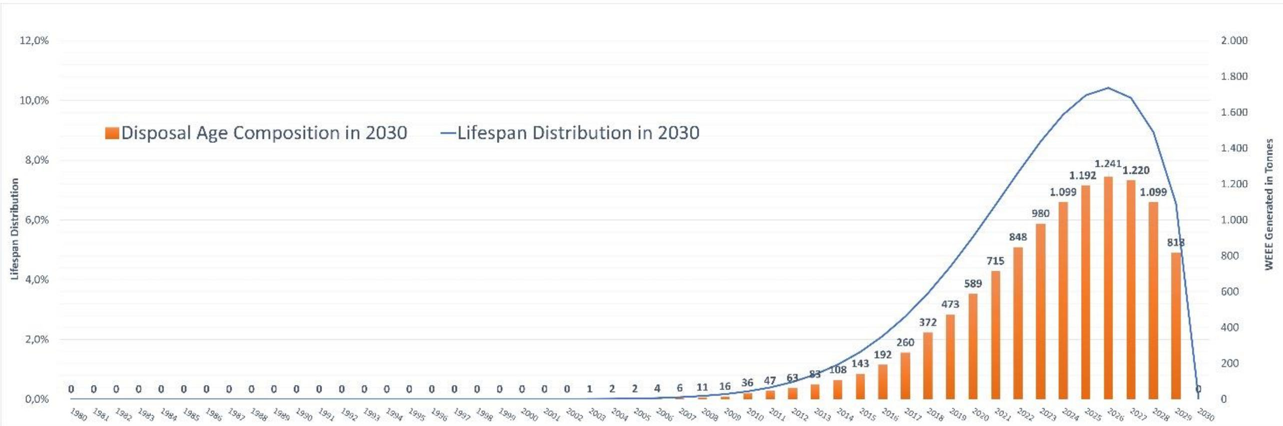
The results of this projection are dependent on the Weibull parameters α and β that establish a different disposal rate for every electronic product. In Figure 3.4A and Figure 3.4C, we can observe that a fair quantity of vacuum cleaners and electric shavers waste disposed in 2030 might originate from the products purchased between 1990 – 2010, equal to 5,3% and 9,7% respectively. In contrast, less than 1% of coffee machines waste would origin from the same purchasing years (Figure 3.4B). Vacuum cleaners and electric shavers thus have a slower disposal (obsolescence) rate compared to coffee machines. Several reasons underlie this trend, such as technological advancement and consumers behavior. Market conditions trigger faster disposal of the old electronic products, especially when significant changes were made in the functionality of an electronic product (Tran, 2018; Mudgal et al., 2011; AEA, 2009). This pattern can also indicate that some defective or used electronic products, like vacuum cleaners and electric shavers, stay for quite a long time in the consumers' possession before finally discarded and brought to the WEEE collection points. In Figure 3.4, the sum of vacuum cleaners that was purchased between 2020 – 2029 accounts for 75% of the total vacuum cleaner waste in 2030 while the same purchasing period accounts for 84% of the total coffee machine waste in the same evaluation year. Ultimately, by gathering and interpreting this information from the developed model, we can then estimate the potential resources of recycled plastics, including the treat to encounter hazardous substances therein, which will be discussed in the next section.

Using the simplification that the overall efficiencies of collection, recycling technologies, and the average material composition of the selected EEEs remain the same by 2030 compared to our reference year 2016 (Table 3.1), the WEEE recycling facilities are expected to receive 17.445 tonnes of plastics from the formally collected WEEE and from WEEE import, whereas 12.895 tonnes would end up in informal WEEE management systems. After the recycling processes, 6.783 tonnes of WEEP will be potentially recycled from the selected WEEEs while a total of 23.557 tonnes of plastics will be sent for incineration, exported, or undocumented (Figure 3.5).

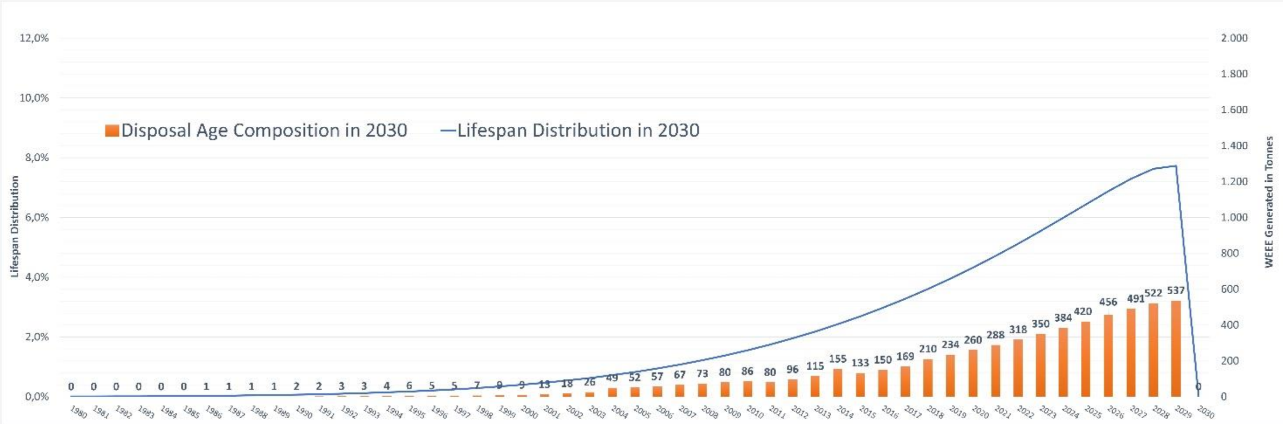
If there is a status quo in the collection and separation efficiencies, the potential use of recycled content of plastic in 2030 would thus remain in 19% (assuming closed-loop recycling within EEE sector). This value might even decrease because the regranulates can also be used in other industries such as automotive or construction (open-loop recycling). Nevertheless, to achieve ambitious recycling targets and pledges related to use of recycled content, these parameters would thus need to improve.



(A) *Vacuum cleaners*



(B) *Coffee machines*



(C) *Electric shaver*

Figure 3.4 Lifespan distribution profile of the selected products and estimation of waste generation in 2030. In this figures, the blue curves represent the lifespan distribution profile, shown in percentage (%), and the orange bars represent the disposal age composition, shown in mass (tonnes), of (A) vacuum cleaner, (B) coffee machine, and (C) electric shaver waste.

Table 3.3 Results of MIOA in combination with MFA to predict the flows of WEEP in the evaluation year 2030, considering the reference process efficiencies in Table 3.1.

	Unit	Vacuum Cleaner Waste	Coffee Machine Waste	Electric Shaver Waste
Total Mass of WEEE	tonnes	21.722	11.621	5.951
• Mixed WEEP	tonnes	15.857	6.392	3.035
• Other materials	tonnes	5.865	5.229	2.916
WEEP Entering Recycling Chain				
• Collected WEEP	tonnes	7.770	3.132	1.487
• Import WEEP	tonnes	3.171	1.278	607
WEEP at Recycling Facility				
• Pre-processing	tonnes	10.941	4.410	2.094
• Dry and wet separation	tonnes	5.252	2.117	1.005
• Compounding	tonnes	4.727	1.905	905
Outputs				
• Regranulates	tonnes	4.254	1.715	814
• To Incineration	tonnes	10.018	4.038	1.916
• Exported WEEP	tonnes	1.586	639	304
• Unidentified Stream	tonnes	3.171	1.278	607

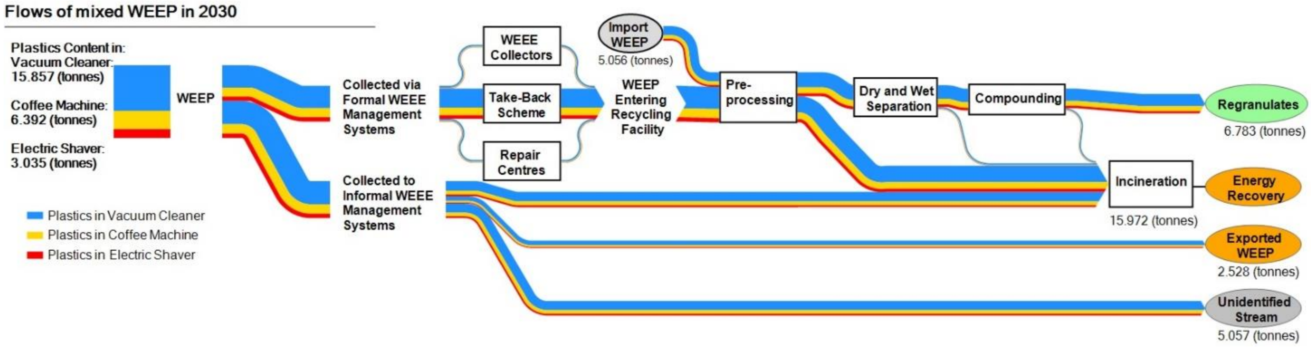


Figure 3.5 Estimated flows of mixed WEEP from vacuum cleaner, coffee machine, and electric shaver waste towards different end-of-life final destinations in evaluation year 2030 (only expressed in tonnes of mixed WEEP in respective electronic product based on material composition explained in section 3.2.1)

3.3.4 Predicting the future amount of WEEE generation and recycled WEEP in case of significant improvements of the recycling chain

In the base scenario the collection rates, pre-processing efficiency increases from 49% to 63% and from 48% to 62% by 2030. Because the composition of plastics content in RVC drop to 55%, the average plastics content is expected to drop from 73% to 69% by 2030. For the intermediate scenario, the collection rates and pre-processing efficiency increase up to 69% and 78% respectively, while the plastics content will drop to 66% by 2030. Lastly, the collection rates and pre-processing efficiency will increase up to 94% and 93% respectively, whilst the plastic content will decrease up to 62% in the positive scenario. Detailed information about the incremental changes of these parameters every year from 2020 to 2030 can be found in the Appendix B–section 3.

The impact on the availability of regranulates from three scenarios can be found in Figure 3.6. With a slow improvement of collection rates and pre-processing technologies such as in base scenario, the target in using 30% of recycled plastics content can be achieved by 2026 or 2027 while the ambition to use 50% of recycled plastics seem to be unfeasible (red line in Figure 3.6). As we accelerate the improvement of collection rates and pre-processing technologies the ambition to use 30% recycled content of plastic can be achieved in 2022 or 2023 for positive and intermediate scenarios respectively (blue and green line in Figure 3.6). Moreover, the intention to use 50% of recycled plastics in electronic products become feasible but it can be realized in different time. In the intermediate scenario the 50% recycled plastics will be available by 2029 whereas 50% of recycled plastics content can already be realized by 2026 in the positive scenario.

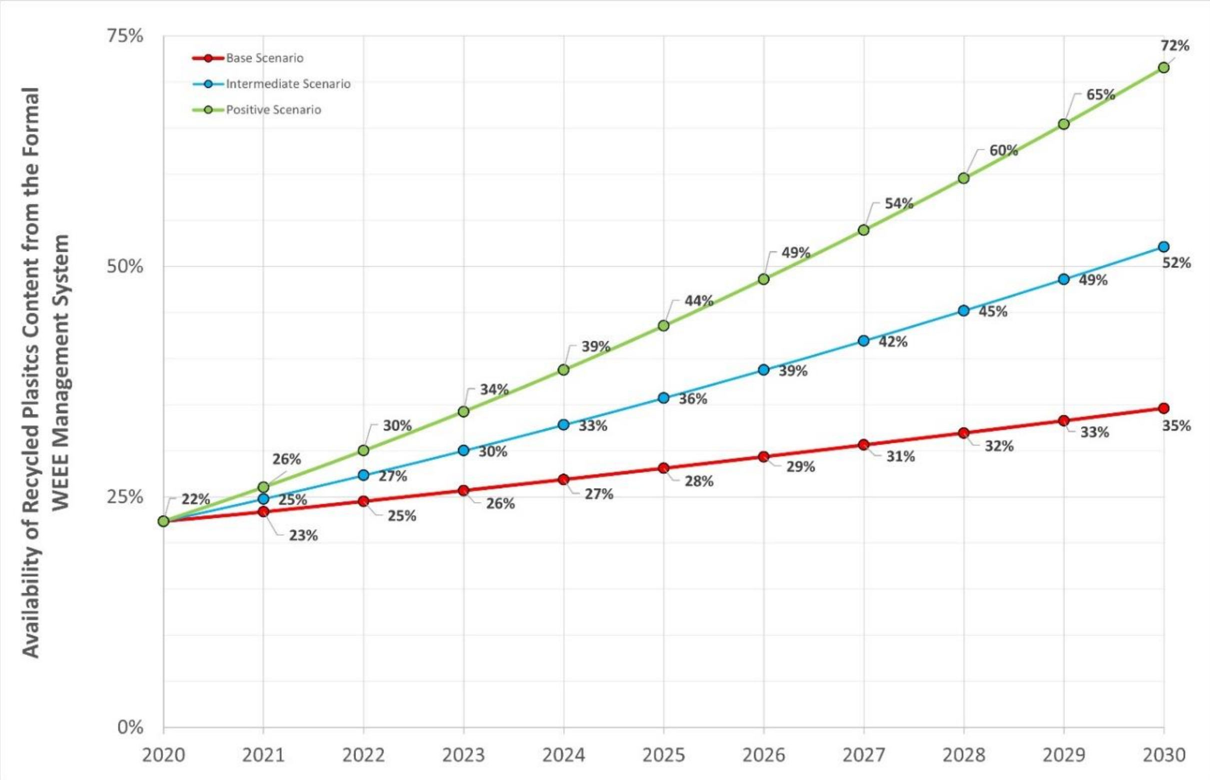


Figure 3.6 Prediction of the availability of recycled plastics from the formal WEEE management system for base scenario (red line), intermediate scenario (blue line), and positive scenario (green line).

3.3.5 Assessment of potentially hazardous substances from WEEE in 2030

The presented model has the ability to determine the origin of the incoming waste in 2030, which include the inherited risk of legacy chemicals therein.

BFRs, for example, have been commonly applied in a wide range of plastics material to prevent fire but have been prohibited from being used in electronic products (Delva et al., 2018; Julander et al., 2005; Sjödin et al., 2001). Despite the restriction of BFRs since 2006 by the European Commission, there is a risk of recyclers still encountering these legacy substances due to the dynamic disposal rates of electronic products. Figure 3.7 reveals the predicted portion of mixed WEEP from the selected products that potentially still contain BFRs in 2030 because the products were sold before the implementation of the RoHS Directive in 2006. The amount of mixed WEEP potentially still containing BFRs from the coffee machine wastes is significantly lower (0,08%) compared to vacuum cleaners (2,23%) and electric shavers (4,61%), mainly because the coffee machine has a faster disposal rate than the other products (as discussed and elaborated in section 3.3.3).

The assessment of potentially hazardous substances from generated WEEE in 2030 shows that BFRs might still be encountered years after the implementation of RoHS Directive was put into force in 2006, hence technologies to prevent the adverse impact of BFRs still need to be considered. The WEEE recycling industry has been strongly regulated in Europe in processing hazardous wastes, like WEEP containing BFRs, and the level of hazardous substances in such streams must be monitored and traced (Bates et al., 2019; Wagner and Schlummer, 2020). This was done to prevent the adverse effect to human health and environment because a number of these flame retardants are considered as Persistent Organic Pollutants (POPs), such as Deca-BDE and the group of PBDEs. According to the WEEE Directive Annex VII, WEEP containing POP BFRs has to be removed and treated at a specialized WEEE recycling facility to not end up in the regranulates (Haarman et al., 2020; European Commission, 2012). The recycling facility shall ensure the segregation of these mixed WEEP containing POP BFRs if the total bromine content is above 2.000 ppm (0,2%) or POP BFRs are above 1.000 ppm (0,1%) (Haarman et al., 2020; RoHS Guide, 2020; EERA, 2018). In this context, BFR separation process such as XRT, XRF or density separation will still remain relevant for a while. Such processes keep Bromine concentrations below the limit. Another options such as dissolution based recycling or chemical recycling could be promising to avoid BFRs to end up in the secondary resource.

Although a detailed characterization of BFRs and other hazardous substances is not done in this research, the developed model serves as the baseline to further assess the risk and opportunity to recycle WEEP containing hazardous substances.

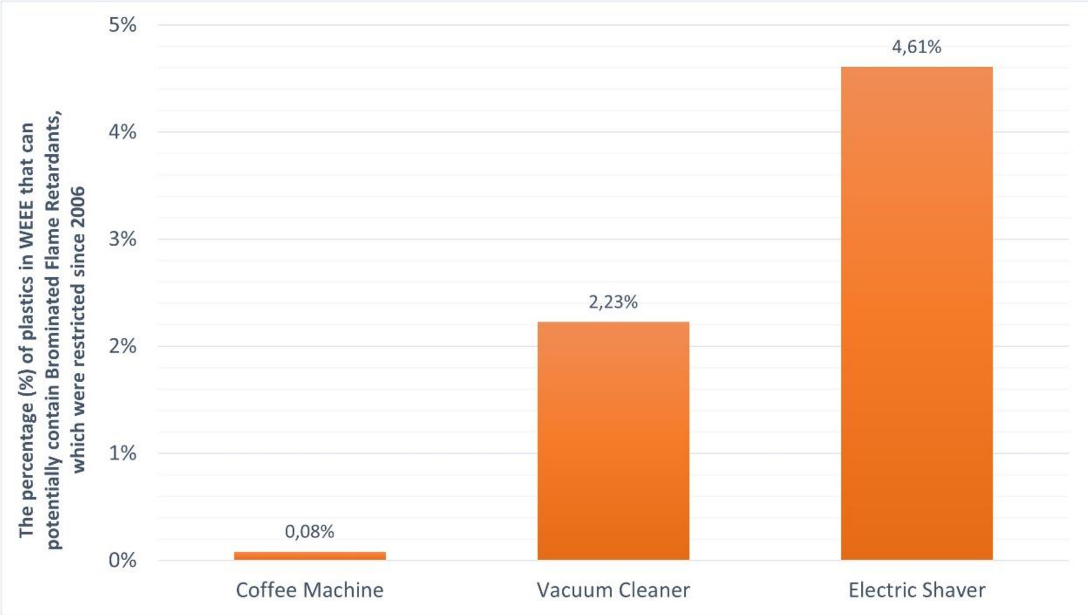


Figure 3.7 The probability (shown in %) of WEEP that potentially still contain brominated flame retardants (BFRs) in the evaluation year 2030 based on its disposal age composition of WEEE [W(t,n)] in Figure 3.4, which implies that the WEEP originated from products being sold before the effective implementation of RoHS Directive in 2006.

3.5 CONCLUSION

Electronic waste is vastly growing, leading to an enormous source of secondary resources. One of the challenges to enable circular economy for EEE is to better predict evolutions in the WEEE recycling chain in the future. In this study MFA and MIOA techniques are applied to investigate the current and future performance of WEEE management systems, with a focus on the SHA category and a geographical scope of Belgium and The Netherlands.

Our findings suggest that the plastics recycling rate from WEEE in Belgium and The Netherlands is 22% when the import streams are included and 19% without the import streams. A sensitivity analysis highlights the limitation of current pre-processing technology and collection schemes, which have become the WEEE management systems’ bottleneck. The model's outputs also suggest that the target to recycle and use 25% of plastics from WEEE can be achieved by improving the collection rate or pre-processing technologies efficiency to at least 64%. Moreover, there is plenty of room for improvement to go beyond the established target by improving the collection and pre-processing technology within the WEEE recycling chain.

From a scenario development improving different factors starting from base scenario up to a positive scenario leads to a different time to achieve the target. With a relatively slow improvement in the base scenario, the target to use at least 30% of recycled plastics content can be achieved by 2027, and it is unfeasible to use 50% recycled plastics content by 2030. On the other hand, the target to use 30% recycled plastics can be achieved sooner in 2023 if we accelerate the improvement of collection and pre-processing technologies. Moreover, significant improvement within the WEEE recycling chain makes it feasible to pledge to use 50% recycled plastics, as shown in intermediate and positive scenario. The intention to use 50% recycled plastics can be accomplished by 2029 in intermediate scenario and by 2025 in positive scenario.

The MIOA reveals how the disposal rate of EEEs impacts the WEEE age composition in 2030. The risk of encountering restricted hazardous substances like BFRs persist years after the implementation of RoHS Directive in 2006. It is expected that 4,6% and 2,3% of WEEP originating from electric shavers and vacuum cleaner waste still potentially contains BFRs while only 0,08% of WEEP from coffee machines waste will still potentially contain BFRs by 2030. This finding implies that WEEP containing BFR might still need to be separated in 2030 to ensure that the concentration of BFRs stays below the threshold set by regulation in Europe.

CHAPTER 4: RECYCLING PERFORMANCE OF AN IMPROVED MECHANICAL RECYCLING PROCESS FOR HOUSEHOLD FLEXIBLE PLASTICS

Redrafted from:

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With additions of economic and cost-benefit analysis modeling from:

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Chapter 4

Material flow analysis, recycling performance, and economic balance of an improved mechanical recycling process for post-consumer household flexible plastics

4.1 INTRODUCTION

4.1.1 General overview of the flexible packaging waste management

Management of post-consumer flexible packaging (PCFP) waste from households is a pressing issue globally as it is considered as one of the major contributor to the losses of macroplastics to the environment (Ryberg et al. 2019; Peano et al., 2020; OECD, 2022). This is caused by improper treatment and disposal of plastic waste, which is due to the fact that PCFP can be expensive to collect and they have a low market value (SYSTEMIQ, 2022; Peano et al., 2022). On top of the effort to reduce the use of plastics, design for recycling efforts should further be introduced such as changing the design from multi- to mono-material, and changing the business model such as building reuse or refill stations (SYSTEMIQ, 2022; OECD 2022; Feber et al., 2020). Next to this, also the waste management infrastructure of PCFP can be further improved such as establishing separate collection (i.e., source separation), sorting and recycling systems (PRI, 2018; Ellen MacArthur Foundation, 2016; Horodytska et al. 2019).

The current status of PCFP waste treatment is that it is exported, recycled into products like park benches, garden furniture, or bin bags. In this context, there is an urgent need to deploy more recycling capacity that is capable of reprocessing PCFP. To improve this, several options exist, including improved mechanical recycling, chemical recycling (e.g., pyrolysis), solvent-based recycling (e.g., dissolution-precipitation or delamination), and energy recovery (Horodytska et al. 2019; Civancik-Uslu et al., 2021; Larrain et al., 2021; 2020; Ragaert et al. 2017; Simon and Martin, 2019; Vollmer et al. 2020; Ügdüler et al., 2021; Kol et al., 2021). While some promising chemical recycling approaches are still under development such as solvent-based recycling (Simon and Martin, 2019; Schwarz et al., 2021; Vollmer et al., 2020; Hann and

Connock, 2020), other technologies can be readily implemented at higher technological readiness level (TRL) and at commercial scale such as improved washing and extrusion (Horodytska et al. 2019; Horodytska et al. 2020). In any case, the requirement for increased recycling rates is urgent, and thus it is crucial to assess how far currently available technologies can be ‘pushed’ to increase the process yield and regranulates’ quality from PCFP recycling. Therefore, this study focuses on assessing the potential improvement in terms of process yield and regranulates’ quality of mechanical recycling of PCFP based on potential flowsheets that are proposed by industry with technologies that are commercially used (e.g., hot washing, extrusion with degassing, and deodorization processes).

PCFP can be sorted into plastic film bales for mechanical recycling in North America (Pressley et al., 2015; Tanguay-Rioux et al., 2022; Kessler Consulting, 2009), Europe (Picuno et al., 2021; Kleinhans et al., 2021a; Antonopoulos et al., 2021), and Asia (Wang et al., 2020; Nakatani et al., 2017; Kawai et al., 2022). In the United States and Canada, flexible packaging waste is sorted into a mixed plastic film bales using drum screens, ballistic separators, and optical sorters for recycling (Tanguay-Rioux et al., 2022; Kessler Consulting, 2009; Pressley et al., 2015). Some improvements are also being explored by sorting flexible packaging further by optical sorters (Materials Recovery for the Future, 2020). Similarly, PCFP is sorted into mixed plastic film bales using a series of mechanical sorting equipment for recycling in Japan and China (Kawai et al., 2022; Nakatani et al., 2017; Wang et al., 2020). In Europe, similar processes are employed to sort and recycle PCFP, which is elaborated in the next section.

4.1.2 Flexible packaging waste management from households in Europe

In Europe, plastic packaging accounted for 40% of the total plastics demand in 2019, which is equivalent to ~20 million tonnes of rigid and flexible packaging. It is estimated that out of 20 million tonnes plastics packaging, nearly 9 million tonnes are flexible plastics (consumer and industrial plastics), in which ~3.7 million tonnes become PCFP (KIDV, 2020; PlasticsEurope, 2021; Eunomia, 2020). Flexible plastics, also referred as films, includes bags, pouches, envelopes, sachets, and wraps which are widely used as consumer packaging with the main function to ensure proper product delivery to the customers. This type of packaging can provide excellent barriers against aspects such as microorganisms, light, oxygen, carbon dioxide, and water vapor, which increases the shelf life of the product and reduce (food) waste

(KIDV, 2020; Hou et al., 2018; Wagner & Marks, 2010). The market for flexible plastics is also continuously growing due to its low cost, versatility, light weight, resistance, and printability (Ügdüler et al., 2021; Horodytska et al., 2018). The main polymers for PCFP are the polyolefins (PO) low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), and polypropylene (PP) (CEFLEX, 2020; Horodytska et al., 2018; Faraca & Astrup, 2019). Flexible plastics can be produced from a single component or can be multi-layered, consisting of different material types such as polymers (e.g., PO, Polyethylene Terephthalate (PET), Polyamide (PA), Ethylene Vinyl Alcohol (EVOH), …), paper, aluminum, or any combination of these. These films may be transparent, printed, coated and/or laminated. It is estimated that around 20% (750 kilotonne) of the total PCFP are multi-material and between 70 – 80% (~3 million tonne) are reported as mono-material PO packaging (CEFLEX, 2020; KIDV, 2020; Faraca & Astrup, 2019).

To enable a circular economy for plastics, including PCFP waste, the European Union (EU) has set a target to recycle 55% of plastics packaging waste by 2030 (European Commission, 2018c). To realize this ambition, the Commission has recently launched the Circular Plastics Alliance (CPA) to support and boost the EU market for recycled plastics to 10 million tonnes by 2025 (European Commission, 2022b; 2020b). The EU has also recently passed a tax on plastics waste that charges €800 per tonne of non-recycled plastic packaging waste (European Commission, 2020b). However, up to now most of the waste management infrastructure is developed to process rigid plastics (e.g., HDPE and PET bottles). In many countries, PCFP is still not correctly source separated and is usually sent to landfill or incineration, or is exported (Lopez-Aguilar et al., 2022). In Europe, only in a limited number of countries, for example in The Netherlands, Germany, and recently Belgium, PCFP is source separated typically together with rigid plastics packaging, metals, and beverage carton (e.g., in P+MD system in Belgium). When source separated, PCFP are sent for sorting and recycling (Kleinhans et al., 2021a; Picuno et al., 2021; Horodytska et al., 2018; Brouwer et al., 2018).

A typical PCFP waste treatment can be seen in Figure 4.1. Correctly source separated PCFP waste are sent to material recovery facilities (MRFs) for sorting. At MRFs, PCFP are sorted into different bales for recycling. Typical bales for PCFP waste are: (i) *bale rich in PE film* and (ii) *bale rich in PO film*, in which >70% of PCFP is forwarded to these two bales at MRF level (Antonopoulos et al., 2021; Kleinhans et al., 2021a; Picuno et al., 2021; Mastellone et al., 2017;

Cimpan et al., 2016). One of the standards used for bales specification is ‘Duales System Deutschland (DSD) GmbH’, which is commonly used and accepted specifications to benchmark quality of the sorted bales in Europe (Der Grüne Punkt – Duales System Deutschland GmbH, 2018). Using this standard, hereafter the bale rich in PE film and PO film is referred as DSD 310-1 and DSD 323-2 bales, respectively.

Thereafter, the two sorted bales are sent to recycling facilities to be reprocessed into regranulates, in which the ‘r’ is added to the nomenclature referring to different regranulate types (Figure 4.1). Mechanical recycling is to date the most commonly used technique to process PCFP waste. A typical conventional mechanical recycling (will be called ‘conventional recycling’ from hereafter) of DSD 310-1 bale consists of shredding, additional (yet limited) optical sorting based on near-infrared (NIR) technology, washing, density separation, mechanical and thermal drying, and extrusion (Ragaert et al., 2017; Faraca & Astrup, 2019; Brouwer et al., 2018; Faraca et al., 2019; Civancik-Uslu et al. 2021). In the conventional recycling, PCFP waste is shredded into flakes and washed to remove the contaminants such as organic remnants, wood, rocks, sand, and metals. In the water-based medium, PO will mostly float and the other materials will mostly sink. After being dried using mechanical and thermal drying, PCFP waste undergoes a final regranulation step by extrusion (Ragaert et al., 2017; Larrain et al., 2021). Regranulation of PCFP flakes is usually processed at 180 – 220°C, which would remove the remaining paper, woods, metals, and polymers with higher melting temperature (e.g., PET) by a melt filter (Stenvall et al., 2013). The process yield of a typical conventional recycling for PE and PO films ranges from 60 – 80% depending on the input quality (sorted bales) and efficiency of the recycling equipment (Picuno et al., 2021; Brouwer et al., 2018; Faraca & Astrup, 2019; Larrain et al., 2021).

However, there are still many challenges in the conventional recycling of PCFP waste. Despite recent technological innovation, regranulates from PCFP are often considered inferior to virgin plastics, partly due to inefficient sorting at MRFs, complex polymer compositions, and inadequate contamination removal. For example miscibility can become a problem upon extrusion despite the structural similarities of PO (Ragaert et al., 2017 and Van Belle et al., 2020). Next to that, a noticeable amount of odours remains even after washing and separation processes leading to recycling issues, which must be removed from the stream to allow more closed-loop recycling (Roosen et al., 2021; Demets et al., 2020; Alvarado Chacon et al., 2020;

Mumbach et al., 2019). These challenges result in the fact that the part of PCFP which is recycled often finds its way to applications such as park benches or garden furniture, and not to flexible plastics again. Alternatively it is mixed together with virgin or commercial and industrial (C&I) recycled plastics to produce products such as garbage bags (Faraca & Astrup, 2019; Brouwer et al., 2018; Ragaert et al., 2017). The above-mentioned challenges highlight the importance of improving current mechanical recycling process to enhance the quality and to allow more market applications of PCFP regranulates.

As a step to mitigate this status quo of PCFP waste recycling, an improved mechanical recycling process (Figure 4.1) for PCFP waste is proposed by CEFLEX, called the *quality recycling process* (QRP), that consists of *additional sorting* and either *Tier 1* or *Tier 2 recycling* processes (Mosora, 2020). QRP can be perceived as a more elaborated route to the current conventional recycling process. As shown in Figure 4.1, QRP could start from DSD 310-1 and DSD 323-2 bales created at MRFs, in which QRP adds additional sorting (called *QRP additional sorting* in this research) to these bales prior to the actual recycling process (either *Tier 1* or *Tier 2* recycling). The QRP additional sorting creates intermediate bales: PE Film Natural, PE Flex, PP Film, and PO New bales, which can be processed later either through Tier 1 or Tier 2 recycling of QRP, depending on the targeted market applications. More information on QRP can be found in section 4.2.2.3.

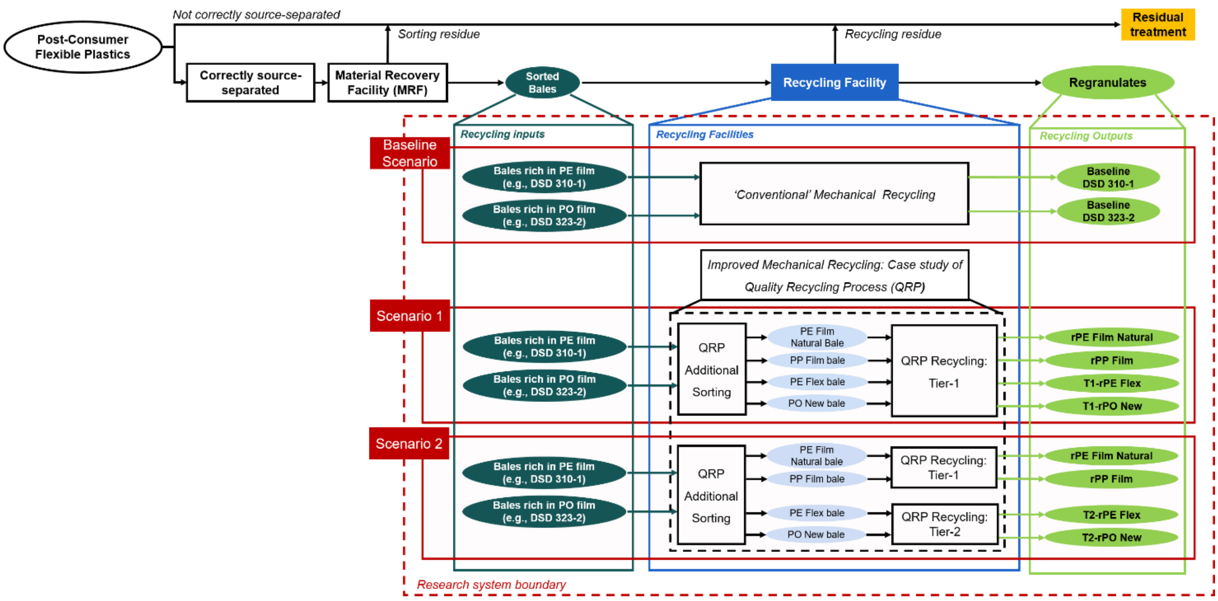


Figure 4.1 A conceptual figure depicting PCFP waste treatment in Europe (adapted from Picuno et al., 2021; Antonopolous et al., 2021; Kleinhans et al., 2021a; CEFLEX, 2020; Horodytska et al., 2018; Brouwer et al., 2018).

To assess the performance of different recycling processes, material flow analysis (MFA) and performance indicators are often used. The outputs of MFA are compositional data and mass balances, which are often linked to define the performance of the studied systems based on quantity or quality indicators (Kleinhans et al., 2021a). *Quantity indicators* refer to the amount of valuable products created from sorting or recycling facilities and the amount of recovered material (Roosen et al., 2022; Kleinhans et al., 2021a). *Quality indicators* refer to the compositions and potentially technical processability (Demets et al., 2021), technical properties (Demets et al., 2021; Alvarado Chacon et al., 2020; Grant et al., 2020; Eriksen & Astrup, 2019), or functionality or circularity potential of the regranulates (Eriksen & Astrup, 2019; Eriksen et al., 2018; Vadenbo et al., 2016). These studies also link the contamination level (i.e., non-polymer and undesired polymer content) to reflect regranulates quality. However, previous studies are mainly done for rigid plastics waste recycling and their associated quantity and quality (Alvarado Chacon et al., 2020; Eriksen & Astrup, 2019; Grant et al., 2020), whilst research into PCFP waste recycling performance is scarce (Horodytska et al., 2018).

To obtain higher recycling rates, next to the technical challenges, there are several economic concerns as well. To date, the technology and economic assessments of packaging waste treatment mainly focus on MRFs (Cimpan et al., 2016; Da Cruz et al., 2014; 2012; Marques et al., 2014) and research on the actual mechanical recycling, especially for PCFP waste, is scarce. The results of these studies on the economic structure for household waste collection and sorting at MRFs indicate that most of the costs should be supported by an Extended Producer Responsibility (EPR) scheme, e.g., gate fees. This demonstrates that the revenues from their sales are lower than the cost incurred by collection and sorting of household packaging waste. One of the costs related to recycling of Polyethylene (PE) films is the price of the feedstock that can range from positive to negative values depending on the quality of the bales created at MRFs (Cimpan et al., 2016; Larrain et al., 2021). The current price of regranulates is significantly lower for films compared to other rigid polymers, mainly due to high contamination of bales. Literature highlights the need to increase high quality regranulates production from the collected PCFP (Brouwer et al., 2018; Faraca and Astrup, 2019; Ragaert et al., 2017). In this context, an improved mechanical recycling process such as QRP can be deployed to recycled PCFP and produce higher value regranulates to be used as flexible plastic again (incl. other end market applications too such as rigid applications).

Therefore, this research investigates the recycling performance (from technical perspectives) and economic balance of conventional and improved mechanical of PCFP waste recycling by using QRP as a case study. A mathematical model is developed and applied that is based on a modular material flow analysis (MFA) approach, which is expanded from the MFA sorting model developed and validated by Kleinhans et al. (2021a). Thereafter, the MFA results are used for economic assessment, followed by the estimation of capital investment and annual costs of PCFP waste recycling through QRP. The first part of this paper focuses on describing the development of the MFA model. The inputs for the model are: experimental data (i.e., pilot trials and bales sampling at a sorting test facility), expert judgment, and literature. In the second part of this paper, the developed model is applied to trace the flow of wastes from the selected bales throughout QRP and compared to the conventional recycling. The third part of the paper assesses QRP and conventional recycling performance by applying selected performance indicators to the model outputs. Four performance indicators related to quantity (*process yield* and *net recovery*) and quality (*polymer grade* and *transparency grade*) are used to compare the results. Moreover, the compositional data produced by the MFA model (called 'modeled compositional data' hereafter) at the flakes and regranulates levels is compared with experimentally obtained compositional analyses (i.e., compositional analysis of flake and regranulate of the actual samples) for model validation. The last part of this research investigates the economic performance of QRP by means of quantifying the difference between process costs incurred by QRP and revenue generated from regranulates sales. The data points for economic assessment are collected from literature (Cimpan et al., 2016; Larrain et al., 2021), machine builders' specifications, and expert judgment. Note that the evaluation of the technical properties of regranulates is out of scope of this study and is investigated by Bashirgonbadi et al. (2022).

4.2 MATERIALS AND METHODS

4.2.1 Overview of material flow analysis model and economic assessment

An overview of the material flow modeling procedure of QRP and conventional recycling is presented in Figure 4.2. The needed inputs for the MFA modeling are the waste composition of the bales (section 4.2.2.2), a defined plant configuration of QRP and conventional recycling process at equipment level (section 4.2.2.3), and the associated separation efficiency of these equipment (section 4.2.2.4). To quantify these parameters, we have used three data source: experimental data, expert judgment, and literature sources.

The model outputs are mass balances and compositional data, which is converted to a selection of performance indicators adapted from previous studies (Roosen et al., 2022; Kleinhans et al., 2021a), which are described in section 4.2.3. To validate the model outputs, experimental compositional analyses of flakes and regranulate samples are compared with the modeled compositional data (section 4.2.3). Finally, the economic assessment of PCFP waste is assessed by combining the MFA results (mass balance) with capital investment and energy usage needed to estimate the total CAPEX and OPEX (total annual costs) for QRP. (section 4.2.4).

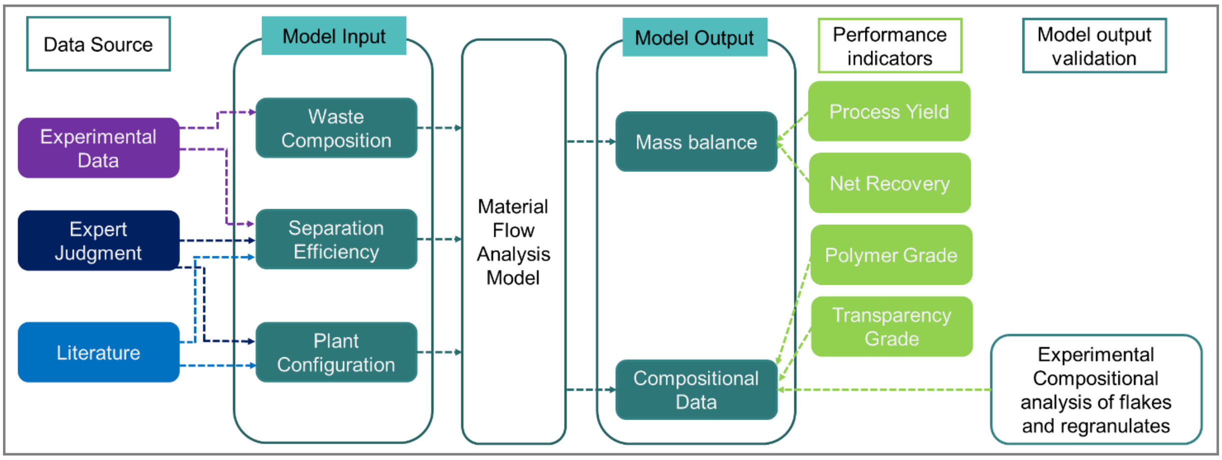


Figure 4.2 A diagram that summarizes the MFA model to assess the recycling performance (from technical perspectives) of QRP, including the required model inputs and generated outputs.

4.2.2 Material flow analysis model

4.2.2.1 Description of study area and scenarios

This research is conducted with an assumption that QRP would be installed (but not limited to) in Europe, thus assumptions and modeling parameters are linked to the European data. The starting point of the research is DSD 310-1 and DSD 323-2 bales created at MRFs, while the end point is regranulates (an '*r*' is added to the nomenclature, e.g., rPP Film refers to PP Film regranulate). In this research, the bales are products from a German MRF which is transported to a sorting test facility in The Netherlands for compositional analysis and pilot sorting trials (more in section 4.2.2.2). Based on expert judgment, all created DSD 310-1 bale and approximately 30% of DSD 323-2 bales are currently processed in Europe through the conventional recycling process. The remaining 70% of DSD 323-2 bale is exported outside Europe. However, for the purpose of a fair comparison by the MFA, in this research it is assumed that all DSD 323-2 bale would be processed in Europe.

This research focuses on three mechanical recycling scenarios. *Baseline scenario*: conventional recycling. In the baseline scenario, DSD 310-1 and DSD 323-2 bales are processed through conventional recycling process to produce regranulates. *Scenario 1*: QRP with Tier 1 recycling. In scenario 1, DSD 310-1 and DSD 323-2 bales are processed through QRP additional sorting. Thereafter, all created QRP bales are processed through Tier 1 recycling to produce regranulates. *Scenario 2*: QRP with Tier 1 and Tier 2 recycling. Based on expert judgment, and as demonstrated by Bashirgonbadi et al. (2022), Tier 1 and Tier 2 recycling produces regranulates for different market applications because of the differences in regranulates quality. Therefore, in scenario 2, the products of the QRP additional sorting of DSD 310-1 and DSD 323-2 bales, namely PE Film Natural and PP Film bales are processed through Tier 1 recycling, while PE Flex and PO New bales are processed through Tier 2 recycling. Detailed information on the plant configuration of QRP and conventional recycling can be found in section 4.2.2.3.

4.2.2.2 Waste input composition to MFA model: DSD 310-1 and DSD 323-2 bales

The compositional analysis of DSD 310-1 and DSD 323-2 bales is conducted at the Nationaal Testcentrum Circulaire Plastics (NTCP) sorting test facility in Heerenveen – The Netherlands. Seven DSD 310-1 (approx. 3.6 tonnes) and six DSD 323-2 bales (approx. 3.5 tonnes) are transported from a German MRF in Autumn 2020. There are three different level of waste classification used in this research: *Main group*, *sub-group*, and *sub-category* levels, as shown in Table 4.1.More detailed information on the sampling procedures can be found in Appendix C – section 1.

Table 4.1 Waste composition of bales rich in PE film (e.g., DSD 310-1) and PO film (e.g., DSD 323-2). The sampling was done at a sorting test facility in The Netherlands, based on the bales from a German MRF in Autumn 2020.

Waste categories			**Waste Composition (shown in %)**	
Main Group	Sub-Group	Sub-Category	Bale rich in PE film (e.g., DSD 310 – 1)	Bale rich in PO film (e.g., DSD 323 – 2)
PE Films	PE Film Transparent Clear		48.4	23.9
	PE Film Others	Transparent clear printed	8.0	2.4
		Transparent coloured	6.6	2.4
		Opaque coloured	7.6	4.2
		Black	3.1	1.4
		Metalized	0.7	0.4
PP Films	PP Film Transparent	Transparent clear	1.4	5.3
		Transparent clear printed	0.9	5.0
		Transparent coloured	0.1	0.4
	PP Film others	Opaque coloured	0.6	3.3
		Black	0.5	0.4
		Metalized	0.2	2.3
Other-film		Plastic	1.1	0.1
Multi-material Film		Aluminium laminate	0.8	1.5
		Paper laminate	0.3	0.6
		Other laminate	4.3	4.7
PE-Rigid			1.8	0.7
PP-Rigid			1.3	10.8
Other Plastics Film		Textile, fabric	0.8	1.2
		Nets	0.7	1.2
		Foamed	1.1	1.8
Paper		Print, cardboard	1.2	2.8
		Hygiene tissue	0.3	0.9
Residue		Compound	0.0	1.2
		Clogged	3.3	8.0
		Others	0.0	1.8
Fines			4.8	11.2
Total			**100**	**100**

4.2.2.3 Mechanical recycling plant configuration

Experts from the industry are involved in the development of the conventional recycling and QRP configuration (Figure 4.3), i.e., HTP GmbH & Co. KG develop the process flow diagram with subsequent consultation with waste management operators and equipment manufacturers such as Attero B.V., EREMA Group GmbH, and Herbold Meckesheim GmbH. Based on expert judgment and previous study by Larrain et al. (2021), plastics recycling plants can operate at 20,000 tonnes/year and 7,000 hours/year, equivalent to 3 tonnes/hour processing capacity. This capacity is used in our model: 20,000 tonnes/year of DSD 310-1 and 20,000 tonnes/year DSD 323-2 mass input for both QRP and conventional recycling.

4.2.2.3.1 Conventional recycling

The flow diagram of conventional recycling line of DSD 310-1 and DSD 323-2 bales can be found in Figure 4.3a. The recycling process starts by shredding the materials into roughly 10 cm in size. For DSD 310-1, metals and non-PE are typically removed before washing using overbelt magnet and NIR sorters. For DSD 323-2, metals and non-PO materials are removed only during washing, density separation, and extrusion.

The cold washing consists of washing by water at 25 – 40˚C, wet granulation, and friction washer. In the cold washing, the materials will be further size-reduced into flakes sized roughly 1 cm. Contaminations, such as organic residues, paper and labels, are further removed by a friction washer, in which a high-speed screw is used to remove contaminants by centrifugal forces. Thereafter, the remaining heavy polymers and metals are removed in a density separation bath. Before extrusion, the materials are dried using mechanical and thermal drying to remove moisture. Lastly, a single melt filter extruder is used to remove some of the remaining contaminants (Larrain et al., 2021; Faraca & Astrup, 2019; Faraca et al., 2019; Horodytska et al., 2018; Brouwer et al., 2018; Ragaert et al., 2017). In this research, the regranulates produced from the conventional recycling DSD 310-1 and DSD 323-1 bales are referred as "*Baseline DSD 310-1*" and "*Baseline DSD 323-2*", respectively.

4.2.2.3.2 Quality recycling process (QRP)

QRP consists of *QRP additional sorting* and *Tier 1* or *Tier 2* recycling depending on the targeted regranulate quality.

QRP Additional sorting. It is assumed that QRP additional sorting sorts 20,000 tonnes/year DSD 310-1 and 20,000 tonnes/year DSD 323-2 bales. Both bales are processed separately in two different additional sorting lines (per bale) working in parallel after debaling (see Figure 4.3b). The QRP additional sorting starts by overbelt magnets and fine screens separation, removing magnetic material and fine residue, respectively. Thereafter, NIR-based optical sorters are used: i) NIR-VIS LDPE Natural, positively sorts PE film transparent clear, ii) NIR PE Cleaner, negatively sorts non PE materials, iii) NIR PP Film, positively sorts PP films, iv) NIR PP

Cleaner, negatively sorts non PP materials, and v) NIR PO Cleaner, negatively sorts non PO materials.

From DSD 310-1, the QRP additional sorting aims to sort PE film transparent clear, as indicated in Table 4.1. Next to this, a second bale is created consisting of the colored/printed PE films that are present in the DSD 310-1 bale. For these purpose, the QRP additional sorting uses optical NIR-VIS LDPE Natural sorters to 'positively' targeting PE film transparent clear and next cleaning the stream by 'negatively' removing all non-PE items using NIR LDPE Cleaner. The bale rich in PE film transparent clear is called "*PE Film Natural*" and bales rich in all colors PE films is called "*PE Flex*".

From DSD 323-2 bale, the QRP additional sorting 'positively' sorts PP films (transparent and coloured) by NIR PP Film sorters. Thereafter, the stream is cleaned by 'negatively' removing non-PP items using NIR PP Cleaner sorter, creating a bale rich in PP called "*PP Film*". The non-PP fraction of the DSD 323-2 bale is cleaned from the non-PO materials using NIR PO Cleaner sorters, creating a bale called "*PO New*".

QRP Tier 1 and Tier 2 recycling. The four bales of the QRP additional sorting are sent to the QRP recycling, in which the bales will be shredded, washed, and extruded. Two recycling lines can be used in QRP: Tier 1 (Figure 4.3c) and Tier 2 recycling lines (Figure 4.3d). The Tier 1 and Tier 2 recycling both start with shredding, followed by cold washing (i.e. identical to cold washing process in the conventional recycling, section 4.2.2.3.1). In the case of Tier 1 recycling, an additional hot washing step is applied to remove more odours, papers, and adhesives from the waste stream. The recycling process ends with an extrusion process including a two steps filter with degassing and deodorization (hot air-based treatments) for Tier 1 and a single step filter with degassing (without deodorization) for Tier 2 recycling. In this research, the regranulates produced from QRP Tier 1 is referred as "*rPE Film Natural*", "*T1-rPE Flex*", "*rPP Film*", and "*T1-rPO New*", while the regranulates produced from QRP Tier 2 is referred as "*T2-rPE Flex*" and "*T2-rPO New*".

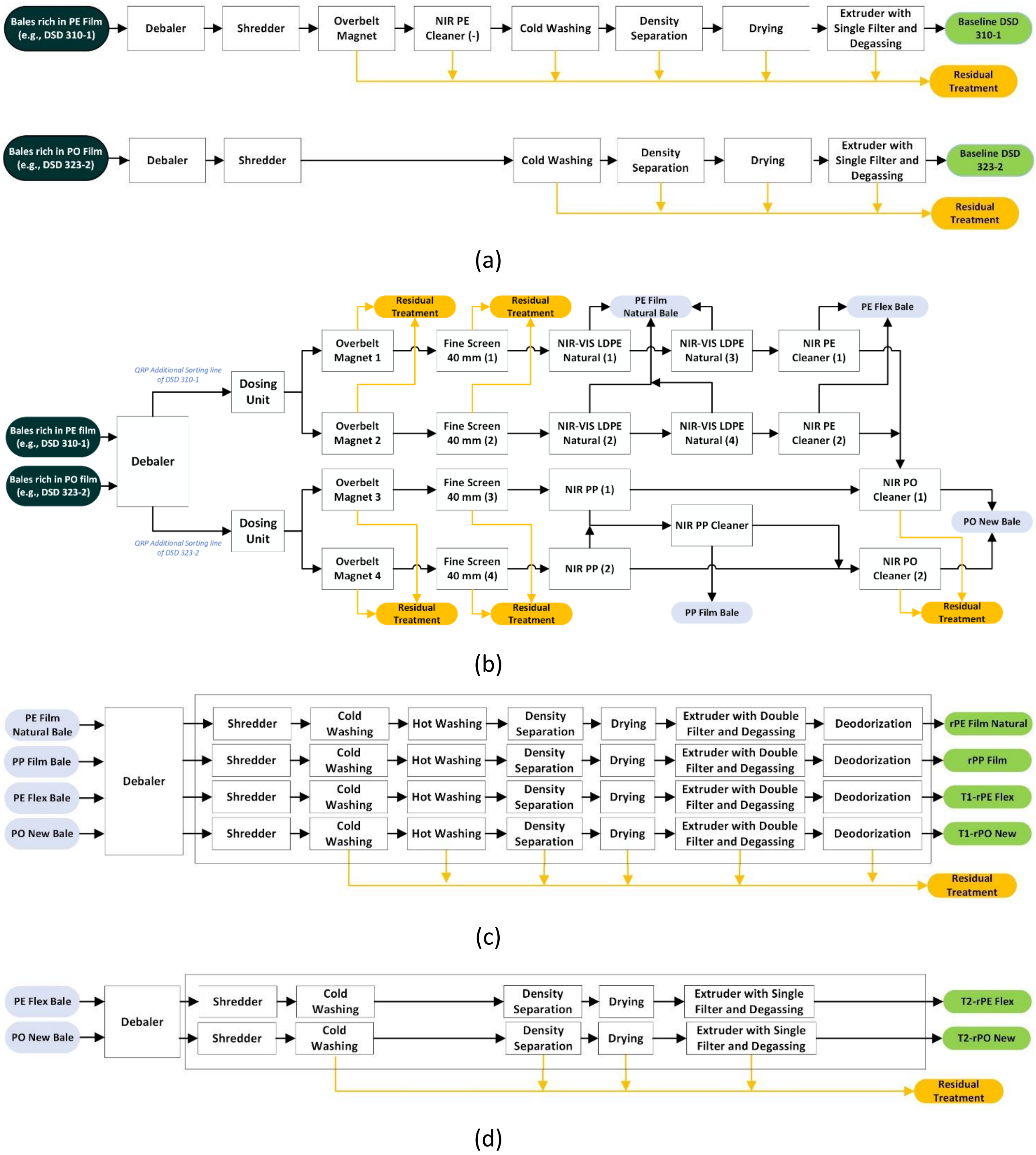


Figure 4.3 Process flow diagrams of (a) conventional recycling, (b) QRP additional sorting, (c) QRP Tier 1 and (d) QRP Tier 2 recycling (adapted from CEFLEX, 2021).

4.2.2.4 Separation efficiency

The modular MFA approach is based on separation efficiencies to predict material flows throughout the described plant configurations. The separation efficiency (expressed in %) represents the separation of the waste items (categories) at each sorting or recycling unit, which is called transfer coefficient or split factors in other studies (Kleinhans et al., 2021a; Brouwer et al., 2018; Cimpan et al., 2016, Mastellone et al., 2017). This includes positive and negative separations as well as potential missorting.

Data on separation efficiencies is collected from relevant literature from plastic waste treatment in Europe (Kleinhans et al. 2021a; Brouwer et al. 2018; Cimpan et al. 2016) combined with pilot trials performed in the frame of the CEFLEX activities. Experts from industry are interviewed to further validate the separation efficiencies used in the MFA model. The separation efficiency for every waste category (Table 4.1) at every optical NIR sorter is determined by performing a mass balance analysis before and after sorting. During the trials, a few sampling points are defined in which the outputs of optical sorting are collected into three bags. Two out of three bags are chosen and the materials are mixed on the floor. Later, approximately 7.5 kg of sample is randomly collected for material composition characterization and used as the basis for the separation efficiency determination. Detailed procedures of material sampling protocol can be found in Appendix C – section 1.

The separation efficiencies of sorting equipment used in MFA modeling can be found in Appendix C – Table C.2. Further information regarding the quantification of separation efficiency based on mass balances and detailed values per waste category at different NIRs can be found in the Appendix C – section 1. Additionally, the separation efficiency of the overbelt magnet is obtained from the study of Kleinhans et al. (2021a), while the fines fraction (< 40 mm in size) can be removed effectively (60%) in the fine screen.

The aggregated separation efficiency for recycling equipment used in the conventional recycling, QRP Tier 1 and Tier recycling can also be found in Appendix C, Table C.4. It is assumed that up to 95% of the organic residue, papers, and fine fraction remnants can be removed after the cold and hot washing. Under elevated temperature of >80^{o}C and by adding washing agents such as caustic soda and detergents, the hot washing step can effectively remove adhesive and organic waste residues as well as partially remove some of the inks. More than 95% of low-density monolayer PO floats and most of the heavier materials (70%) such as non-PO based (other) laminated films, metallic materials, fibers, and other rigid polymers are assumed to sink in the density separation. As for the metalized PE / PP and other films listed in Table 4.1, it is assumed that 70% floats and 30% sinks. After water-based washing process, the materials are dried using thermal and mechanical drying, removing more than 97% of the moisture content. Later, in the extrusion process, materials and polymers with higher melting points (higher than 200°C) are assumed to be retained at the extrusion filter with the sieve size of 90 – 110 µm (plus an additional 125 µm for two steps filtration technology). The efficiency of the melt filter extrusion process is relatively high (95%) towards non-PO flexible materials. Lastly, the level of odors are reduced by 55% - 90% in the deodorization process (Roosen et al., 2021; Larrain et al., 2021; Picuno et al., 2021; Demets et al., 2020; Strangl et al., 2020; Faraca & Astrup, 2019; Strangl et al., 2019; Brouwer et al., 2018).

4.2.3 Assessment of the recycling performance

To facilitate interpretation and allow proper comparison of different scenarios, two indicators related to quantity: (i) *process yield* (ii) *net recovery* and two indicators related to quality: (iii) *polymer grade* (iv) *transparency grade* are used, adapted from previous studies (Roosen et al., 2022; Kleinhans et al., 2021a). Afterwards, the modeled composition at flakes and regranulate levels produced by the MFA model is compared with the experimental compositional analyses from the flake and regranulate samples.

4.2.3.1 Performance indicators

The summary of the selected quantity and quality indicators can be found in Table 4.2 and is illustrated in Figure 4.4. The *process yield* (*Y*) measures the share of total waste entering a recycling facility (μ^{I}) that is finally converted into regranulates (μ^{r}). The *net recovery* (*R*) indicates the fraction of waste *T* entering a recycling facility ($\mu_{\mathrm{I}}^{\mathrm{T}}$) that is found in the correct regranulates. In Appendix C Table C.4, the targeted regranulates ($\mathrm{f}_{\mathrm{regranulate}}^{\mathrm{T}}$) for all listed waste categories (listed in Table 4.1) can be found.

The *polymer grade* at bales ($\mathrm{G_b}$), flakes ($\mathrm{G_f}$), and regranulates ($\mathrm{G_r}$) level is a simple proxy to measure the quality of the products as it reflects the *concentration of the PE and/or PP* (films and/or rigid) at QRP bales, flakes and regranulates level over the total mass of all materials in the respective product, i.e., total mass of all materials in bale ($\mathrm{f}_{\mathrm{bale}}^{\mathrm{m}}$), flakes ($\mathrm{f}_{\mathrm{flakes}}^{\mathrm{m}}$) and regranulates ($\mathrm{f}_{\mathrm{regranulate}}^{\mathrm{m}}$) level. The targeted materials at bales ($\mathrm{f}_{\mathrm{bale}}^{\mathrm{T}}$), flakes ($\mathrm{f}_{\mathrm{flakes}}^{\mathrm{T}}$), and regranulates ($\mathrm{f}_{\mathrm{regranulate}}^{\mathrm{T}}$) level for PE Film Natural, PE Flex and Baseline DSD 310-1 (from conventional recycling) are all PE from the DSD 310-1 bale. The PP Film targets all PP, while PO New and Baseline DSD 323-2 (from conventional recycling) target all PO materials from

DSD 323-2. Moreover, an indicator as a proxy to measure the quality of the color of the regranulates is added, called *transparency grade* indicator (t_r), which indicates the *concentration of transparent film* in the regranulates ($f_{regranulate}^{t_{film}}$),i.e., concentration of *PE film transparent clear* or *PP film transparent* (t_{film}). The value of these indicators ranges between 0 – 100%.

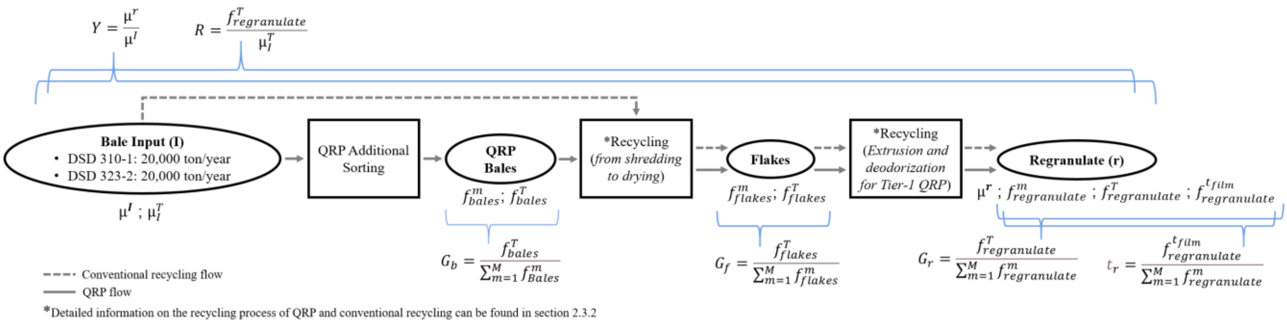


Figure 4.4 A diagram with indicated symbols used to define the selected performance indicators applied in this research.

Table 4.2 The summary of the selected indicators, their corresponding definitions and formulas applied to evaluate the performance of the recycling process, adapted from Roosen et al. (2022) and Kleinhans et al. (2021a).

Performance indicator	Definition	Equation
Quantity indicator		
Process yield	The share of mass waste input *I* (in tonne/year) that is being converted into regranulates *r* (in tonne/year)	$Y = \frac{\mu^r}{\mu^I}$
Net Recovery	Fraction of waste *T* entering recycling plant that is found in the desired regranulate *r*	$R = \frac{f^T_{regranulate}}{\mu^T_I}$
Quality indicator		
Polymer Grade		
QRP Bale grade	The concentration of the targeted waste categories *T* in the QRP bales	$G_b = \frac{f^T_{bales}}{\sum_{m=1}^{M} f^m_{Bales}}$
Flakes grade	The concentration of the targeted waste categories *T* in the flakes	$G_f = \frac{f^T_{flakes}}{\sum_{m=1}^{M} f^m_{flakes}}$
Regranulate grade	The concentration of the targeted waste categories *T* in the regranulates	$G_r = \frac{f^T_{regranulate}}{\sum_{m=1}^{M} f^m_{regranulate}}$
Transparency Grade	The concentration of transparent films (i.e., PE film natural and PP film transparent) t_{film} in the regranulates	$t_r = \frac{f^{t_{film}}_{regranulate}}{\sum_{m=1}^{M} f^m_{regranulate}}$

4.2.3.1 Compositional analysis at the flakes and regranulate level

The modeled compositional data is compared against experimental compositional analyses of the flakes and regranulates samples to validate the model. The samples were shredded, washed, and extruded according to QRP chains in the test centers, as such creating flakes and regranulates of PE Film Natural, PE Flex, PP Film, and PO New.

The composition of hot-washed flakes of PE Film Natural, PE Flex, PP Film, and PO New, is determined by Fourier-Transform Infrared Spectroscopy (FTIR). A representative sample of flakes was isolated from each fraction following a standardized mass reduction and sampling procedure CEN/TS 16010. Each sample contained 120-130 flakes. Thereafter, the polymeric composition at both sides of flakes was characterized. After weighting the flakes of a similar polymer, the overall composition could be obtained. Bruker Tensor 27 device, with OPUS 6.5 software, equipped with Attenuated Total Reflection on ZnSe-crystal, is used for the FTIR study at a resolution of 4 cm^{-1} in 16 sample scans and frequency range from 4000 to 600 cm^{-1}.

The composition of the materials at regranulate level is determined by Differential Scanning Calorimetry (DSC). In this research, six different regranulates are characterized i) rPE Film Natural, ii) rPP Film, iii) T1-rPE Flex, iv) T2-rPE Flex, v) T1-rPO New, and iv) T2-rPO New. To estimate the compositions of the regranulates, the melting enthalpies of PE and PP of the second heating cycle in these materials are compared with master curves as described by Kisiel et al. (2018). For DSC measurements 10 mg per sample is prepared and Polyma 214 device is used in two runs of 25°C to 290°C to 25°C (10K/min) in a nitrogen atmosphere. The average crystallinity of the PE and PP phases is considered to be 38% and 50%, respectively (Kisiel et al., 2018).

4.2.4 Economic assessment

In the economic assessment, a cost-benefit analysis (CBA) is performed of the recycling process by quantifying the difference between the processing costs incurred and the revenue generated from regranulates sales. The economic balance is given without external financial support, which are normally paid by the Producer Responsibility Organization (PRO) (Da Cruz et al., 2014; 2012; Marques et al., 2014). The capital investment and annual operational costs (incl. OPEX and CAPEX) associated with the recycling processes are estimated by collecting the required data points from previous studies (Cimpan et al., 2016; Larrain et al., 2021), machine builders' specifications, and expert judgment. The data points from literature are used when the information suit the investigated (improved) mechanical recycling processes in this research (e.g., machines' processing capacity), followed by consultation with the machine builders and experts from the recycling industry (members of
the CEFLEX consortium).

The approach to estimate the capital investment is adopted from Sinnott and Towler (2019), which is applied in estimating the capital investment to build plastics waste sorting and recycling plant in previous studies (Cimpan et al., 2016; Larrain et al., 2021, 2020). The calculation of the capital investment includes the equipment prices plus the additional costs associated with investing in the equipment, namely the installation of equipment costs (IC) and engineering and project management costs (EPMC). On top of these additional costs, the investment cost includes the building and construction (BC) of the actual plant itself (more in Appendix C, Table C.23 – TableC.25). The values of these parameters can be found in Table 4.3. The total annual costs (OPEX and CAPEX) is quantified by calculating the cost of energy consumption, transport, disposal fee, general expenses, direct production costs (labors, repair, and maintenance), and fixed costs (depreciation and insurance). The value for each cost parameter can be found in Appendix C, Table C.25. The assessment is based on the plant configuration, scale and material flows as presented in section 4.2.2.3. Moreover, the investment is annualized for 6 – 7 years for the processing equipment (i.e., NIR, washing equipment, dryer, extrusion, etc.) and 10 years for the plant itself.

The price of regranulates from the conventional recycling is obtained from Larrain et al. (2021). Because the projection of low density polyethylene (LDPE) and mixed polyolefin (MPO) regranulates can differ depending on the market condition, here in this research the prices of T2-rPE Flex and T2-rPO New (which are comparable to LDPE and MPO regranulates in a previous study from Larrain et al.(2021) are set to be €400 and €300 per tonne, respectively, on the basis that regranulates quality might improve with deodorization process, allowing the regranulates to be used in more demanding applications. When processed through Tier 1 recycling, the regranulate price of T1-rPE Flex and T1- rPO New is assumed to be higher, i.e. €500 and €400, respectively. Additionally, this improved recycling process creates two more bales (i.e., rPE Film Natural and rPP Film), for which prices are not yet determined in the market for post-consumer regranulates. However, as the technical properties of these regranulates are significantly improved (Bashirgonbadi et al., 2022), we assume that the price will get close to the price of virgin plastics. Hence, the price of rPE Film Natural and rPP Film is set to be €1200 and €1300 per tonne respectively (Plastic Portal EU, 2021; Plasticker, 2021).

Table 4.3 Cost modeling parameters to quantify the total capital investment and total annual costs (OPEX and CAPEX).

Cost Modeling Parameters	Value	Source
Capital investment		
Price of Equipment (PoC)	Depend on the sorting unit	Expert judgment and (Cimpan et al., 2016)
Additional Costs		
• Installation and running test (IC)	60% of PoC	Expert judgment and (Cimpan et al., 2016; Larrain et al., 2021, 2020; Sinnott and Towler, 2019)
• Engineering and project management (EPMC)	10% of PoC and 10% of IC	
• Building construction (BC)	25% of total CAPEX	
Total Capital Investment = PoC + IC + EPMC + BC		
Costs (OPEX and CAPEX)		
Labor use	1 person/kilotonne.annum processing capacity for sorting 1,5 person/kilotonne.annnum processing capacity for recycling	Expert judgment and Cimpan et al. (2016)
Labor cost	45,000 €/person.year	Averaged value (Eurostat, 2021; Larrain et al., 2021; Sinnott and Towler, 2019)
Electricity	0.10 €/kWh	EEA, (2021) and PwC (2019)
Fuel	1,310 €/m^3	
Repair and maintenance	4% * Total Capital investment	
Insurance	0.7% * Total capital investment	Cimpan et al. (2016) and Larrain et al. (2021)
Depreciation	10 – 15% * Total capital investment	
Disposal fee	140 €/tonne residual	
General expenses	10% of total cost	

4.2.5 Sensitivity analysis of material flow analysis and economic modeling parameters

As elaborated in section 4.2.2, the MFA model is developed based on separation efficiencies that are obtained from a combination of sorting trials and stakeholder consultations. In Chapter 4, the collected datasets are averaged and a single point estimate is used in MFA model (more about the trials in Appendix C). Moreover, the economic assessment is developed based on an averaged datasets from historical datasets (e.g., regranulate prices, energy costs, etc.) or specific information from the equipment manufacturers (e.g., capital investment for extruder, cold washing, hot washing, etc.). In this respect, a sensitivity analysis is performed to identify the relative importance of these separation efficiencies and economic parameters towards the MFA and CBA model results (Bisinella et al., 2016).

For the material flow analysis results, a sensitivity analysis is carried out to assess the impact of potential variations of the selected modeling parameters towards the performance indicators of QRP (i.e., process yield, net recovery, polymer grade, and transparency grade). Nine modeling parameters are varied in the sensitivity analysis such as the bale compositions, five separation efficiencies of the optical sorters, and four separation efficiencies of the recycling equipment. This approach is applied to gain insights into the most sensitive parameters. In this study, the sensitivity analysis is done by changing each individual modeling parameter by ± 25% one by one while maintaining the other parameters at a constant value. More information on the new bale compositions and separation efficiencies (±25%) can be found in the Appendix C – section 6.

For the economic assessment results, a sensitivity analysis is carried out to examine how cost modeling parameters and the price of regranulates can influence economic balance of QRP. Sensitivity analysis is done by altering each of the selected parameters (electricity cost, depreciation, and labor costs), price of regranulates, and investment of the selected recycling equipment (debaler, shredder, washing equipment, and extruder) individually by ± 25 % (detailed information is provided in Appendix C – section 7).

4.3 RESULTS AND DISCUSSION

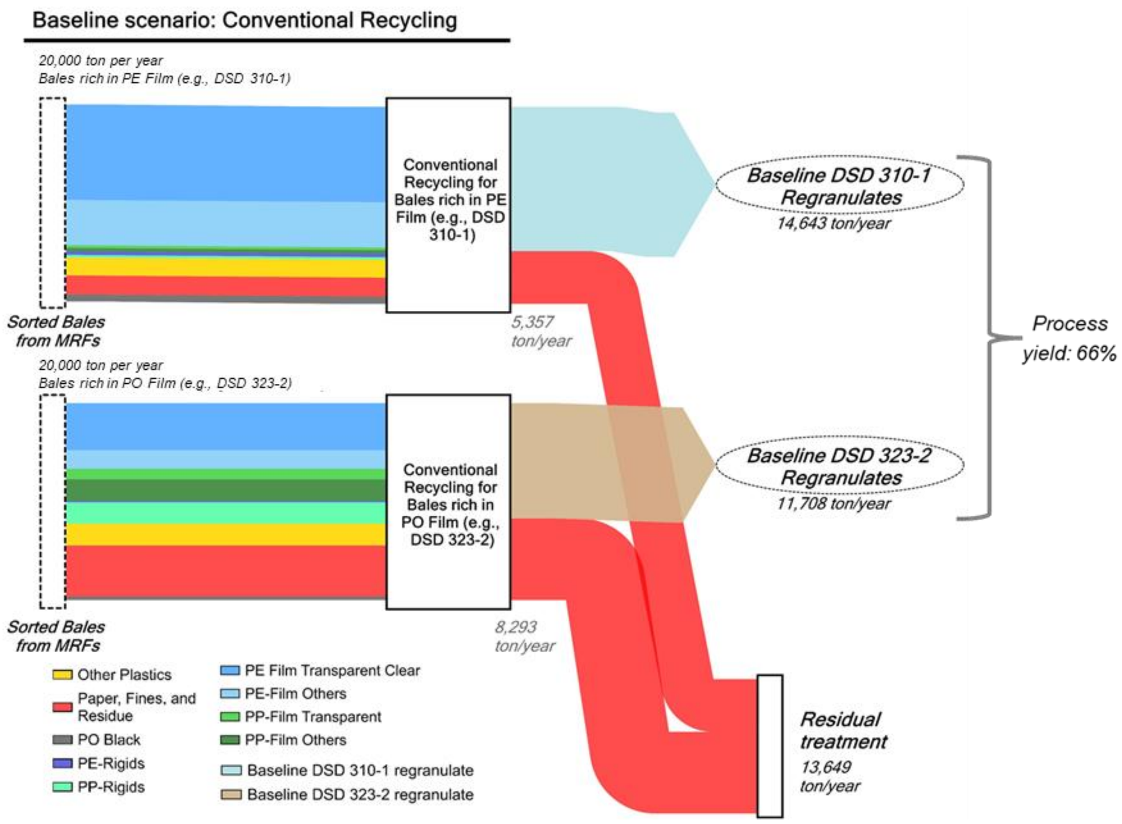
4.3.1 Material flow analysis and process yield

Figure 4.5 shows the material flows of the baseline scenario (Figure 4.5a), scenario 1 (Figure 4.5b), and scenario 2 (Figure 4.5c). From Figure 4.5, it can observed that of the total 20,000 tonnes/year of DSD 310-1 bales, 14,643 tonnes/year is converted into Baseline DSD 310-1 regranulate, while 11,708 tonnes/year Baseline DSD 323-2 regranulate is produced from 20,000 tonnes/year DSD 323-2 bales through the conventional recycling. Less regranulates are thus produced from recycling DSD 323-2 bale because of a higher degree of contamination in the input bales that accounts for more than 35% of the total mass (i.e., residue and non-PO materials). The process yield of PCFP waste recycling through conventional recycling of DSD 310-1 and DSD 323-2 bales is 66%, equal to 26,351 tonnes/year regranulate production. Previous studies suggest that Baseline DSD 310-1 regranulate would typically end-up in open-loop recycling such as garbage bags or agriculture pipes. On the other hand, the Baseline DSD

323-2 regranulate is mainly used in robust applications such as garden furniture or benches (Faraca & Astrup, 2019; Horodytska et al., 2018; Bashirgonbadi et al., 2022).

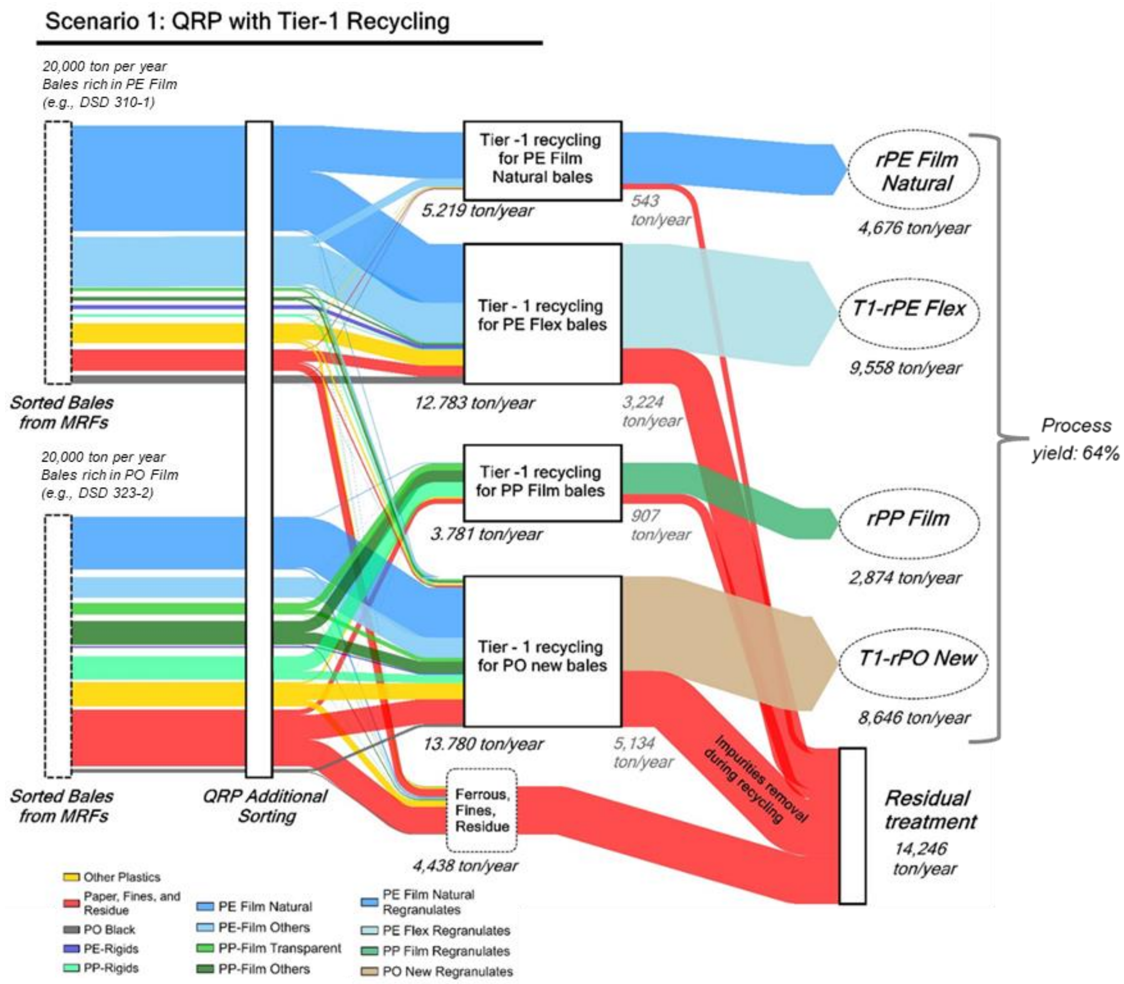
In scenario 1 and 2 of QRP, first the QRP additional sorting creates 5,219 tonnes/year PE Film Natural bales, 12,783 tonnes/year PE Flex bales, 3,781 tonnes/year PP Film bales, and 13,780 tonnes/year PO New bales. Relative to the 20,000 tonnes input of DSD 310-1, the QRP additional sorting sorts 26% of the input into PE Film Natural bales, 64% into the PE Flex bale, and 5% into the PO New bale. Furthermore, from the 20,000 tonnes of DSD 323-2, 19% is sorted into the PP Film bale whilst 64% is sorted into the PO New bale. The QRP additional sorting removes a fraction of the residue (incl. papers and fine fractions) and non-PO materials, which accounts for 11% (equal to 4,438 tonnes/year) of the total mass input.

After the QRP additional sorting, these bales go to the recycling process. Four regranulates are created in scenario 1: rPE Film Natural (4,676 tonnes/year), T1-rPE Flex (9,558 tonnes/year), rPP Film (2,874 tonnes/year), and T1-rPO New (8,646 tonnes/year). Thus, from DSD 310-1 and DSD 323-2 bales up to four regranulate types through sencario 1, the process yield is 64% (in total 25,754 tonnes/year regranulates). In scenario 2, PE Flex and PO New bales are processed through Tier 2 recycling and a slight difference can be observed. The amount of rPE Film Natural and rPP Film remain, while the production of T2-rPE Flex and T2-rPO New increases to 9,878 tonnes/year and 8,953 tonnes/year respectively. The 3% increase of regranulates production in scenario 2 can be explained by the fact that Tier 2 recycling employs less recycling equipment and consequently generates less residue. The process yield of recycling PCFP waste through scenario 2 slightly increases to 66% (in total 26,381 tonnes/year regranulate). Concluding, the process yields of recycling PCFP waste via conventional recycling and QRP are relatively similar, which is in line with the typical reported process yield in previous studies of 60%–80% (Picuno et al., 2021; Brouwer et al., 2018; Faraca & Astrup, 2019). Potential differences in composition of the regranulates will be investigated in section 4.3.3.



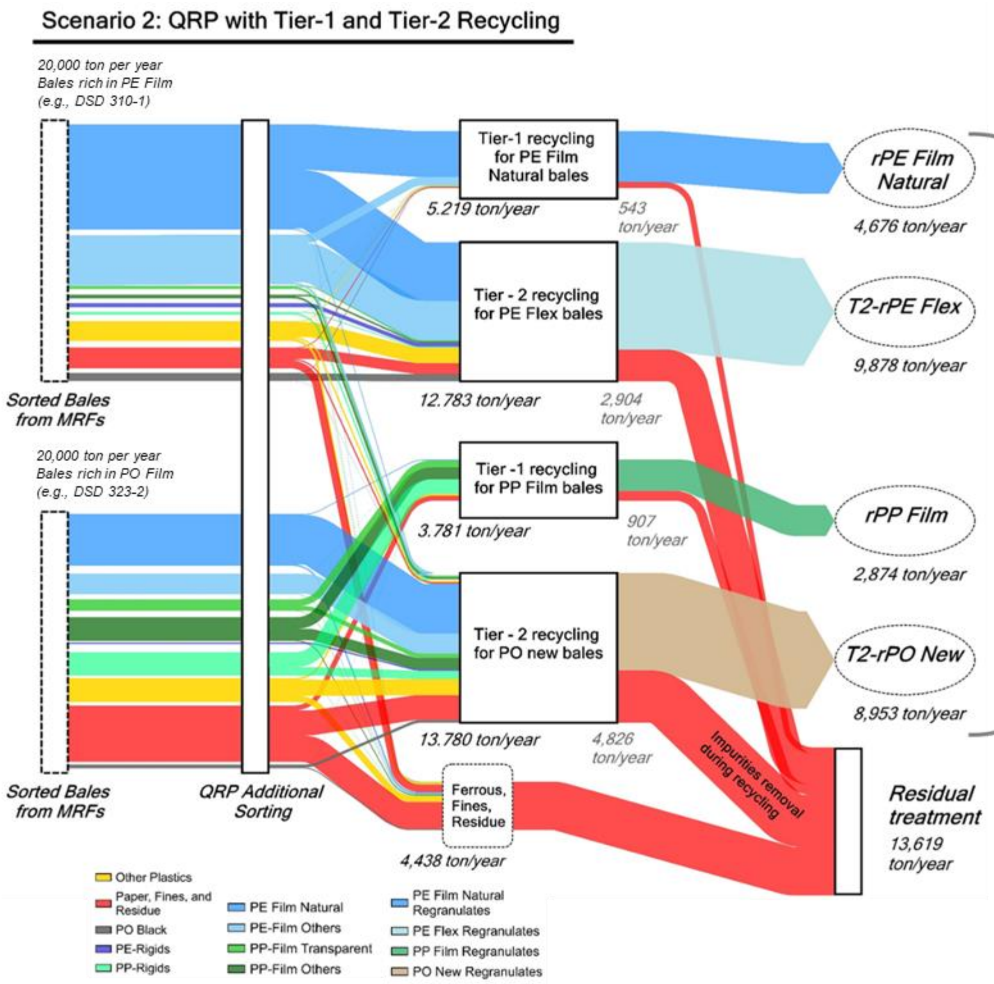
Baseline scenario: Conventional Recycling
20,000 ton per year
Bales rich in PE Film (e.g., DSD 310-1)
Conventional Recycling for Bales rich in PE Film (e.g., DSD 310-1)
Baseline DSD 310-1 Regranulates
14,643 ton/year
Sorted Bales from MRFs
5,357 ton/year
Process yield: 66%
20,000 ton per year
Bales rich in PO Film (e.g., DSD 323-2)
Conventional Recycling for Bales rich in PO Film (e.g., DSD 323-2)
Baseline DSD 323-2 Regranulates
11,708 ton/year
Sorted Bales from MRFs
8,293 ton/year
Residual treatment
13,649 ton/year
Other Plastics
Paper, Fines, and Residue
PO Black
PE-Rigids
PP-Rigids
PE Film Transparent Clear
PE-Film Others
PP-Film Transparent
PP-Film Others
Baseline DSD 310-1 regranulate
Baseline DSD 323-2 regranulate

(a)



Scenario 1: QRP with Tier-1 Recycling
20,000 ton per year
Bales rich in PE Film
(e.g., DSD 310-1)
Sorted Bales from MRFs
20,000 ton per year
Bales rich in PO Film
(e.g., DSD 323-2)
Sorted Bales from MRFs
QRP Additional Sorting
Tier -1 recycling for PE Film Natural bales
5.219 ton/year
543 ton/year
Tier - 1 recycling for PE Flex bales
12.783 ton/year
3,224 ton/year
Tier -1 recycling for PP Film bales
3.781 ton/year
907 ton/year
Tier - 1 recycling for PO new bales
13.780 ton/year
5,134 ton/year
Ferrous, Fines, Residue
4,438 ton/year
Impurities removal during recycling
rPE Film Natural
4,676 ton/year
T1-rPE Flex
9,558 ton/year
rPP Film
2,874 ton/year
T1-rPO New
8,646 ton/year
Process yield: 64%
Residual treatment
14,246 ton/year
Other Plastics
Paper, Fines, and Residue
PO Black
PE-Rigids
PP-Rigids
PE Film Natural
PE-Film Others
PP-Film Transparent
PP-Film Others
PE Film Natural Regranulates
PE Flex Regranulates
PP Film Regranulates
PO New Regranulates

(b)



(c)

Figure 4.5 The material flow of aggregated waste category from bales rich in PE film (e.g., DSD 310-1) and PO film (e.g., DSD 323-2) through (a) conventional recycling (b) QRP where all four regranulates are produced from Tier 1 recycling (c) QRP where rPE Film Natural and rPP Film are produced from Tier 1 whilst rPE Flex and rPO New are produced from Tier 2 recycling.

4.3.2 Net recovery

The estimated net recovery of PE film transparent clear, PE film others (i.e., colored/printed PE films), PP film transparent, PP film others (i.e., colored/printed PP films), PE rigid and PP rigid waste (aggregated in the *sub-group* level, see Table 4.1) from DSD 310-1 and DSD 323-2 through conventional recycling and two QRP scenarios can be found in the Figure 4.6.

It can be observed that the net recovery of all presented waste categories (shown in aggregated *sub-group* level, see Table 4.1) is always higher in scenario 2 compared to scenario

1 of QRP, because the scenario 2 processes PE Flex and PO New bales through Tier 2. This finding is expected because it is unavoidable that hot washing and extrusion with extra filtration also remove a small fraction of PE and PP.

However, when we compare QRP with the conventional recycling, the differences in net recovery range between 1 to 10%. Little difference can be observed on the recovery rate of PE film transparent clear in QRP (91%–93%) compared to the conventional recycling (94%). As for the PE film others, we can observe a drop of net recovery from the conventional recycling (92%) to QRP (79%–81%). For, PP film transparent the net recovery increases from 79% in the conventional recycling to 82%–85% in QRP. While, the net recovery of PP film others slightly drops from 57% in the conventional recycling to 51%–52% in QRP. However, PP film transparent and PP film others end up in a separate PP regranulate type, whereas in the conventional recycling these materials end up in Baseline DSD 323-2, which is a mixed PO regranulate type. As for the two other fractions, we can note little differences (by a margin of 1%–4%) between the conventional recycling and QRP.

Amongst PCFP waste, a relatively higher net recovery rates can be observed for PE film transparent clear (> 90%) and PP film transparent (> 80%). One of the reasons for a relatively higher values for the two PCFP waste items is the fact that these waste items fall under these waste categories are usually found as monolayer films. One of the additional advantages of such mono PE or PP flexible packaging types in recycling, next to the potential compatibility issues later in the regranulate, is that they also float more effectively in the density separation tank compared to multilayer films (Mumbach et al., 2019; Eriksen et al., 2020). Furthermore a monolayer structure is often regarded as one of the reasons for a more efficient sorting, as monolayer waste items are more correctly detected by optical NIR sorter (Kleinhans et al., 2021a). And finally, during extrusion, parts of multilayer films can be retained on the extrusion filter and thus removed into the residual stream (Alvarado Chacon et al., 2020).

On the contrary, PE film others and PP film others have a relatively lower net recovery, e.g., roughly 80% and 50% in QRP respectively. The presence of multilayer films and black plastic items in these waste categories can be regarded as one of the reason for a relatively lower net recovery. A considerable amount of PE film others is missorted during the QRP additional sorting, in which up to 13% of PE film others is forwarded into PE Film Natural bale that only targets PE film transparent clear waste. Detailed information can be found in the Appendix C – section 4.

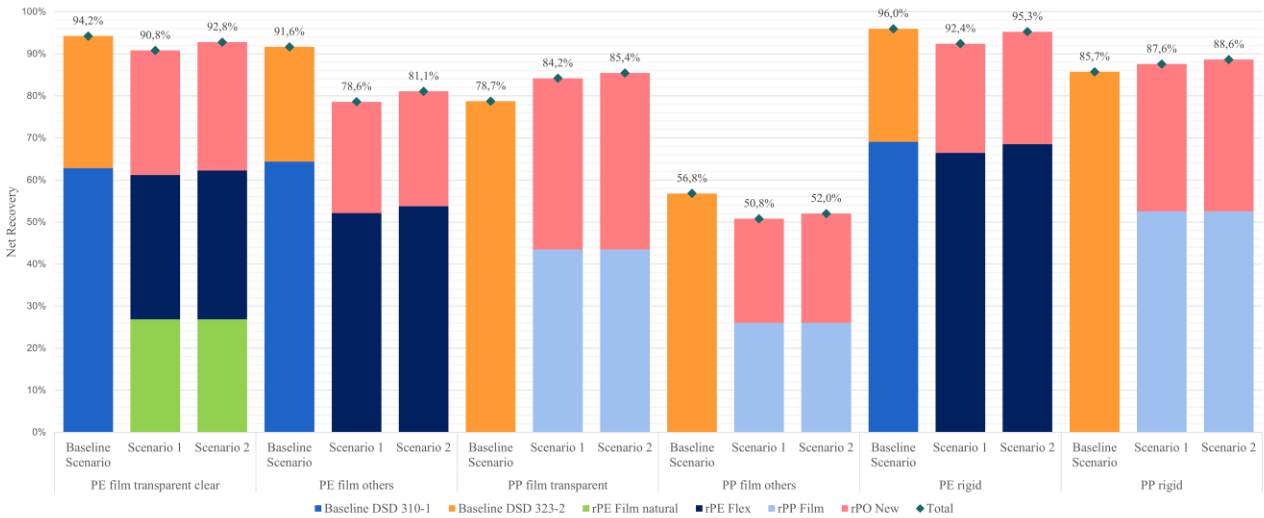


Figure 4.6 Overview of net recovery indicator of PE film transparent clear, PE film others, PP film transparent, PP film others, PE rigid and PP rigid (x-axis: aggregated waste category in sub-group level, see Table 4.1) in the conventional recycling and two QRP scenarios. The materials are recovered when they are found in correct regranulates (see Appendix C – section 3).

4.3.3 Polymer grade and transparency grade

The modeled compositional data of the Baseline DSD 310-1 and Baseline DSD 323-2 flakes and regranulates from the conventional recycling can be found in Figure 4.7, which also demonstrates the evolution of waste composition from the original bales (i.e., DSD 310-1 and DSD 323-2 bales) to the respective flakes and regranulates (as shown in Figure 4.8). The summary of the evolution of polymer grade (and process yield) of the conventional recycling in Figure 4.8 indicates that the polymer grade is improved, while the process yields of conventional recycling and QRP are similar.

In Figure 4.7, the S1 and S2 refers to the scenario 1 and scenario 2, respectively, while *other plastics* are all non-PO plastics and *other residues* are non-polymer materials. From Figure 4.7 it can be observed that other residues (including paper and fine fractions) as well as other plastics (i.e., all non-PO plastics) from the original bales to Baseline DSD 310-1 and Baseline DSD 323-2 flakes and regranulates are removed after washing, density separation, and extrusion by >90%. The polymer grade of Baseline DSD 310-1 flake and regranulate is 93% and 97% respectively, because PP can still be present after washing and extrusion process. Similarly, 7% of non PO can still be expected at Baseline DSD 323-2 flakes, making the grade of this flake to be 93%. However, the polymer grade of Baseline DSD 323-2 can reach up to 100% as it consists of a mixed PO materials, i.e., 57% PE and 43% PP, after extrusion process (Figure 4.7).

In the case of QRP scenarios, the modeled compositional data of QRP bales, flakes, and regranulates is also presented Figure 4.7. The polymer grade of PE Film Natural bale, PE Flex bale, PE Film bale, and PO New bale is 97%, 78%, 81%, 72% respectively. In fact, the QRP PE Flex bale is bale rich in PE films (75% transparent and colored PE films) and PO New bale are bales rich in a mixed PO films (63% transparent and colored PE/PP films). After washing, density separation, and drying, the polymer grade of PE Film Natural and PP Film flakes is 99% and 95% respectively. Further extrusion does improve the grade of rPP Film as the polymer grade increases to 96%, while the grade of rPE Film Natural remain at 99%, because most of the residue and non-PO have already been removed during washing and density separation. Within the PP Film flakes and regranulates, a small percentage of PE (3%) can still be found, because the density and melting point of PE and PP are close, thus cannot be removed in the density separation and extrusion with melt filter. For Tier 1 and Tier 2 PE Flex flakes and regranulates, the polymer grades are 90% and 95% respectively, while the polymer grade of Tier 1 and Tier 2 PO New flake and regranulate is 93% and 100% respectively. The T1- and T2-rPO New is expected to be composed of mixed PE (73%) and PP (27%).

The transparency grade indicator is added at the regranulates level. From Figure 4.7, it can be observed that the Baseline DSD 310-1 and Baseline DSD 323-2 regranulates have transparency grades of 62% and 57% respectively. For the QRP regranulates, the transparency grade of rPE Film Natural and rPP Film is 83% and 39% respectively. The transparency grade of T1- and T2-rPE Flex is 54%, while the transparency grade of T1- and T2-rPO New is 61%. The transparency grade shows that highest value is achieved by rPE Film Natural (83%) as a result of NIR-VIS sorting in QRP. The value for rPE Flex (54%) is slightly lower than Baseline DSD 310-1 (62%) whilst the value for rPO New (61%) is slightly higher than Baseline DSD 323-2 (57%), however this is in the same range when concerning the potential market applications (as shown by Bashirgonbadi et al., 2022). As for the rPP Film, the transparency grade is considerably lower than the other regranulates (39%), yet the polymer grade is high, thus many applications can still be made from rPP Film (Bashirgonbadi et al. 2022).

When comparing the modeled compositional data of conventional recycling and QRP in Figure 4.7, it can be observed that the rPE Film Natural and rPP Film produced in the QRP scenarios have high modeled PE and PP contents. These are regranulates produced from higher quality bales (i.e., PE Film Natural and PP Film) that are not produced from the conventional recycling. Moreover, the polymer grade of T1- and T2-rPE Flex (95%), which is basically the PE fraction from DSD 310-1 bale with the natural films 'picked out', is just slightly below the polymer grade of Baseline DSD 310-1 regranulate (97%). This finding indicates that rPE Flex would still allow to make similar applications to the conventional recycling with this bale, whereas the new bales with transparent film (i.e., PE Film Natural bale) can be used in higher-valued applications, as also shown in Bashirgonbadi et al. (2022). Similarly, the modeled compositional data of the T1- and T2-rPO New and Baseline DSD 323-2 regranulate is similar. Moreover, there is very little difference (< 1%) between the rPE Flex and rPO New composition in QRP Tier 1 and Tier 2 recycling. However, previous studies have suggested that high odour and ink contamination levels limit the potential use of regranulates (Bashirgonbadi et al. 2022; Horodytska et al., 2018; Hou et al., 2018). Moreover, a study by Grant et al. (2020) shows that high transparency level correlates to high quality regranulates and leads to higher market value as color may cause aesthetic issues and might not be suitable for certain applications (e.g., food packaging) (Schyns and Shaver, 2021; Radusin et al. 2020).

Furthermore, investigation of the technical properties of QRP regranulates by Bashirgonbadi et al. (2022) demonstrates that Tier 1 recycling enables rPE Film Natural and rPP Film to be reprocessed into more demanding applications such as shrink film, sealable pouches, and standing pouches. From the mechanical properties analysis of rPE Flex and rPO New, it is found that rPE Flex can be considered for film blowing but still requires measures (like blending with virgin or C&I) to increase dart drop resistance in a final product, while rPO New is unfit for film blowing. Other potential market for rPE Flex and rPO New is injection molding applications. The economic assessment results (in section 4.3.6) also indicate that processing PE Flex and PO New bales through Tier 2 recycling can be economically more attractive.

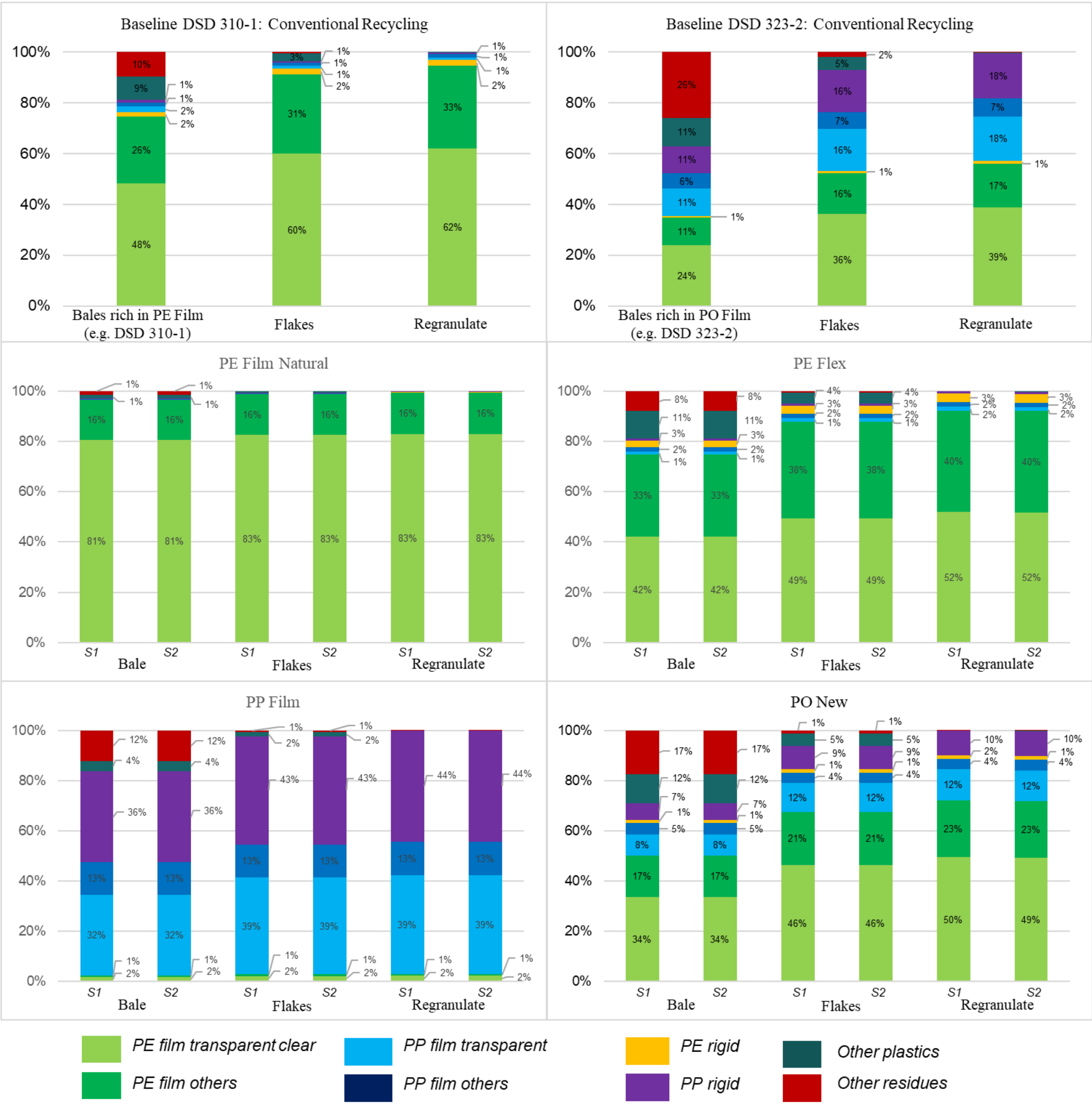


Figure 4.7 Modeled compositional data of flakes and regranulates from the conventional recycling and QRP, including the four QRP bales. S1 and S2 in the figure refers to the scenario 1 and scenario 2 respectively (see section 4.2.2).

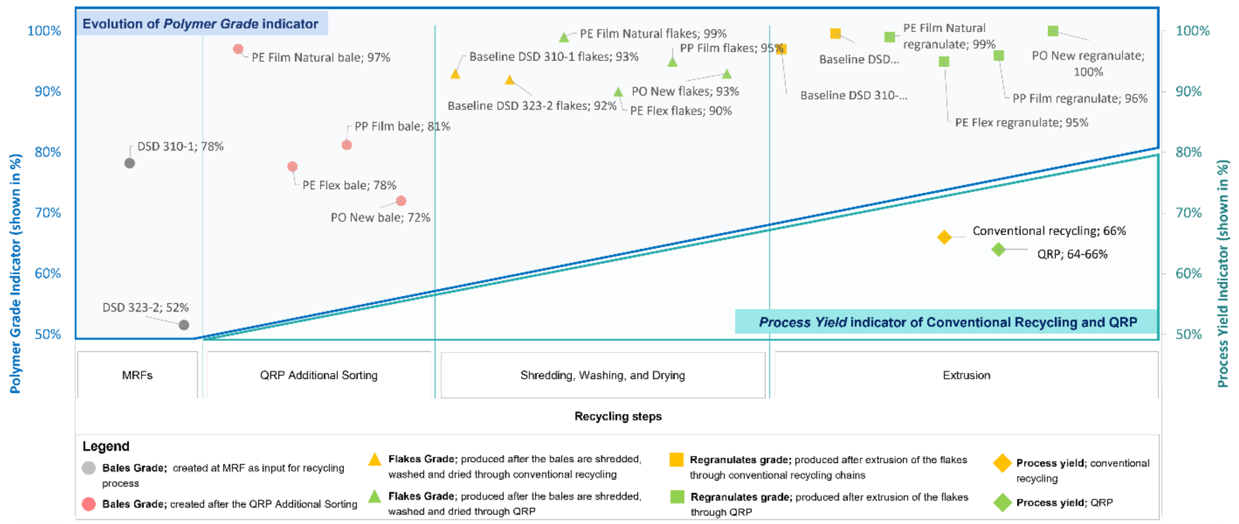


Figure 4.8 Evolution of polymer grade indicator from the original bales to QRP bales, flakes, and regranulates as well as the process yield indicator. The number on the labels denotes the value of the polymer grade or process yield.

4.3.4 Comparison of the modelled compositional data and experimental compositional analyses

The modeled compositional data of QRP flakes and regranulates is compared with the experimental composition analyses. Main results can be found in the Appendix C – section 5, including detailed information on the compositional analysis of the samples. It should be noted that certain disparities between the modeled compositional and experimental data occur. The model overestimates the experimentally found composition of PE content in the QRP PE Film Natural flakes and regranulates by a margin of 7 – 9%. Similarly, the modeled compositional data overestimate the PE Flex flakes and regranulates composition by 1 – 9%. The model overestimates the composition of PE and PP by 4 – 18% in PP Film and PO New flakes and regranulates, in which the biggest difference can be observed in the composition of rPP Film (i.e., overestimation of PP by 18%). The deviation is expected because the PE Film Natural flakes and regranulates contain up to 16% of PE film others waste category, which is potentially multilayer PE films. Similarly, it can be observed that PE Flex contain up to 40% PE film others (Figure 4.7), which can be multilayer films. The detailed composition of potentially multilayer films is not characterized in the MFA model but it might be detected in the experimental analysis, which can be composed of, amongst other, PET, EVOH, Aluminum, or paper (Roosen et al., 2022; Roosen et al., 2020).

The FTIR and DSC also have limitations in determining the composition of multilayer samples. FTIR can only detect the surface of flakes as the infrared beam does not penetrate more than 5 µm (Roosen et al., 2021; Chen et al., 2021). Flakes of one side PE and the other side PP are also assumed to be 50% PP and 50% PE, which is not necessarily the case. The DSC method for composition analysis of blends (Kisiel et al., 2018) is derivative method which significantly decreases the error margin of DSC-based compositional analysis, but remain an estimate at best due to the assumptions it requires (like the averaged out crystallinity of the constituting polymers). Additionally, the heats of fusion for each constituent relative to their content deviate from the linear regression. Deviations are caused by phase morphology transition from sea-island structure to co-continuous structure and a non-linear correction curve. Furthermore, crystallization interactions between the phases in a blend can contribute to faulty characterization (Jose et al., 2004; Madi, 2013; Kisiel et al., 2018; Larsen et al., 2021). This may explain some discrepancies (in Table C.11) where a decrease of PO content can be seen between the PE Flex and PP Film flakes to the point of their respective regranulates (e.g., from 83% to 79% in PP Film). After regranulation we should normally expect an increase of PO concentration because more residue and non-PO materials should be further removed by the melt filter.

These abovementioned findings highlight the current limitation of the MFA model on one hand, but also show the way forward to improve MFA model to assess the performance of plastic recycling. This includes the need for more detailed compositional characterization of the waste categories as well as more experimental work to get reliable quantification of the respective separation efficiencies. For example, a study from Brouwer et al. (2018) suggests that multi-material objects are usually being categorized based on their main material, whereas for the purpose of detailed compositional modeling, the full polymetric composition of the input waste would be more appropriate. Following the more detailed compositional analysis, the quantification of the separation efficiency based on the input-output experimental work should be carried out to get more reliable results.

4.3.5 Sensitivity analysis of the material flows modeling

Figure 4.9 shows the key outputs of the sensitivity analysis towards the performance indicators. More detailed results of the sensitivity analysis can be found in Appendix C –

section 6 (Figure C.5 – C.9). Figure 4.9 also indicates the relative importance of different modeling parameters by examining the relative changes of the performance indicators.

Figure 4.9A shows that bale composition can greatly influence the process yield of QRP, which indicates the importance of maintaining (or even improving) the input bales quality. This result also suggests that if the bales quality is improved by 25%, the process yield can increase from 64% up to 76%. Bale composition is also an important factor towards the polymer grade (Figure 4.9C and 4.9D). Moreover, it can be observed that bale composition influences the transparency grade indicator (in Appendix C – section 6). The influence of bale composition is relatively smaller on the net recovery indicator (Figure 4.9E and 4.9F).

The sensitivity analysis shows that separation efficiencies of the optical sorters are important towards the polymer grade of the respective bales, flakes, and regranulates (Figure 4.9C and 4.9D). For example, the separation efficiencies of NIR PP Film and PP Film Cleaner are most important towards the polymer grade of PP Film bales, flakes, and regranulates (Figure 4.9C). The same findings can also be found for the NIR-VIS LDPE Natural, NIR LDPE Cleaner and NIR PO Cleaner towards the polymer grade of PE Film Natural, PE Flex, and PO New bales, flakes and regranulates (in Appendix C – section 6). As for the PE Film Natural, the optical sorters' efficiencies affect the polymer grade of the bales but have relative small influence towards the flakes and regranulates (Figure 4.9B, and Appendix C – section 6). This can be explained by the fact that NIR-VIS LDPE Natural has already high efficiency to sort transparent PE Film, creating a relatively high polymer grade at bale level. These findings suggest that the efficiency of the optical sorters determine not only the quality of the bales created, but also the subsequent products after washing and regranulation, i.e., flakes and regranulates.

In Figure 4.9, the relative importance of achieving high efficiencies in the recycling equipment, i.e., cold and hot washing, density separation, and extruder, can be observed. The process yield of QRP (Figure 4.9A) and net recovery of the waste can drop considerably if the separation efficiency of the recycling equipment decrease by 25% (Figure 4.9E and 4.9F). As the recycling equipment typically already shows a relatively high separation efficiency, it does not create much improvement on the process yield or net recovery indicators.

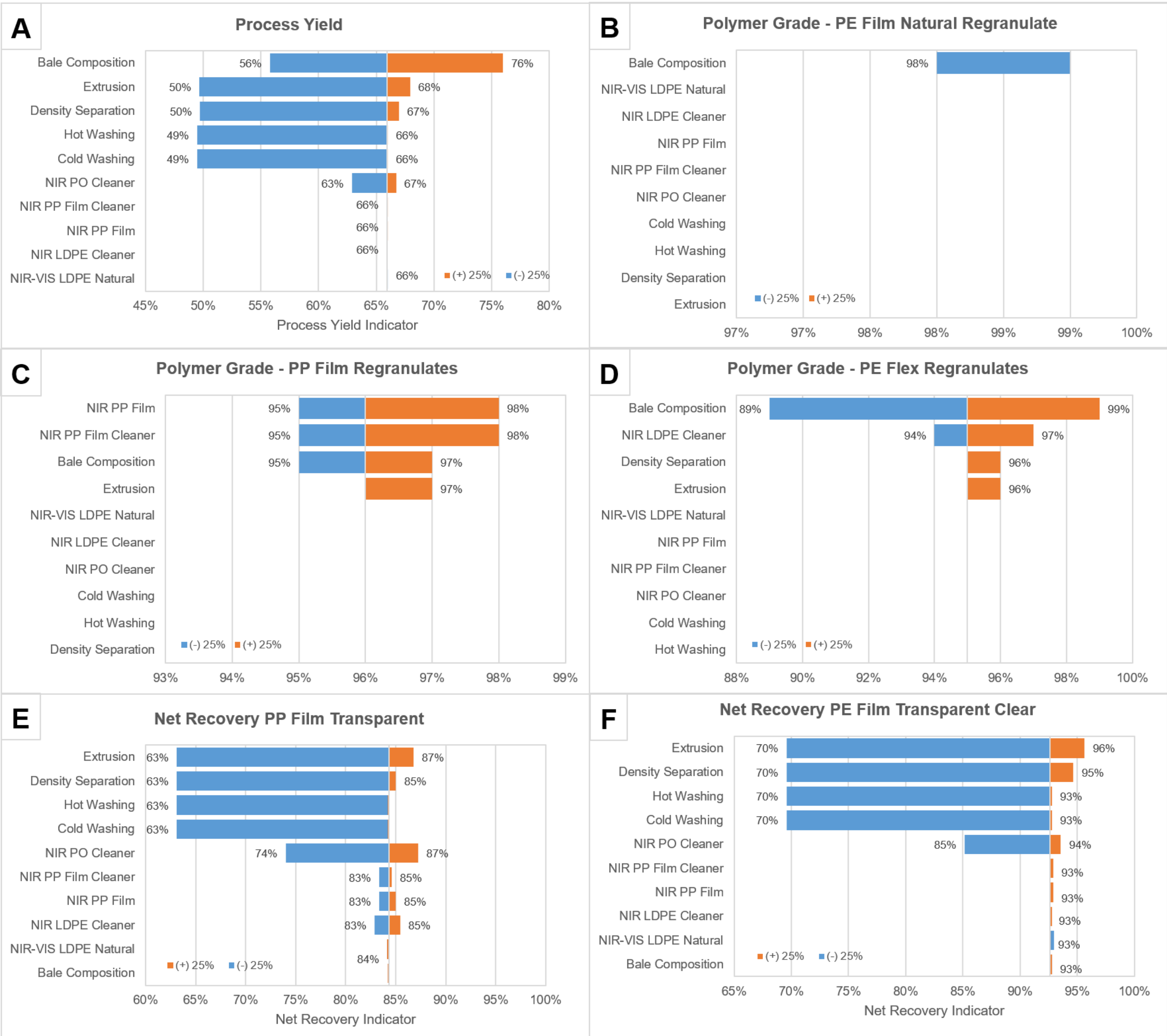


Figure 4.9 Key results of the sensitivity analysis towards the performance indicators. More results can be found in the Appendix C – section 6. The x-axis shows the effect on each performance indicator while the y-axis shows the respective modeling parameters that are varied by ±25%.

4.3.6 Economic assessment of PCFP mechanical recycling

4.3.6.1 Breakdown capital investment and total annual costs (CAPEX and OPEX)

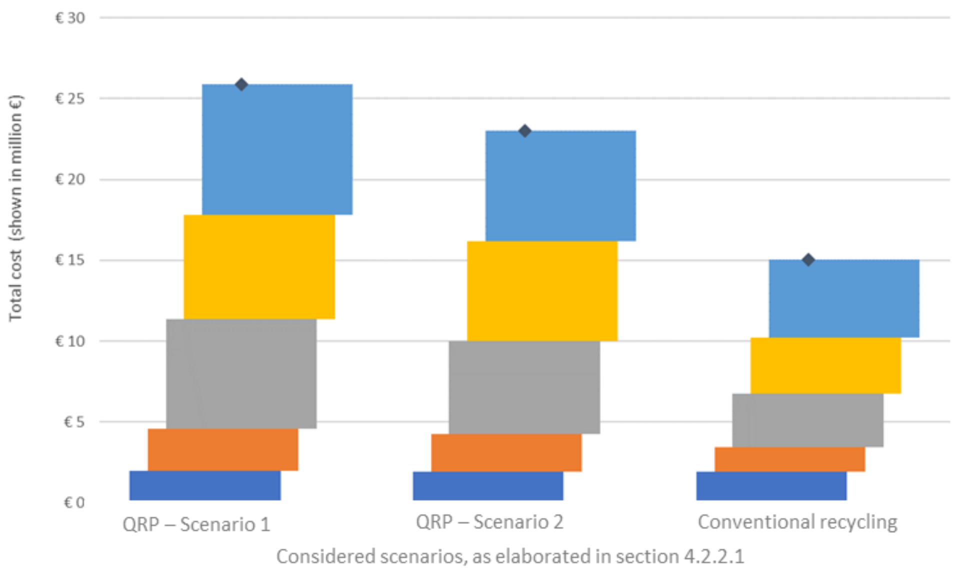
The detailed comparisons of needed capital investment and total costs (OPEX and CAPEX), as well as information on the total costs of different parameters (e.g., energy, residual treatment, etc.) can be found in Figure 4.10, Figure 4.11, and Appendix C – section 8. The capital investment needed for the process is increased from €24 million in the conventional recycling to €49 and €42 million in scenario 1 and scenario 2, respectively (Figure 4.10). The annual costs for QRP increases from €15 million in the conventional recycling to €26 and €23 million in scenario 1 and scenario 2, respectively (Figure 4.11).

In all cases we can observe that the additional sorting units, hot washing equipment, and improved extrusion facility are the most expensive units to invest together with investment in the building and construction. In QRP, the additional sorting is accounted for 6 – 7%, hot washing for 10 – 14%, and improved extrusion for 5 – 8% of the total required investment (Figure 4.10). Moreover, with additional sorting and washing units, investment on the spaces or area to place these equipment also rises as reflected in the increase of the overall investment for the QRP building (both for additional sorting and recycling facilities). From costs perspective, additional sorting, hot washing, and improved regranulation process are accounted for 4%, 9 – 12%, and 4 – 8% of the total annual costs in QRP (Appendix C, Figure C.10). Additionally, extra cost on the building for the additional sorting and the building for the recycling plant account for 9 – 12% of annual costs.

Within the cost parameters (Figure 4.11), the share of energy consumption takes up one third of the annual cost followed by the depreciation and direct production cost (insurance, labor, and maintenance). From the cost component, we can also observe that the depreciation and direct production cost of QRP increase because more equipment is used. Moreover, we can see a slight decrease in the cost of residual treatment because this improved recycling process captures most of the valuable material from the bales (see section 4.2.2).

	The breakdown of indicative capital investment in different scenarios		
	QRP – Scenario 1	QRP – Scenario 2	Conventional recycling
Building for Recycling	€ 9	€ 8	€ 6
Bale and final product handling	€ 2	€ 2	€ 1
Regranulation	€ 11	€ 9	€ 7
Hot washing	€ 7	€ 4	€ 0
Cold Washing	€ 7	€ 7	€ 7
Building for additional sorting QRP	€ 4	€ 4	€ 0
Sorting	€ 4	€ 4	€ 1
Feeding, conditioning, and shredding	€ 5	€ 5	€ 2
◆ Total	€ 49	€ 42	€ 24

Figure 4.10 Total capital investment needed for the QRP scenario 1, QRP scenario 2, and conventional recycling 310-1 and 323-2 bales. Total investment is shown in million € and broken down per plant section or recycling equipment. In QRP scenario 1, all fractions are processed through Tier 1. In QRP scenario 2, PE Film natural and PP Film are processed through Tier 1 and PE Flex and PO New are processed through Tier 2.



	The breakdown of indicative total costs in different scenarios		
	QRP – Scenario 1	QRP – Scenario 2	Conventional recycling
Energy use	€ 8	€ 7	€ 5
Labour, repair and maintenance, insurance	€ 6	€ 6	€ 3
Depreciation	€ 7	€ 6	€ 3
General expenses of the plant	€ 3	€ 2	€ 2
Residual treatment	€ 2	€ 2	€ 2
◆ Total	€ 26	€ 23	€ 15

Figure 4.11 Total annual costs of QRP scenario 1, QRP scenario 2, and conventional recycling 310-1 and 323-2 bales. The cost is shown in million € and broken down per cost parameters (e.g., energy use, depreciation, etc.). In QRP scenario 1, all fractions are processed through Tier 1. In QRP scenario 2, PE Film natural and PP Film are processed through Tier 1 and PE Flex and PO New are processed through Tier 2.

4.3.6.2 Cost benefit analysis of PCFP mechanical recycling

Despite the increase in the annual costs, it can be observed that QRP improves the net balance of DSD 310–1 and DSD 323–2 recycling (Figure 4.12). The negative values indicates the net loss in all scenarios, which gives an important insight into the waste management operation in the market, which to date is not self-sustaining. Yet, our analysis deliberately excludes gate fees, which should be included to assess the final viability of the plants. In fact, many studies suggest that most of the annual costs and annualized capital investment should be supported by an external source of income such as gate fees (Cimpan et al., 2016; Da Cruz et al., 2014, 2012; Marques et al., 2014). Nevertheless, looking at QRP as an improved mechanical recycling process for DSD 310–1 and DSD 323–2, it can be observed that QRP scenario 2 improves the economic value by 38 % (presented in Figure 4.12a). Per bale, the implementation of QRP scenario 2 improves the economic value of processing DSD 310–1 and DSD 323–2 by 57 % and 30 %, respectively. The net loss of processing DSD 310–1 decreases from -€83 per tonne in the baseline to -€36 per tonne in QRP scenario 2, reducing the margin that need to be filled by external parties such as PRO, e.g., via gate fees (Figure 4.12b). Similarly, the net loss of processing DSD 323–2 decreases from -€200 per tonne in the baseline to -€141 per tonne in QRP scenario 2 (Figure 4.12c). Moreover, recycling of DSD 323–2 shows higher net loss, partly because the bale has a higher contamination level and thus result in a relatively lower yield and generates more residue (see section 4.2.2). Nonetheless, the economic value of DSD 323–2 recycling is still improved compared to the conventional recycling process. Therefore, these findings indicate that QRP can potentially reduce the external financial support (e.g., gate fees), which is still subjected to further discussion in the circular economy of plastics waste recycling because the financing schemes still vary currently.

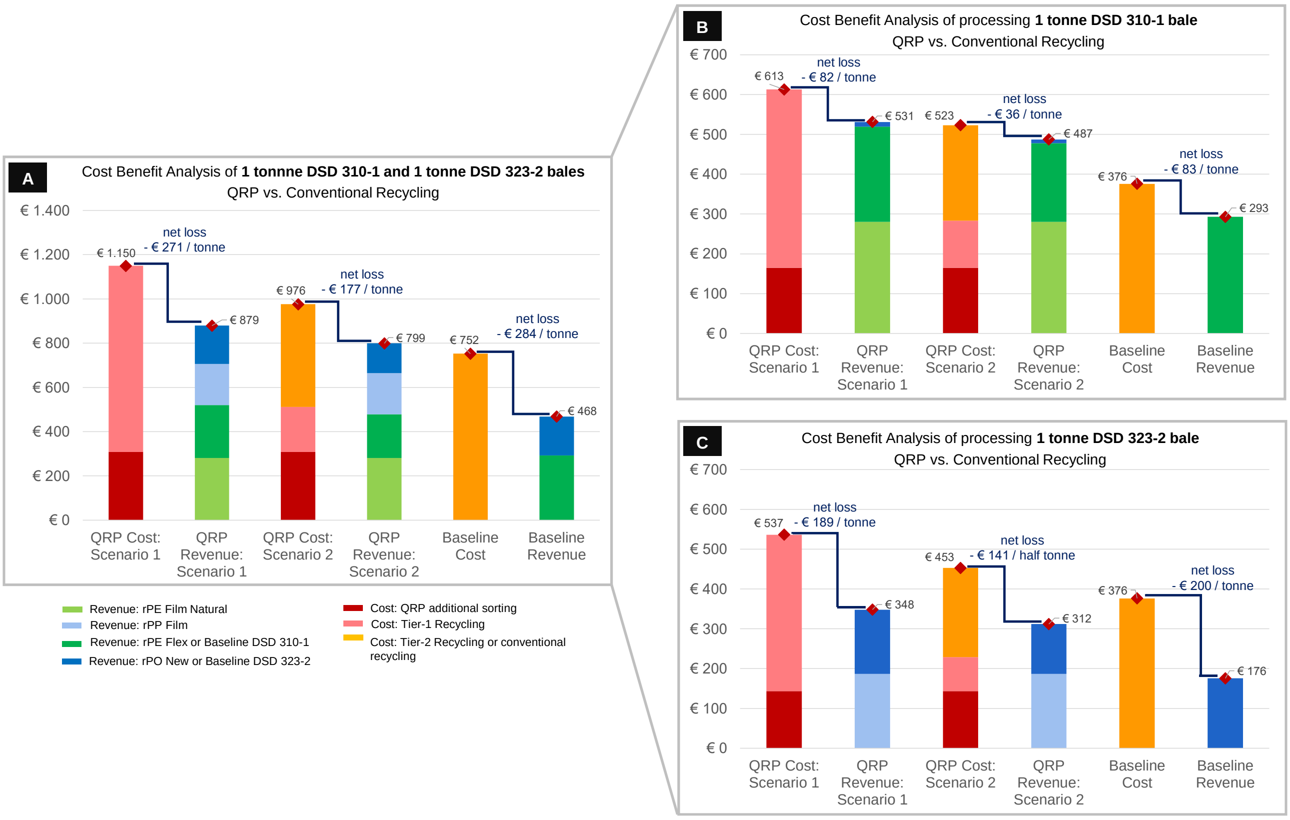


Figure 4.12 Cost benefit analysis (i.e., net profit loss) of the QRP scenario 1, QRP scenario 2, and baseline scenario: (A) processing 1 tonne DSD 310-1 and 1 tonne DSD 323-2 bales, (B) processing 1 tonne DSD 310-1 bale, and (C) processing 1 tonne DSD 323-2 bale.

4.3.7 Sensitivity analysis of the economic assessment of PCFP mechanical recycling

Many of the cost modelling components can greatly fluctuate. Larrain et al. (2021) have shown that the price of regranulates is amongst others influenced by the oil price. Other components such as energy use or labor cost also vary with time and region (Larrain et al., 2021; PwC, 2019). Therefore, the importance of the selected economic components towards the net profit/loss is investigated through a sensitivity analysis (Figure 4.13).

From the results of sensitivity analysis, it can be seen that the price of rPE Film Natural, rPP Film, and rPE Flex are among the most influential parameters, followed by the investment on the selected recycling equipment. This finding indicates the importance of maintaining high quality regranulates, suitable for demanding applications. This also means that a good quality of DSD 310–1 and DSD 323–2 input bales is required so that generated residue can be minimized, thus reducing the cost of residual treatment.

Next, electricity price is the most sensitive parameter followed by depreciation rate. In fact, depreciation accounts for almost one-third of the total cost and ranks amongst the most sensitive parameters. These findings highlight that the strategy on depreciating the investment for each equipment should be properly formulated. The annual costs can be significantly reduced if we invest on an equipment with longer lifespan (e.g. equipment that last for 8–10 years). By annualizing the investment to 8–10 years we can see that the economic value can be improved by 18–20 %. Moreover, as the energy and labor costs may differ from one region or country to another (PwC, 2019), it is imperative to make a detailed and regional feasibility study prior to the implementation of QRP in Europe or other regions/countries.

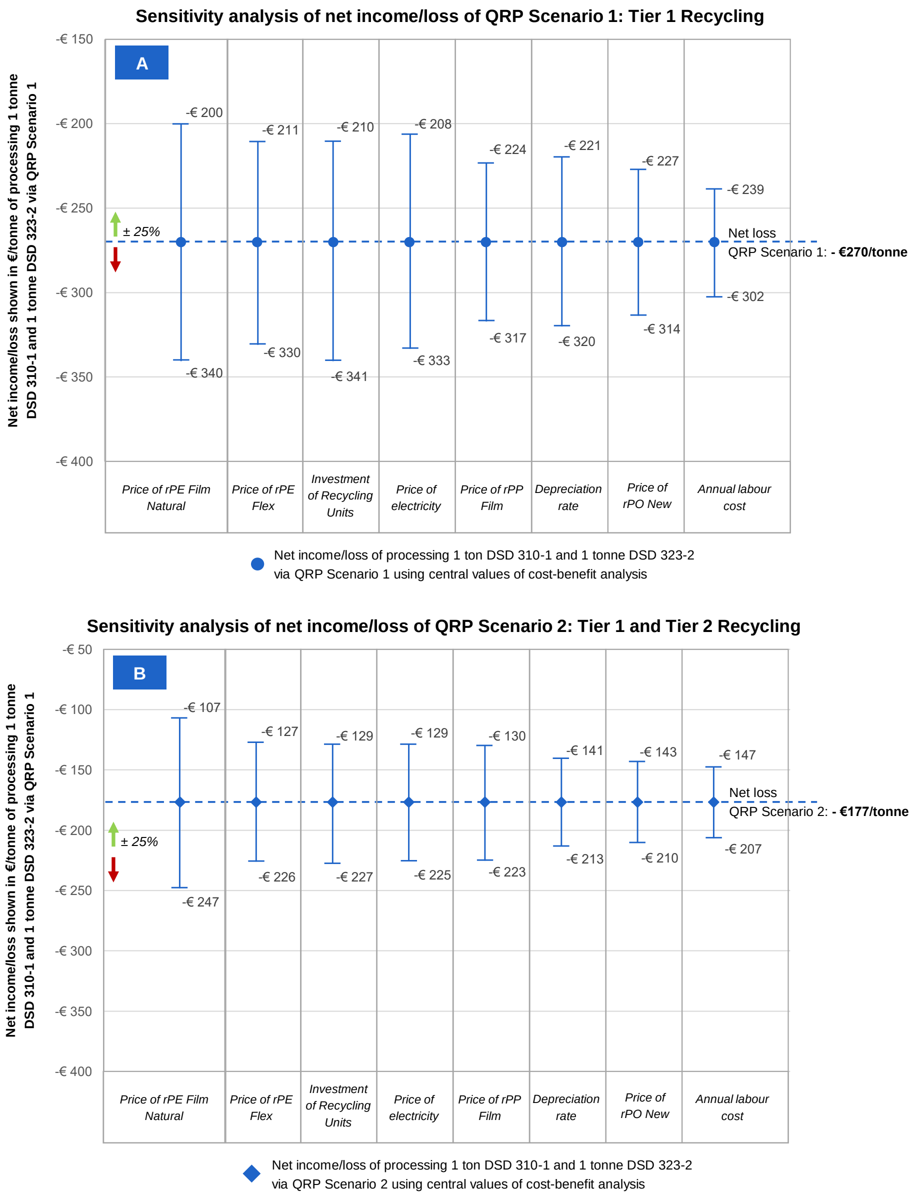


Figure 4.13 Sensitivity analysis of the selected cost modeling parameters towards net profit/loss of QRP. In QRP scenario 1 (A), all bales are processed through Tier 1. In QRP scenario 2 (B), PE Film Natural and PP Film bales are processed through Tier 1, whilst PE Flex and PO New bales are processed through Tier 2.

4.4 CONCLUSION

In this research, a MFA model is developed and applied to evaluate the technical performance of flexible packaging from household through an improved mechanical recycling process, called the *quality recycling process* (QRP). It can be observed that QRP goes beyond conventional mechanical recycling process by introducing additional sorting, hot washing with detergent, improved extrusion and deodorization. Next to MFA, an economic assessment is also performed to investigate the economic balance (viability) of QRP compared to the conventional mechanical recycling scenario.

The MFA shows that the process yield of QRP (i.e., 64%–66%) is similar to the conventional recycling (i.e., 66%). However, higher polymer grades can be obtained for certain regranulates from QRP, e.g., 99% for the rPE Film Natural and 96% for rPP Film. Moreover, rPE Film Natural has the highest transparency grade (i.e., 83%), which correlates to high quality regranulates and potentially leads to higher market value. QRP also produces (T1- and T2-) rPE Flex and rPO New with polymer grades around 95%, which is comparable to the current regranulates produced by conventional mechanical recycling. These findings suggest that rPE Flex and rPO New have similar qualities compared to the regranulates from conventional recycling, which allow the same applications.

Through an economic assessment, it is concluded that the higher operational costs for QRP is compensated by delivering higher quality regranulates (e.g., rPE Film Natural and rPP Film). Overall, it can be observed QRP improves the economic balance of flexible packaging recycling from households waste. It is shown that QRP can improve the economic value of the operation by 5–38 %, compared to the conventional mechanical recycling. Overall, our results show that it is possible to increase the mechanical recycling quality of flexible packaging waste in an economically viable way, yet, as in conventional recycling, financial supports (e.g., extended producer responsibility fees) still need to sustain QRP.

Thus, QRP has the potential to produce regranulates that have a better quality compared to conventional mechanical recycling, which is key to fulfill a larger market segment by plastic regranulates. Hence, the implementation of QRP by recyclers can be an important step to improve flexible packaging recycling rates and, finally, towards a more circular economy for flexible packaging.

CHAPTER 5: COST-BENEFIT ANALYSIS OF COLLECTING AND RECYCLING NON-HOUSEHOLD END-USE PLASTIC FILM WASTE FROM URBAN AREAS

Redrafted from:

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Chapter 5

Cost-benefit analysis of collecting and recycling non-household end-use plastic film waste from urban areas

5.1 INTRODUCTION

It is estimated that 29.1 Mt of plastic waste was generated in 2019, of which 32% (i.e., 9.4 Mt) was sent to recycling facilities and 68% (i.e., 19.7 Mt) were landfilled and incinerated. This resulted in an estimated 4.0 Mt of recycled plastic (as regranulate) production, which equals a recycling rate of around 15% in 2019 (Plastics Europe, 2020; Material Economics, 2022; Agora Industry, 2022). Out of 29.1 Mt of Waste generated, it is estimated that the packaging sector accounted for 61% (i.e., approximately 17.8 Mt) of the total waste generated (Plastics Europe, 2019). Hestin et al. (2017) estimate that 52% (equals around 9 Mt) of the plastic packaging waste in Europe is *non-household end-use plastic waste* – a terminology introduced by Kleinhans et al. (2021b), sometimes also called commercial and industrial (C&I) waste. Non-household end-use plastic waste is generated by 'end-users' from commercial activities (e.g., wholesales, retail stores, restaurants, coffee shops, cafés, etc.), industrial activities (e.g., manufacture, mining, construction, etc.), and institutional facilities (e.g., schools, offices, etc.). Much of this plastic waste is generated in urban areas such as cities or provinces (ISO, 2016; Kleinhans et al., 2021b).

Typically, non-household end-use plastic waste is not subjected to public waste management-related legislation (Kleinhans et al., 2021b). Without such binding regulations, the private market has sent a considerable amount of non-household end-use plastic waste to landfills, incineration, or export outside Europe (Jacobs et al., 2018). A study by Kleinhans et al., 2021c indicates that a significant amount of non-household end-use plastic waste is still thrown away in residual bins because of the absence of separate collection systems or economic incentives to recycle their waste.

However, data and studies regarding the flows and recycling potential of non-household end-use plastic waste remain scarce (Huysveld et al., 2019; Hestin et al., 2017; Salhofer, 2016). One study by Jacobs et al. (2018) indicates that more than half of the non-household end-use plastic waste is shipped to countries outside Europe (e.g., Malaysia or Vietnam). This finding aligns with data reported from analysis in the Belgian market, which suggests that a substantial quantity of C&I packaging waste is shipped to countries outside Europe (Valipac, 2022). The waste management practices of the shipped waste at their final destinations are poorly documented, but it is stated that there are concerns related to environmental impact and sustainability (Salhofer et al., 2021). Unlicensed waste management operators in these countries treat plastic waste with improper operating conditions (e.g., obsolete recycling infrastructure and inadequate personal protective equipment). Other possible waste treatments in these countries are illegal dumping, unsanitary landfill, or open burning (Liang et al., 2021; Petrlik et al., 2019; Chen et al., 2021; Yong et al., 2019). Thus, for this reason, in 2021 Valipac started a program that allows tracing of the collected and sorted Belgian non-household end-use plastic waste to their final (recycling) destination via external editors to ensure documented legal and operational complaints.

Currently, few economic incentives for non-household end-use plastic waste exist for recycling in Europe, which results in low recycling capacities (Essenscia, 2019; Mazzanti and Zoboli, 2013; D'Amato et al., 2019). One of the key drivers for a considerable amount of plastic waste export is thus cheaper export tariffs compared to domestic waste treatment. Nevertheless, from the waste management perspective, recycling non-household end-use plastic waste also has enormous potential to improve regranulates production, increase recycling rate targets and play a crucial role in the circular economy of plastic in Europe (Kleinhans et al., 2021b; Lange, 2021).

Non-household end-use plastic waste seems to be 'forgotten' as a separate category in waste statistical databases and reports (OECD, 2018; De Weerdt, 2020; Kleinhans et al., 2021b). Yet, it is an important stream for achieving recycling targets in certain regions, as indicated by Hestin et al. (2017). Next to quantity, there is limited information on the waste composition of non-household end-use plastic packaging waste in Europe. However, Hestin et al. (2017) estimate that 58% is film (e.g., shrink films, stretch films, refuse sacks, etc.), while the remaining 42% is rigid (e.g., bottles, tubes, trays, etc.). This finding aligns with the study by Bracken (1990) and OECD (2018), which indicate that plastic film is the most prevalent type of C&I waste in the United States and Australia, respectively. Within the non–household end-use plastic film waste, polyethylene (PE) is estimated to be the largest fraction (i.e., 83%), followed by polypropylene (PP) (i.e., 16%) and polyethylene terephthalate (PET) (i.e., 1%). Moreover, it is estimated that the non-household end-use rigid plastic waste consists of 64% high-density polyethylene (HDPE), 19% PP, and 16% PET (Hestin et al., 2017; Horodytska et al., 2018). Some studies indicate that non-household end-use plastic waste tends to have less contamination and impurities than household plastic waste (OECD, 2018; Hestin et al., 2017; Horodytska et al., 2018; Nimmegeers and Billen et al., 2021). Horodytska et al. (2020) show that non-household end-use plastic film waste has better feedstock quality for mechanical recycling because the waste stream has a relatively homogenous composition.

Currently, the business cases of selective collection and recycling non-household end-use plastic waste from urban areas are done by commercial or voluntary agreements between the waste producers and waste management companies. For example, waste producers and operators in the construction sector can come to an agreement to selectively 'pick' only certain high-value waste items, such as windows and doors, for recycling (Bendix et al., 2021; Gardner, 2020). In the agriculture sector in many European countries, waste management is done voluntarily (and agreed upon) between the farmers and recyclers. The recyclers usually collect the waste through 'a bring' or 'a pickup' system, depending on the waste quantity (Bauer, 2019; Scarascia-Mugnozza et al., 2012; Agriculture Plastic Environment, 2021). Usually, businesses are encouraged by local governments and extended producer responsibility (EPR) organizations to (voluntarily) sort their waste by material types (e.g., plastic, paper, cardboard, etc.). In some cases, rewards are given to businesses, such as in Belgium where authorized waste operators collect the waste for recycling, and in return, waste producers receive a one-time premium incentive of €150 (starter incentive) and a recycling incentive of €30/tonne of plastic packaging waste (Valipac, 2022). Recently, significant progress on non-household end-use plastic waste treatment has been made in the Flanders region–Belgium by the ratification of VLAREMA[1] regulations in 2021. In article 8 of VLAREMA, companies are obliged to perform a source separation of up to 24 waste categories,

[1] *VLAREMA stands for the 'Flemish regulations concerning the sustainable management of material cycles and waste'*

including plastic waste (Vlaamse Regering, 2021). In compliance with the regulations, companies must establish a partnership with authorized waste collectors and a compliance certificate will be given by local (regional) authorities (i.e., OVAM[2]) (Renewi, 2023a; OVAM, 2023).

In the context of non-household end-use plastic, urban areas are important because of high business densities (Tonini et al., 2020; Acke et al., 2020). This makes urban areas crucial to improve the material utilization efficiency of a region (Derrible et al., 2021) and become a source of concentrated secondary resources that can be recycled into valuable materials (Zhang et al., 2015). Extra costs and environmental footprint arise from the conservation of raw materials in urban areas, for example, caused by selective waste collection and recycling (Da Cruz et al., 2014; 2012). Studies from Boskovic et al. (2016) and Marques et al. (2014) indicate that costs associated with selective collection can account for up to half of the costs of the recycling system. Thus, properly estimating collection costs is crucial in assessing the business case development of non-household end-use plastic waste recycling. The estimation of selective collection costs can be improved by understanding key parameters such as waste quantity and composition from the urban areas, number of collection points, vehicle capacity, and collection frequencies (Boskovic et al., 2016).

Furthermore, literature suggests that recycling non-household end-use plastic waste is still scattered, less organized, and driven mainly by initiatives between waste producers and waste management companies (Bendix et al., 2021; Gardner, 2020; Bauer, 2019; Agriculture Plastic Environment, 2021). As a result, the recycling rates of non-household end-use plastic waste are relatively low and are estimated to be around 20–30% (Hestin et al., 2017; Kleinhans et al., 2021b). Yet, from the environmental perspective, mechanical recycling of non-household end-use plastic film still outperforms incineration with energy recovery (Horodytska et al., 2020; Huysman et al., 2017).

Therefore, this study develops and applies a method to develop (or predict) potential business cases of selective collection and mechanical recycling of non-household end-use plastic waste from urban areas, focusing on the largest plastic film fraction, as indicated by Hestin et al., 2017. The City of Ghent and its twelve neighboring municipalities in Belgium are selected as the case study. The potential business cases of different selective collection and

[2] *OVAM stands for the 'Public waste agency of Flanders', which responsibles in developing environmental policies and reinforcements*

recycling scenarios are predicted by building a cost and benefit analysis (CBA) model. Granular logistic simulations, modeling the process flows within mechanical recycling facilities, and quantifying the economics and greenhouse gas (GHG) emission of the entire process are considered in the CBA. The logistic simulations are done in OptiFlow© software (Conundra, 2023), based on the input from waste operators. The material flows and economic modeling is developed by following material flow analysis (MFA) and economic assessment modeling approach (Bashirgonbadi et al., 2022; Larrain et al., 2021; Hernández et al., 2023; Cimpan et al., 2016). Finally, the GHG emission (in kg CO2-eq) is quantified by following the life cycle assessment (LCA) modeling, as also presented in Civancik-Uslu et al. (2021), Zero Waste Europe (2020) and Quantis (2020).

5.2 MATERIALS AND METHODS

5.2.1 Overall modelling approach

An overview of the business case development using cost benefit analysis (CBA) modeling of selective collection and recycling non-household end-use plastic film waste is presented in Figure 5.1. The system boundary (as a case study) considered in this study is elaborated in section 5.2.2. Two data sources are used in this study: i) primary data collected from real waste sampling combined with ii) literature and databases (Lase et al., 2022; Bashirgonbadi et al., 2022; Orbis, 2022) (Figure 5.1). Two waste sampling campaigns were conducted for i) estimation of film waste quantity and ii) waste compositional analysis, which is elaborated in section 5.2.3. Next, the annual costs of different selective collection schemes from urban areas (weekly, fortnightly, or monthly collection frequencies) are estimated using OptiFlow© Route Optimization software (Conundra, 2023), and are explained in section 5.2.4. The annual costs of mechanical recycling non-household end-use plastic film are estimated by combined material flow analysis (MFA) in the recycling plant and economic assessment, as suggested by Larrain et al. (2021), Hernández et al. (2023), and Bashirgonbadi et al. (2022). The required inputs for the MFA model are waste quantity and composition (as elaborated in section 5.2.3), recycling plant configuration, and separation efficiency of the equipment used in the recycling plant (section 5.2.5). Later, the MFA results and data on capital investment and utility consumption are used as the basis for the economic assessment, as elaborated in section 5.2.6. Furthermore, a sensitivity analysis is carried out to see how residue content in the collected waste (in %) impacts the economic balance of mechanical recycling non-household end-use plastic film. Lastly, the GHG emission associated with collecting and recycling non-household end-use plastic film waste from urban areas in this study is estimated and compared with the baseline scenario (i.e., virgin PE granulate production with incineration as EoL treatment), as elaborated in section 5.2.7.

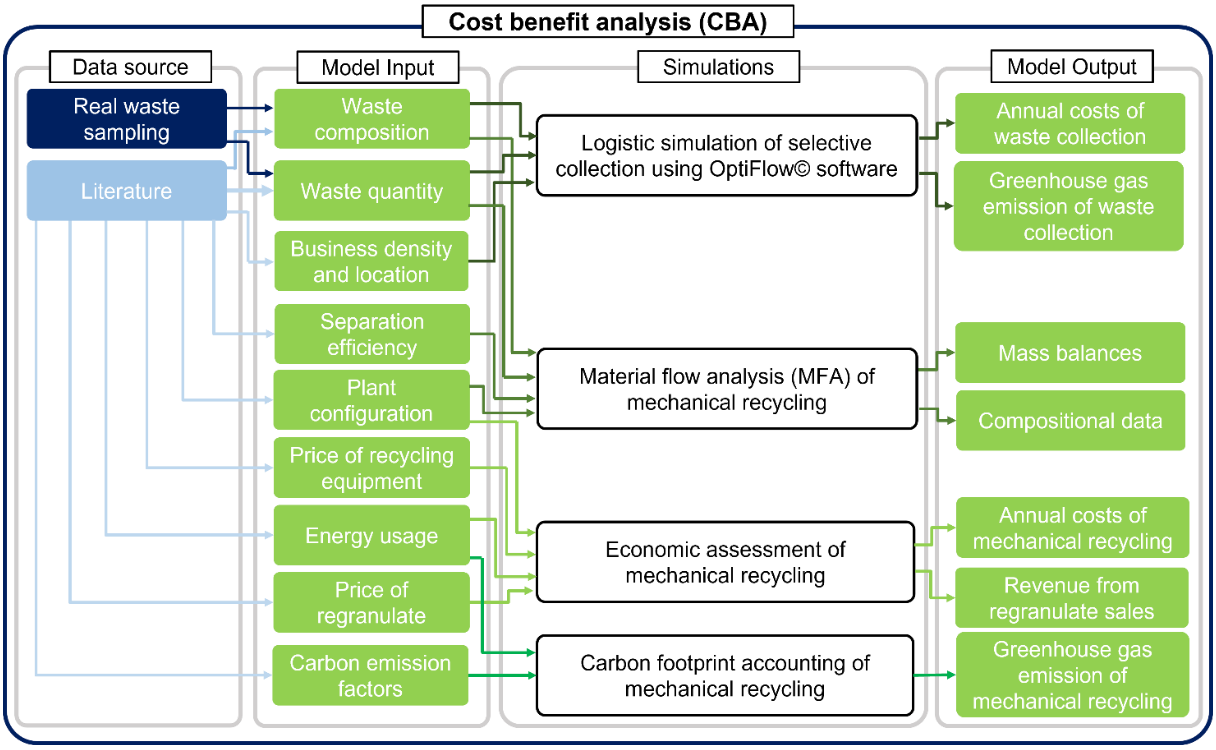


Figure 5.1 Summary of the cost-benefit analysis (CBA) model inputs, simulations and outputs.

5.2.2 Description of the system boundary and scenarios

This study considers the urban areas of Ghent and its twelve neighboring municipalities in Belgium as a case study (system boundary). The City of Ghent (postcode: 9000–9070) is located in the Flemish Region of Belgium that covers an area of approximately 156 km^2 with a total of 261,483 inhabitants (Kerselaers et al., 2020), which equals a population density of 1,655 inhabitant/km2. This study also includes the effect of processing scale (in tonne/year waste processed) on recycling operations. For this purpose, twelve neighboring municipalities within approximately ten kilometers (radius) of Ghent are considered, from which the non-household end-use plastic film waste can be collected and processed at the recycling plant hub in Ghent. These municipalities are Sint-Martens-Latem (postcode: 9830), Melle (postcode: 9090), Zelzate (postcode: 9060), Wetteren (postcode: 9230), Merelbeke (postcode: 9820), De Pinte (postcode: 9840), Lokeren (postcode: 9160), Deinze (postcode: 9800), Nazareth (postcode: 9810), Lochristi (postcode: 9080), Evergem (postcode: 9940), and Eeklo (postcode: 9900).

Six NACE sectors (standardized classification for economic activities in Europe; Eurostat, 2022c) are selected in this study: NACE A. *Agriculture, Forestry, and Fishing*, NACE B. *Mining and Quarrying*, NACE C. *Manufacturing*, NACE D. *Electricity, Gas, Steam and Air Conditioning Supply*, NACE F. *Construction*, and NACE G. *Wholesale and Retail Trade; Repair of Motor Vehicles and Motorcycles*. These sectors are selected because they are Europe's biggest non-household end-use plastic producers (Kleinhans et al., 2021b; Eurostat, 2022c). NACE sector E. *Water Supply, Sewage, Waste Management and Remediation*, NACE sector G 46.77 *Wholesale of Waste and scrap*, and NACE C.20–22 (*Manufacture of chemical, pharmaceutical, rubber, and plastic products*) are excluded from this study because these sectors do not fall under the definition of 'non-household end-use plastic waste'. The exclusion of these sectors also prevents double counting on estimating the total waste generation (e.g., from NACE G 46.77) from the considered urban areas in this study (Kleinhans et al., 2021b).

The four baseline scenarios considered in this study (in Table 5.1) consist of two waste compositions (high and low feedstock quality, in section 5.2.3), three waste collection frequencies (weekly, fortnightly, and monthly; in section 5.2.4), and two recycling plant layouts (basic and advanced recycling plants, section 5.2.5). Moreover, in each scenario (S1–S4, Table 5.1), the processing capacity (i.e., mass input to recycling plant, in tonne/year) is varied from 2,500 tonne/year to 20,500 tonne/year (i.e., maximum processing capacity in tonne/year, after Larrain et al., 2021). This approach is taken to investigate how i) waste composition (i.e., feedstock quality), ii) selective collection frequencies, and iii) recycling processing capacity affect the overall economic balance and viability of the whole recycling chain.

Table 5.1 Summary of non-household end-use plastic film waste recycling scenarios considered in this study. Three collection scenarios (weekly, fortnightly, and monthly) are included in each recycling scenario (S1–S4). Waste input composition can be found in Section 5.2.3. The recycling plant configuration can be found in Section 5.2.5.

Scenarios	**Collection frequencies**	**Waste input composition**	**Recycling plant configuration**	**Processing scale** (in tonne/year)
S1	Weekly, fortnightly, and monthly	Higher quality	Basic recycling plant	2,500 – 20,500
S2	Weekly, fortnightly, and monthly	Lower quality	Basic recycling plant	2,500 – 20,500
S3	Weekly, fortnightly, and monthly	Higher quality	Advanced recycling plant	2,500 – 20,500
S4	Weekly, fortnightly, and monthly	Lower quality	Advanced recycling plant	2,500 – 20,500

5.2.3 Estimation of non-household end-use plastic film waste quantity and composition

5.2.3.1 Waste quantity estimation

Table 5.2 provides key examples of the dataset used to estimate the quantity of non-household end-use plastic film. The waste quantity is estimated based on real waste sampling in 2018 done by Valipac[3] in the City of Ghent–Belgium, from 3,470 companies within NACE sector A–G. The data collection from waste sampling provides us with the total waste quantity per NACE sector (in tonne) from several companies. For example, in Table 5.2, 58 and 400 tonne of non-household end-use plastic film waste were collected from NACE sector G.45 and NACE sector G.46 during the sampling campaign, respectively. A total of 58 tonne and 400 tonne of plastic film waste were collected from 261 and 564 companies within NACE sectors G.45 and G.46, respectively. Therefore, the (average) quantities of the non-household end-use plastic film generated per company within NACE sector G.45 and NACE sector G.46 are estimated to be 0.22 tonne/year.company and 0.71 tonne/year.company, respectively. These estimations are calculated by dividing the weight of non-household end-use plastic film waste collected (in tonne) by the total number of companies that participated in the sampling campaign in 2018, as shown in Table 5.2.

The next step is estimating the total non-household end-use plastic film waste generation per NACE sector in the whole selected region (see section 5.2.2). This is done by combining (and extrapolating) the dataset built from waste sampling in 2018 and Orbis (2022)

[3] *Valipac is a producer responsibility organization (PRO) or 'Green Dots company' in Belgium responsible for the extended producer responsibility (EPR) applied to commercial and industrial waste (non-household sectors)*

databases. The extrapolation is done by multiplying the (average) waste generated per company with the total active companies listed in Orbis (2022) database (Table 5.2) within Belgian postal codes 9000–9940 (more in section 5.2.2). For example, it is estimated that one company within NACE sector G.45 generates 0.22 tonne of plastic film waste annually, while there are 484 companies within the same NACE sector in Ghent (Postal code: 9000–9070). Therefore, the amount of non-household end-use plastic film waste generated from NACE sector G.45 from urban areas Ghent is estimated to be 107 tonne/year (i.e., 0.22 tonne × 484 companies), as shown in Table 5.2. The complete dataset on waste quantity can be found in Appendix D, Table D.1and Table D.2. Moreover, it is important to note that we discounted the total active company listed in Orbis (2022) databases by 20%. This assumption is made because we observe that some of the offices are empty buildings, which generate no plastic waste. Lastly, a similar approach is used to estimate total non-household end-use plastic film waste generated in the 12 neighboring municipalities (see section 5.2.2). More information on waste quantity from the 12 selected municipalities can be found in Appendix D – section 3.

Table 5.2 Examples of datasets from waste sampling conducted in Ghent–Belgium in 2018 and total active companies based on Orbis (2022) database. The complete dataset is available in Table D.1 and D.2. Units: Waste quantity (tonne), waste generated per company (tonne/year.company), Total Waste generated (tonne/year.NACE sector).

NACE sectors: codes and names	Dataset from waste sampling in 2018 done by Valipac			Orbis (2022)	Extrapolation
	Waste quantity	*Number of Companies*	*Waste generated per company*	*Total active companies*	*Total waste generated*
G–*Wholesale and retail trade; repair of motor vehicles and motorcycles*					
G45–*Wholesale and retail trade and repair of motor vehicles and motorcycles*	58	261	0.22	484	107
G46–*Wholesale trade, except of motor vehicles and motorcycles*	400	564	0.71	2,128	1,508
G47–*Retail trade, except of motor vehicles and motorcycles*	429	1,065	0.40	3,386	1,354

5.2.3.2 Waste compositional analyses

Two waste compositions in the baseline scenarios (higher or lower quality), as feedstocks to recycling plants, are considered in this study (Table 5.3). Our real waste sampling was performed between December 2021–February 2022 by GRCT (a waste management company in Belgium), with a total of 34 companies participating. The results of our waste sampling are provided in Table 5.3. The waste sampling campaign was performed to determine waste compositional data of non-household end-use plastic film covering *Wholesale* (e.g., NACE G.46), *Retail* (e.g., NACE G.47), *Construction* (e.g., NACE F.41), *Logistics* (e.g., NACE H.49), and '*other*' sectors (e.g., NACE C.10, NACE C.18, etc.). A few key examples of the collected waste during the waste sampling campaign are provided in Figure 5.2. More information on the waste samples is available in Appendix D – section 4. Moreover, Table 5.3 also provides non-household end-use plastic film composition estimated by Hestin et al. (2017). Finally, the waste composition of these two studies is averaged and used as input for the CBA. In general, the collected samples of non-household end-use plastic film waste are similar to the household counterparts, such as film bags, shrink and stretch films, etc. However, there might a difference in terms of quantity, for example the Construction sector can have a considerable higher quantity of transparent PE film, as shown in Figure 5.2.

The residue content was not determined systematically during waste sampling (e.g., level of moisture and dirt measurement), hence it is estimated from literature. The higher feedstock quality is assumed to contain 5% of residue, which is taken from Roosen et al., 2021; Thoden Van Velzen et al., 2016. The lower feedstock quality assumes a higher residue content (i.e., 25%; Thoden van Velzen and Brouwer, 2014), while the share of the waste composition of the other waste categories is maintained.

Figure 5.2 A few images of the collected samples from *Construction* sector (e.g., NACE sector F.41) from urban areas of Ghent. More information on the samples is available in the Appendix D – section 4.

Table 5.3 Waste compositions used as input for the CBA model. The composition used in the model is averaged from the waste sampling campaign conducted in urban areas of Ghent in December 2021–February 2022 and Hestin et al. (2017). A more detailed compositional analysis based on the waste sampling in urban areas of Ghent is available in Appendix D – section 4.

Waste Category	Characteristics	Composition (in %)		Averaged composition (in %)	
		Waste sampling	Hestin et al. (2017)	[1]*Higher feedstock quality*	[1]*Lower feedstock quality*
PE film	Transparent	50	79	48	38
	Colored	36		35	27
PP film	Transparent	3	15	5	4
	Colored	3		5	4
Other films (PVC, PET, etc.)		4	1	2	2
Residue		[2]5	[2]5	[2]5	[3]25
Total		100	100	100	100

[1]*The higher feedstock quality corresponds to 5.0% residue content. The lower feedstock quality corresponds to 25% residue content. In waste compositions, the share of the other waste category (i.e., PE transparent, PP Colored, etc.) remains proportionally the same.*

[2]*Residue content (i.e., 5%) is taken from Roosen et al. (2021) and Thoden Van Velzen et al. (2016).*

[3]*Residue content (i.e., 25%) is taken from Thoden van Velzen and Brouwer (2014), while the share of the other waste category (i.e., PE transparent, PP Colored, etc.) is maintained*

5.2.3.3 Sensitivity analyses on residue content

In Chapter 5, a sensitivity analysis is performed only on the waste input compositions because this is considered as new primary datasets, while the sensitivity of the remaining datasets (e.g., separation efficiencies and economic parameters) have been investigated in Chapter 4. Moreover, as shown in Chapter 4, ± 25% of changes in waste composition can affect the recycling performance of household flexible packaging waste treatment. Therefore, in this study, a sensitivity is carried out to assess the impact of potential variation on the non-household end-use plastic film waste composition (by means of higher residue content, in %) entering the two recycling plants (i.e., basic and advanced recycling plants; more in section 5.2.5) towards net cost or benefit results (in €/tonne output, as elaborated more in section 5.2.6). In the sensitivity analysis, the recycling plant capacity of both plants is fixed on the amount of waste collected from the urban areas considered in this study (section 5.2.2 and discussed in section 5.3.1). The residue content is increased incrementally (5% interval) from 5% up to 50%. At every interval variation, the results of recycling yield and net economic balance are recorded and discussed.

5.2.4 Logistic simulation of non-household end-use plastic film waste collection from urban areas

A logistic simulation of collecting non-household end-use plastic film waste from urban areas is carried out using OptiFlow© Route Optimization software (Conundra, 2023). Three selective collection scenarios are developed: weekly, fortnightly, or monthly waste collection frequencies. It is assumed that the diesel garbage trucks (Euro 6 standard garbage trucks with 40 m^3 capacity) begin the selective collection from the mechanical recycling facility (hub) located in the Port of Ghent–Belgium. Averaged data for loose LDPE films (17 kg/m3) is used to convert the mass-based data of waste quantity (in tonne, section 5.2.3) into volume-based data needed for logistic simulations (Tellus, 2021; Federal Recycling & Waste Solutions, 2022). Moreover, a compaction factor of 10 (estimated value communicated by waste operators) is used in the logistic simulations when the garbage trucks compress the collected plastic film waste.

The garbage trucks collect non-household end-use plastic film waste from companies listed in Table D.2 and Table D.3, in which the addresses are collected from Orbis (2022) databases. The number of garbage trucks needed for collecting non-household end-use plastic waste depends on the number of companies and collection frequencies per municipality, in which the data points are provided in Appendix D – section 5. It is assumed that the truck's speed is limited to 30 km/hour, following the standard speed limit in Belgian urban areas (European Commission, 2021). The average service time stop (at each address) is 8 minutes and the unloading time at the recycling facility is assumed to be 10 mins. Moreover, the truck will make another trip if there is still time available to make another waste collection, assuming that the waste collection is done from 08.00–18.00. The estimated waste collection and unloading time is obtained from waste operator input. Finally, the estimated driver cost is €19.5/hour with an operational cost (incl. fuel and costs associated with purchasing the truck) to be €0.74/km (on average), which is also based on the communication with waste operators.

5.2.5 Modeling material flows in the mechanical recycling plants

5.2.5.1 Plant design

This study assumes that the recycling plant is designed for recycling PE film waste, as it is found to be the largest fraction of the non-household end-use plastic film waste (Hestin et al., 2017 and Table 5.3). Two recycling configurations are considered, i.e., the *basic* recycling plant (Figure 5.3A) and *advanced* recycling plant (Figure 5.3B), adapted from Larrain et al. (2021) and Lase et al. (2022). It is assumed that the recycling plants can process up to (max. capacity) around 20,500 tonnes/year of waste, equivalent to up to around 2.5 tonne/hour processing capacity (Larrain et al., 2021).

The basic recycling plant (Figure 5.3A) consists of a bag opener, shredder, cold washing, density separation, dryers, and a single melt filter extruder. The non-household end-use plastic film waste is assumed to be collected in plastic bags, which are open and then shredded into materials the size of roughly ten millimeters. After that, the plastic waste stream is washed with 'cold' water (25–40°C), removing contaminants like organic residue, paper, and labels. The cold washing is then followed by density separation to remove higher-density polymers (e.g., PET), metals, and other residues. The floating plastics (mainly polyolefin) are dried using mechanical and thermal drying and then extruded (Brouwer et al., 2018; Larrain et al., 2021; Lase et al., 2022). According to Bashirgonbadi et al. (2022), additional sorting and hot washing can improve recycling performance, regranulates' quality, and net economic balance of recycling operation. Hence, in the advanced recycling process (Figure 5.3B), a NIR PE Film Cleaner (i.e., negatively sorting non-PE film items) and 'hot' washing (up to around 80°C with detergents) are introduced. The described recycling process is expected to produce regranulates rich in PE film, which is called '*rPE_{basic}*' or '*$rPE_{advanced}$*' in this article.

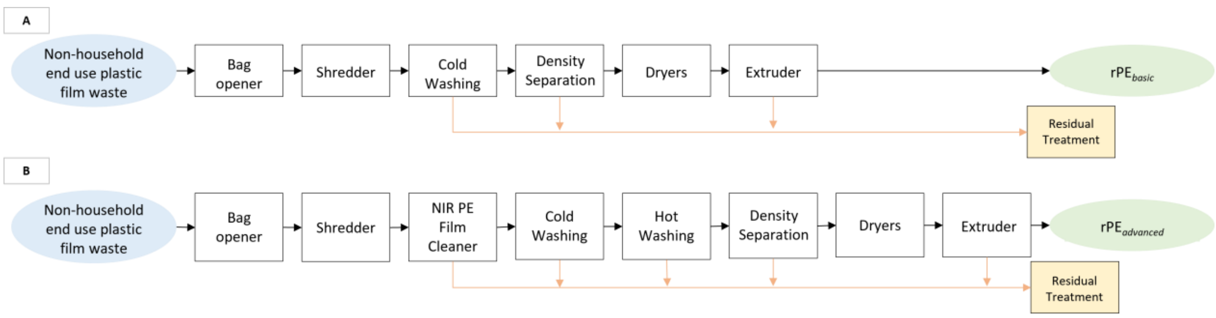


Figure 5.3 Flow diagram of (A) basic recycling process and (B) advanced recycling process considered in this study (adapted from Larrain et al., 2021 and Lase et al., 2022).

5.2.5.2 Separation efficiency

The MFA of non-household end-use plastic in the recycling plant is predicted based on separation efficiency (shown in %), representing the separation of waste items or categories at each recycling equipment (Lase et al., 2022). The summary of the separation efficiency used in this research is presented in Appendix D – section 6. Specifically, the separation efficiency of NIR LDPE Cleaner is averaged from the studies of Lase et al. (2022) and Kleinhans et al. (2021c). As for the cold washing, density separation and extrusion with a single filter and degassing unit, the separation efficiency is averaged from the study by Lase et al. (2022) and Brouwer et al. (2018).

5.2.6 Economic assessment of collecting and recycling non-household end-use plastic film waste

The economic assessment of non-household end-use plastic film waste management demonstrates the difference between the costs incurred by waste collection (i.e., the results of the logistic simulation, see section 5.2.4) and mechanical process, and the revenue from regranulate sales, i.e., $rPE_{basic/advanced}$ (Bashirgonbadi et al., 2022; Sartori et al., 2015; Denne et al., 2007; Hernández et al., 2023). The estimation of capital investment for the mechanical recycling plant follows the approach described by Sinnott and Towler (2019), which is also applied in previous studies (Larrain et al., 2021; Bashirgonbadi et al., 2022; Cimpan et al., 2016). The estimated total capital investment includes the price of individual recycling equipment (in Figure 5.3) and additional procuring, transport, installation and running test of the equipment, engineering and project management, and site infrastructure (i.e., building the recycling plant itself). The total capital investment per equipment is provided in Table D.5 and the economic modeling parameters are provided in Table D.6.

The annual costs of recycling are estimated by calculating the energy costs (i.e., electricity, natural gas, water, and fuel), residual treatment (incl. transport of residue), fixed and variable production costs (i.e., labour, repair and maintenance, depreciation and insurance), and general plant overhead expenses (i.e., office expenses, human resources, finance, legal, information technology, etc.). The energy consumption data are estimated from Cimpan et al. (2016), Larrain et al. (2021), Civancik-Uslu et al. (2021), and WRAP (2009b) studies. In this research, the investment of recycling equipment is depreciated for six years, and the recycling plant for ten years. The annual cost of insurance, repair, and maintenance for the recycling equipment is set to be 1.5% and 4.0% of the total capital investment, respectively (Bashirgonbadi et al., 2022). More information on the energy consumption (i.e., electricity, natural gas, etc.) of each recycling equipment can be found in Table D.7.

The revenue stream of the recycling operation is generated from the regranulates sales (i.e., $rPE_{basic/advanced}$). The range of regranulate prices in this study is taken from the literature (Bashirgonbadi et al., 2022; Larrain et al., 2021; Plastic Portal EU, 2021; Plasticker, 2021). The price of rPE_{basic} is assumed to range from €600/tonne (lower price) to €1,000/tonne (higher price) (with a central price of €800/tonne). On the other hand, it is assumed that $rPE_{advanced}$ can reach up to €1,500/tonne (higher price). The lower price for $rPE_{advanced}$ is set to €900/tonne, and the central price is set to €1,200/tonne. Note that the regranulate prices used in this study are on the higher end of a typical regranulate price shown in the literature (Bashirgondbadi et al., 2022; Larrain et al., 2021). This assumption is made because non-household end-use plastic film waste is typically a homogeneous waste stream containing fewer contaminants or impurities than household film waste recycling (Horodytska et al., 2020; Huysman et al., 2017).

Next to calculating the incurred costs and revenue from regranulate sales, a simplified future cash flow model over the entire project lifetime of a non-household end-use plastic film recycling plant is simulated, as used in previous studies (Lubongo et al., 2022; Fivga and Dimitriou, 2018). The aim of this analysis is to assess the economic feasibility of the recycling plant based on the projected future cash flows and discounts them to present values taking into account discount rate and interest rate (assuming that the capital investment is loaned from the bank) (Lubongo et al., 2022). This also result in a discounted payback period of capital investment (Larrain et al., 2020). For this purpose, S3 – an advanced recycling plant with higher feedstock quality – is selected as a case study, assuming 15 years project lifetime (Larrain et al., 2021; 2020), 15% discount rate (Larrain et al., 2021), and 15% interest rate (Fivga and Dimitriou, 2018).

5.2.7 Estimation of greenhouse gas emission associated with collection and mechanical recycling

The system boundary for carbon footprint calculations (kgCO_2-eq.) starts when the non-household end-use plastic film waste is selectively collected from urban areas (in different collection frequencies, as elaborated in section 5.2.4). Starting with the zero burden assumption of the waste (Civancik-Uslu et al., 2021; Aryan et al., 2021), the selectively collected non-household end-use plastic film waste will be transported to a recycling facility (hub), which is assumed to be located at Port of Ghent–Belgium. The functional unit of this calculation is defined as 1 tonne of $rPE_{basic/advanced}$ produced through mechanical recycling. While comparing the results, the GHG emission of producing virgin PE granulate and incineration (as EoL treatment in *status quo*) is considered as the benchmark, which is also applied in previous studies (Zero Waste Europe, 2020; Quantis, 2020).

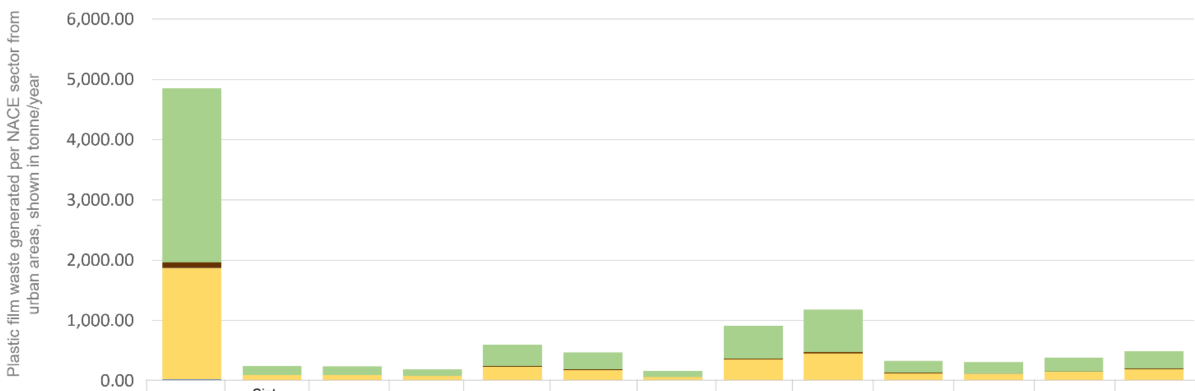
The estimated GHG emission from the selective collection in different frequencies (weekly, fortnightly, monthly) is obtained from logistic simulation in OptiFlow© software (Conundra, 2023), which is estimated to be 0.165 kg CO_2-eq/tkm and benchmark against the Ecoinvent v3.8 database (Table D.9). Note that the emission factor for selective collection is well-to-wheels (WTW), which implies that the GHG includes the emission at fuel production, transport, distribution, and during waste collection from urban areas. The GHG emission from mechanical recycling of non-household end-use plastic film is estimated by calculating the energy usage (electricity, natural gas, and fuel) and assuming that the recycling residues are treated by incineration. The GHG emission (in kg CO_2-eq.) is estimated by multiplying the carbon emission factors with the associated energy usage in mechanical recycling operation (section 5.2.5). Data on energy usage for mechanical recycling is obtained from literature (Cimpan et al., 2016; Larrain et al., 2021; WRAP, 2009b; Civancik-Uslu et al., 2021; Bashirgonbadi et al., 2022), and is available in Table D.8. The emission factors (e.g., kgCO_2-eq/kWh) are obtained from Ecoinvent v3.8 databases used in SimaPro v.9, which is also used in previous studies (Civancik-Uslu et al., 2021; Ügdüler et al., 2020). A list of emission factor datasets can be found in the supplementary information Table D.9, which is based on ReCiPe 2016 (H) Midpoint impact assessment method (Huijbregts et al., 2017).

5.3 RESULTS AND DISCUSSION

5.3.1 Estimated quantity of non-household end-use plastic film waste

Figure 5.4 highlights the estimated total waste quantity of non-household end-use plastic film waste generated in urban areas of Ghent and its 12 neighboring municipalities in Belgium. From Ghent, it is estimated that 4,858 tonne/year of non-household end-use plastic film waste generated annually. From all urban areas considered in this study, it is estimated that more than 10,400 tonne of non-household end-use plastic film waste can be collected. The amount of waste generated per municipality varies between 160 tonne/year in De Pinte (postcode–9840) to 1,182 tonne/year in Deinze (postcode–9800).

The largest waste producer is NACE sector G. Wholesale and retail trade (i.e., 2,887 tonne/year), followed by NACE sector C. Manufacturing (i.e., 1,848 tonne/year). In the studied areas, a relatively low quantity of non-household end-use plastic film waste is generated from NACE sector A. Agriculture, forestry, and fishing (i.e., 24 tonne/year), NACE sector F. Construction (i.e., 95 tonne/year), and NACE sector D. Electricity, gas, steam, and air conditioning supply (i.e., 3 tonne/year). Figure 5.4 shows that NACE sector G accounts for 61% of the total waste generated, followed by NACE sector C with 38%. Together, the two sectors account for 99% of total non-household end-use plastic film waste generation, which aligns with the findings of Kleinhans et al. (2021b). The next chapter discusses the result of logistic simulations (section 5.3.2), mechanical recycling performance (section 5.3.3) and the economic performance of collecting and recycling non-household end-use plastic film waste from urban areas (section 5.3.5).



	Ghent	Sint-Martens-Latem	Melle	Zelzate	Wetteren	Merelbeke	De Pinte	Lokeren	Deinze	Nazareth	Lochristi	Evergem	Eeklo
NACE G	2,888	146	143	117	358	279	95	543	701	198	184	231	294
NACE F	95	5	5	4	12	9	3	18	23	6	6	8	10
NACE D	3	0.2	0.2	0.1	0.4	0.3	0.1	0.6	0.8	0.2	0.2	0.3	0.3
NACE C	1,848	93	91	75	229	179	61	348	450	127	118	148	188
NACE B	1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
NACE A	24	1	1	1	3	2	1	5	6	2	2	2	2

NACE A. *Agriculture, forestry and fishing* NACE B. *Mining and quarry* NACE C. *Manufacturing*
NACE D. *Electricity, gas, steam and air conditioning supply* NACE F. *Construction* NACE G. *Wholesale and retail trade*

Figure 5.4 Estimated quantity (in tonne/year) of non-household end-use plastic film waste from urban areas considered in this study (per NACE sector A–G), excluding NACE sector E *Water Supply, Sewage, Waste Management and Remediation* because it does not fall under the definition of 'non-household end-use plastic' (Kleinhans et al., 2021b).

5.3.2 Non-household end-use plastic film waste collection (weekly, fortnightly, and monthly)

Results of the logistic simulations of non-household end-use plastic film waste selective collection of different frequencies can be found in Table 5.4. More detailed results are provided in Appendix D – section 8. It can be observed from Table 5.4 that the number of stops is higher than the total companies listed in Orbis (2022) databases (in Appendix D, Table D.2 and Table D.3) because typically the garbage trucks need to make more than one trip to collect the waste generated from urban areas. Moreover, the estimated annual distance (in km/year) of selective collection in Ghent (postcode:9000–9070) ranges between 214,044–319,592 km/year, depending on the collection frequencies. The estimated annual distance for the other considered municipalities in this study ranges between 6,924 km/year (monthly collection in De Pinte–9840) to 98,800 km/year (weekly collection in Deinze–9800). Table 5.4 shows that weekly selective collection in Ghent costs €2,396,264 annually (equals €493/tonne collected waste), while fortnightly and monthly selective collection cost €847,470 (equals €174/tonne collected waste) and €310,624 (equals €64/tonne collected waste) annually, respectively. The annual selective collection costs for the other municipalities considered in this study are estimated to range from €14,484 (equals €91/tonne collected waste) for monthly collection in De Pinte to €420,914 (equals €356/tonne collected waste) for weekly waste collection in Deinze.

From Table 5.4 we can observe that the annual distance traveled (in km/year) for fortnightly and monthly collection (on average) is 15% and 26% less than weekly collection, respectively. Consequently, the fortnightly and monthly collection costs (in €/year) are 62% and 81% lower (on average) than the weekly collection costs. In Ghent, the fortnightly and monthly collection costs are 65% and 87% lower than collecting the waste weekly. For the other municipalities, the weekly to fortnightly and monthly collection reduction ranges from 47–68% and 75–87%, respectively. For the companies (waste producers), different collection schemes would mean purchasing different garbage bin sizes. Companies are required to have bigger garbage bins (e.g., 240–2000 liter capacity) when the collection is less frequent (e.g., monthly) compared to a more frequent collection (e.g., 120–240 liter garbage bins for weekly collection) (D'Onza et al., 2016; Boskovic et al., 2016; Greco et al., 2015). Several options are available for companies such as purchasing (€70–€350/piece, depending on the size) or renting the garbage bins (€10–€25/month, depending on the size). Note that larger garbage bins require companies to make more space to store their waste (Greco et al., 2015; D'Onza et al., 2016; Renewi, 2023b).

Table 5.4 Results of the logistic simulations to collect 10,401 tonne/year non-household end-use plastic film waste from urban areas, as elaborated in section 5.3.1.

Municipality (postcode)	**Number of stops**			**Total traveled distance (in km/year)**			**Total annual costs** (€/year)			**Costs per tonne collected waste in each respective municipality** (€/tonne)		
	Weekly	*Fortnightly*	*Monthly*	*Weekly*	*Fortnightly*	*Monthly*	*Weekly*	*Fortnightly*	*Monthly*	*Weekly*	*Fortnightly*	*Monthly*
Ghent (9000–9070)	13,973	13,973	13,999	319,592	256,386	214,044	€ 2,396,264	€ 847,470	€ 310,624	€ 493	€ 174	€ 64
Sint-Martens-Latem (9830)	552	552	554	16,120	14,196	13,068	€ 111,332	€ 35,321	€ 20,904	€ 454	€ 144	€ 85
Melle (9090)	630	630	631	14,456	13,650	10,416	€ 110,032	€ 35,438	€ 18,936	€ 458	€ 148	€ 79
Zelzate (9060)	517	517	517	14,716	11,856	10,608	€ 91,988	€ 33,605	€ 18,390	€ 467	€ 171	€ 93
Merelbeke (9820)	1146	1146	1151	31,720	34,242	24,300	€ 197,912	€ 75,998	€ 40,218	€ 421	€ 162	€ 86
De Pinte (9840)	337	337	337	9,412	7,904	6,924	€ 57,668	€ 25,116	€ 14,484	€ 360	€ 157	€ 91
Lokeren (9160)	1,778	1,778	1,784	74,828	65,728	56,880	€ 359,476	€ 139,802	€ 75,429	€ 393	€ 153	€ 83
Nazareth (9810)	710	710	715	30,680	27,404	26,076	€ 124,072	€ 65,884	€ 30,996	€ 372	€ 198	€ 93
Deinze (9800)	2,000	2,000	2,000	98,800	82,836	68,892	€ 420,914	€ 160,576	€ 83,982	€ 356	€ 136	€ 71
Lochristi (9080)	932	932	932	18,252	14,196	12,600	€ 159,536	€ 58,643	€ 20,088	€ 514	€ 189	€ 65
Evergem (9940)	1,031	1,031	1,037	25,636	18,642	17,376	€ 171,080	€ 60,424	€ 24,564	€ 441	€ 156	€ 63
Eeklo (9900)	1,232	1,232	1,234	51,480	47,892	38,496	€ 240,916	€ 86,060	€ 49,548	€ 486	€ 174	€ 100
Wetteren (9230)	1,386	1,386	1,391	47,372	43,550	37,644	€ 237,848	€ 100,607	€ 45,414	€ 395	€ 167	€ 75
Total	26,070	26,070	26,125	753,064	638,482	537,324	€ 4,679,038	€ 1,724,944	€ 753,577	€ 450*	€ 166*	€ 73*

**The total collection cost per tonne of all non-household end-use plastic film waste, as shown in Figure 5.4.*

5.3.3 Material flow analysis of non-household end-use plastic film waste recycling

The material flow analysis (i.e., Sankey diagram) of non-household end-use plastic film recycling can be found in Figure 5.5. The recycling yield from a basic recycling plant ranges from 77% when processing higher feedstock quality to 61% when processing lower feedstock quality. As for the advanced recycling plant, the recycling yield ranges from 61% to 48%, when processing higher and lower feedstock quality, respectively.

Furthermore, the rPE_{basic} is expected to consist of 89% PE and 11% PP, while the expected composition for $rPE_{advanced}$ is 95% PE and 5% PP (Figure D.26). The non-polyolefin material in the $rPE_{basic/advanced}$ is expected to be less than 1%. From these results, we can observe that the introduction of additional sorting (using NIR PE Film Cleaner) can improve the $rPE_{advanced}$ quality, at the cost of the recycling yield decreases. More detailed information on the mass input-output from basic and advanced recycling in various processing capacities can be found in Appendix D – section 9.

Overall, the estimated mechanical recycling yields for basic and advanced recycling plants are comparable to the reported mechanical recycling yield in previous studies, i.e., ranges between 60–80% (Lase et al., 2022; Brouwer et al., 2018; Horodytska et al., 2020). Moreover, it can be observed that the advanced recycling plant has a lower recycling yield and, subsequently, lower annual $rPE_{advanced}$ production (more in Appendix D – section 9). This is mainly caused by additional (mis)sorting of non-household end-use plastic film waste at NIR PE Film cleaner and a relatively small loss after the hot washing step. However, this can be considered as an unavoidable loss caused by recycling equipment and operation, but a higher quality of regranulate can be expected from such improved recycling processes (Lase et al., 2022; Bashirgonbadi et al., 2022; Horodytska et al., 2020), as also shown in Figure D.26.

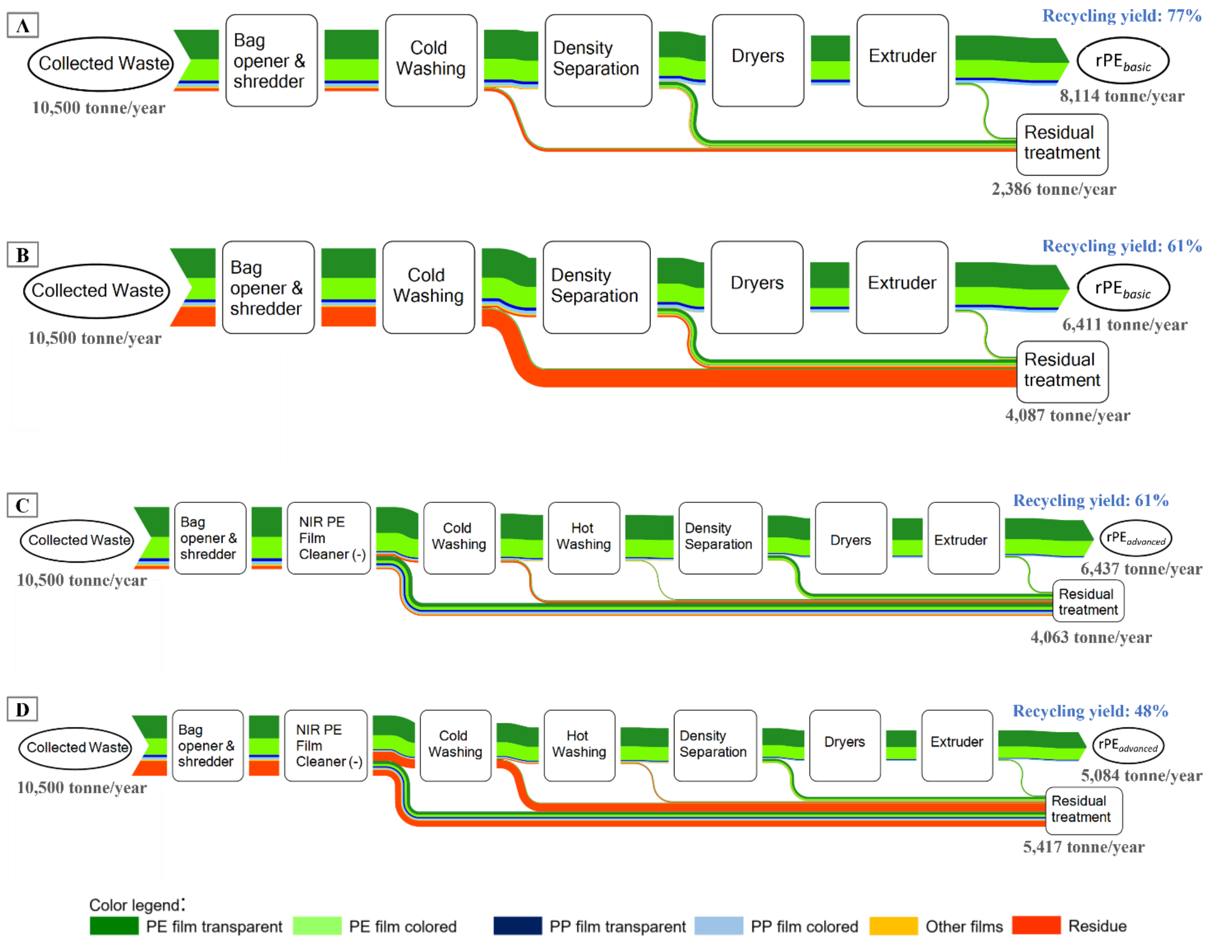


Figure 5.5 Results of material flow analysis of non-household end-use plastic film recycling in different scenarios: S1–basic recycling plant with higher feedstock quality (A), S2–basic recycling plant with lower feedstock quality (B), S3–advanced recycling plant with higher feedstock quality (C), and S4–advanced recycling plant with lower feedstock quality (D). This figure only shows the material flow of 10,500 tonne/year capacity. More information on the other processing capacities (i.e., from 2,500 – 20,500) is available in the Appendix D – section 9.

5.3.4 Economic assessment of mechanical recycling non-household end-use plastic film

5.3.4.1 Breakdown of the capital investment and annual costs of mechanical recycling

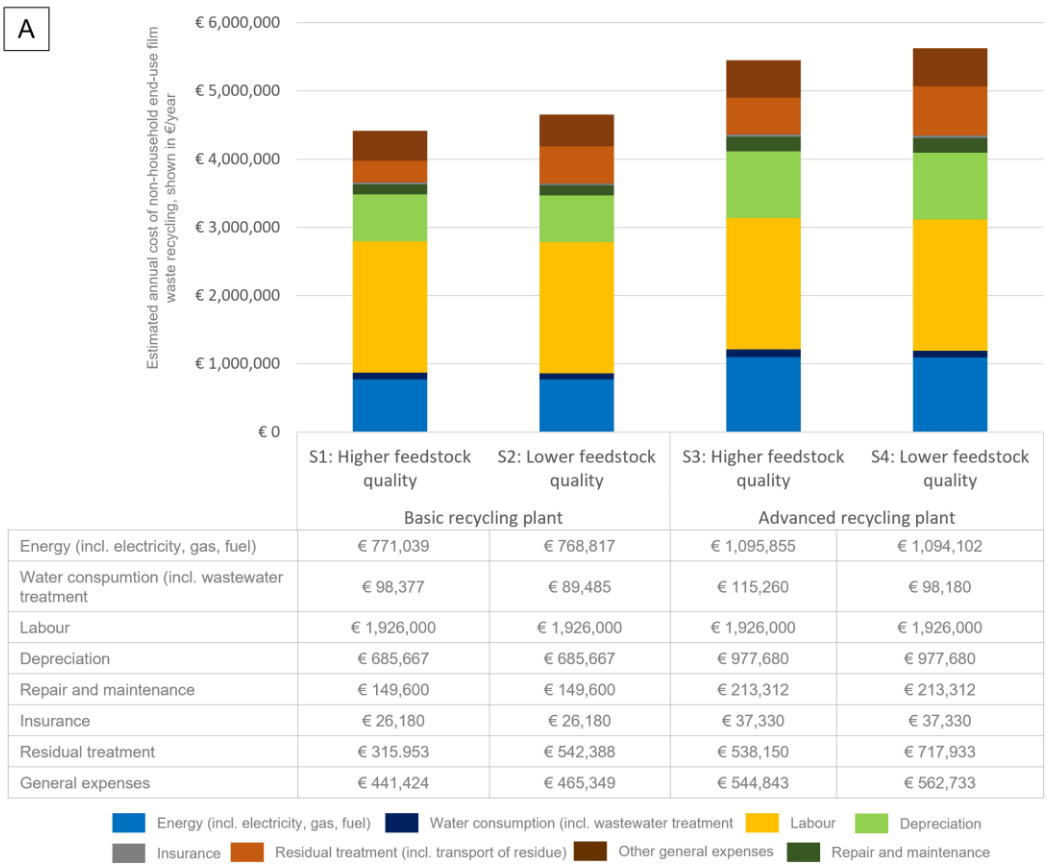
The estimated total capital investment (in Appendix D, Figure D.27) needed to build the recycling plants (basic and advanced layouts) is around €5 million and €7 million respectively, based on the calculations provided in Appendix D – section 6. The investment in washing, extruder, and construction of mechanical recycling plant accounts for 78–82% of the total investment needed. The capital investment in washing and extruder units makes up 28% and 26% of the total investment needed in the basic recycling plant configuration. For the advanced recycling plant, the washing and extruder constitute 39% and 19% of the total investment needed (Figure D.27).

When looking at different processing scales (i.e., ranges between 2,500–20,500 tonne/year), the annual costs of basic recycling plants vary between €4.1–€5.3 million per year. Higher annual costs for advanced recycling plants can be expected, ranging from €4.9 to €6.5 million per year, depending on the scale (available in the Appendix D, Table D.10). Introducing NIR PE Film Cleaner and Hot Washing steps increases the annual costs by 21–23% annually.

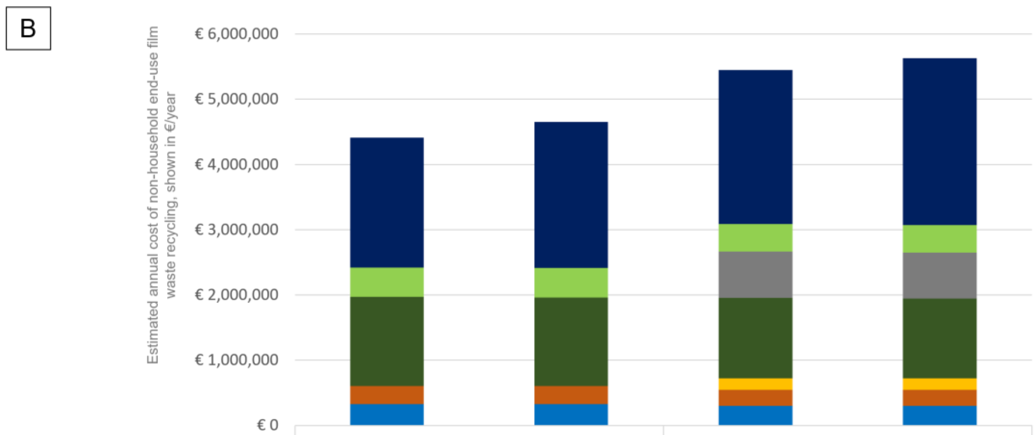
The detailed breakdown of the annual costs of mechanical recycling non-household end-use plastic film with 10,500 tonne/year capacity (fixed capacity, shown as an example) is provided in Figure 5.6. Figure 5.6a shows the annual costs per cost parameter (energy usage, water consumption, etc.), whilst Figure 5.6b shows the annual costs per equipment used in recycling (extruder, washing, etc.). The labour cost, depreciation, and energy usage constitute 35–45%, 15–18%, and 17–20% of annual mechanical recycling costs, respectively (Figure 5.6a). The three cost parameters (labour, depreciation, and energy costs) are estimated to make up 73–77% of the annual costs associated with non-household end-use plastic film waste recycling in this study.

Focusing on the costs per equipment used in the mechanical recycling operation, the cost of recycling plant operations (incl. handling stations, residual treatment and general expenses) accounts for 43–48% of the annual costs (Figure 5.6b). Note that this study assumes that the investment for the recycling plant is depreciated over ten years. Next, the costs associated with washing (cold and hot) and extrusion processes account for 29–36% and 7–10% of the annual costs, respectively. These findings align with the study of Bashirgonbadi et al. (2022) and Larrain et al. (2021), which suggest that washing and extrusion processes are equipment with the highest annual costs in mechanical recycling of polyolefin flexible plastic film.

Looking at different feedstock qualities, we can observe that the annual cost increases by 3–5% (i.e., equals €180,000 to €240,000 annually) when the residue content increases from 5% to 25% (i.e., S1 vs. S2 or S3 vs. S4) (Figure 5.6). For the basic recycling plant (S1–S2), the annual costs of processing 10,500 tonne/year plastic film waste from urban areas increase from around €4.4 to €4.6 million per year. Similarly, the annual costs of processing 10,500 tonne/year of plastic film waste from urban areas through advanced recycling plant (S3–S4) increases from €5.4 to €5.6 million. Such a considerable increase in annual costs is mainly attributed to a higher annual cost of residual treatment (equals €132.5/tonne residue in this study), which is €542,388 and €717,993 in S2 and S4 respectively (light brown bars in Figure 5.6b).



	S1: Higher feedstock quality	S2: Lower feedstock quality	S3: Higher feedstock quality	S4: Lower feedstock quality
Energy (incl. electricity, gas, fuel)	€ 771,039	€ 768,817	€ 1,095,855	€ 1,094,102
Water conspumtion (incl. wastewater treatment	€ 98,377	€ 89,485	€ 115,260	€ 98,180
Labour	€ 1,926,000	€ 1,926,000	€ 1,926,000	€ 1,926,000
Depreciation	€ 685,667	€ 685,667	€ 977,680	€ 977,680
Repair and maintenance	€ 149,600	€ 149,600	€ 213,312	€ 213,312
Insurance	€ 26,180	€ 26,180	€ 37,330	€ 37,330
Residual treatment	€ 315.953	€ 542,388	€ 538,150	€ 717,933
General expenses	€ 441,424	€ 465,349	€ 544,843	€ 562,733



	S1: Higher feedstock quality	S2: Lower feedstock quality	S3: Higher feedstock quality	S4: Lower feedstock quality
	Basic recycling plant		Advanced recycling plant	
Bag opener	€ 327,081	€ 327,081	€ 298,191	€ 298,191
Shredder	€ 274,098	€ 274,098	€ 245,208	€ 245,208
NIR LDPE Cleaner	€ 0	€ 0	€ 176,066	€ 176,072
Cold washing (incl. mechanical and thermal dryers)	€ 1,370,507	€ 1,361,616	€ 1,233,827	€ 1,224,086
Hot washing (incl. mechanical dryers)	€ 0	€ 0	€ 712,351	€ 705,012
Extrusion	€ 449,010	€ 449,010	€ 420,120	€ 420,120
Recycling plant (incl. handling stations, residual treatment & general expenses)	€ 1,993,543	€ 2,241,671	€ 2,362,667	€ 2,558,629

Figure 5.6 Costs breakdown of mechanical recycling (10,500 tonne/year capacity, shown as example) of non-household end-use plastic film waste (A) by cost modeling parameters (energy use, water consumption, depreciation, etc.) and (B) by recycling equipment (incl. residual cost and general expenses that are attributed to the cost of recycling plant).

5.3.4.2 Scale dependency on mechanical recycling

Figure 5.7 presents the net economic balances (i.e., net cost or benefit, in €/tonne rPE$_{basic/advanced}$) of recycling non-household end-use plastic film waste for all scenarios (S1–S4). The green and red lines refer to the net economic balances of the recycling plant (in S1–S4) when the regranulate prices are high and low, respectively. The blue dots refer to the net economic balance of the recycling plant when the central regranulate price is considered. The blue area (between green and red lines) illustrates the potential variations of net economic balances given volatile regranulates prices. More information on the cost and revenue per one tonne rPE$_{basic/advanced}$ production from mechanical recycling in different recycling capacities (ranges from 2,500 – 20,500 tonne/year) is provided in the Appendix D, Table D.10.

The results in Figure 5.7 that recycling non-household end-use plastic film waste benefits from the economy of scale, as shown by an improvement in the net economic balance. When benchmarking our analysis to the low regranulate values (red line in Figure 5.7), a positive economic balance for processing higher feedstock quality via basic and advanced recycling plants (net benefit €56/tonne rPE_{basic} and €54/tonne $rPE_{advanced}$) can be observed from 10,500 tonne/year capacity onwards. However, this holds true only when a higher feedstock quality is maintained (Figure 5.7A and 5.7C). As expected, there is a shift in the overall net economic balance when the feedstock quality gets lower, as shown in Figure 5.7B (for basic recycling plant) and Figure 5.7D (for advanced recycling plant). Selling rPE at higher prices (€1,000/tonne rPE_{basic} and €1,500/tonne $rPE_{advanced}$) is needed to make recycling non-household end-use plastic film waste at 10,500 tonne/year capacity economically viable (net benefit €75/tonne rPE_{basic} and €90/tonne $rPE_{advanced}$, in Figure 5.7B and 7D). This can be explained by the fact that the recycling yield, and subsequently the $rPE_{basic/advanced}$ production, considerably drops when we process waste with lower feedstock quality, as discussed in section 5.3.3. The link between recycling operations and the scale on the economic viability of mechanical recycling of plastics aligns with the previous studies on waste management facilities, which suggest that the economic performance of sorting plants, anaerobic digestion facilities, and mechanical-biological treatment plants becomes more positive as the facilities get bigger (Cimpan et al., 2016; Tsilemou and Panagiotakopoulos, 2006).

The findings shown in Figure 5.7 indicate that collecting non-household end-use plastic film waste from the urban areas considered in this study is crucial to make self-sustaining mechanical recycling operations. Around 10,500 tonne of plastic film waste can be processed from urban areas of Ghent and its neighboring municipalities (Figure 5.4) to make recycling economically viable. A 'partial' collection of the plastic film waste is still possible (i.e., 6,500–8,500 tonne/year), but the regranulates must be sold at higher prices (€1,000 and €1,500/tonne $rPE_{basic/advanced}$) and a high feedstock quality must be maintained, as illustrated in Figure 5.7. Alternatively, it is possible to process household plastic film waste (in different batches) to meet the minimum recycling capacity for economic reasons. However, there is concern about cross-contamination from household waste (typically more contaminated; Horodytska et al., 2020), which can result in a lower $rPE_{basic/advanced}$ quality, and subsequently regranulates price. Furthermore, the net economic balance of collecting and mechanical recycling of non-household end-use plastic film waste from urban areas considered in this study (i.e., 10,500 tonne/year) is discussed in the section 5.3.5.

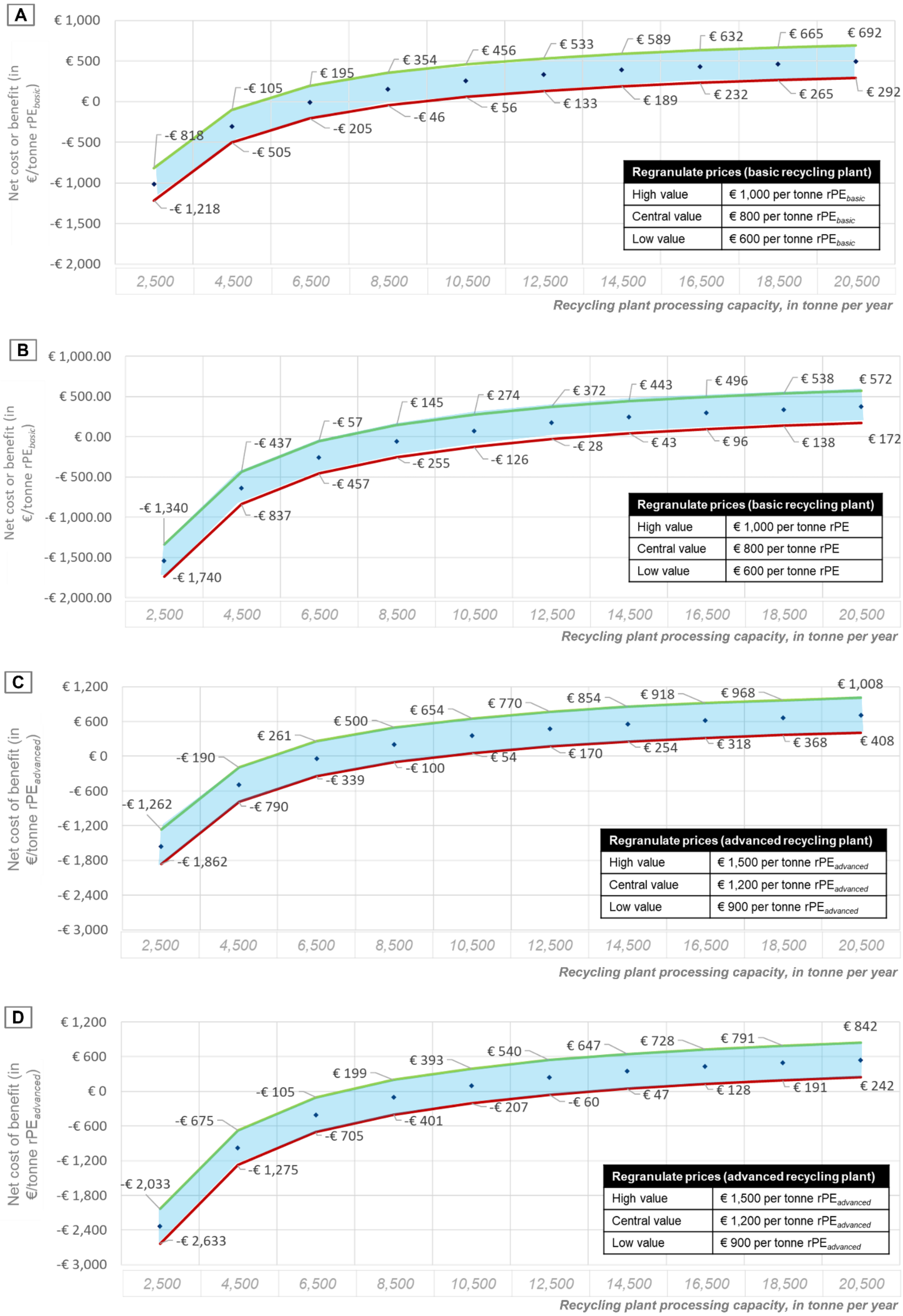
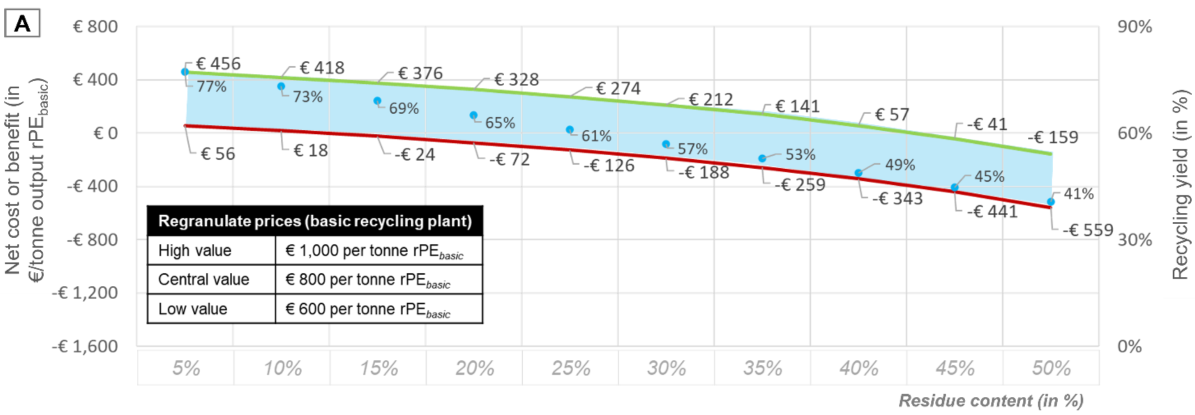


FIGURE 5.7 Estimated net loss or profit (green line, high regranulate price; red line, low regranulate price; blue dots, central regranulate price) of non-household end-use plastic film

waste recycling in S1(A), S2(B), S3(C), and S4(D). The costs and revenue are shown in €/tonne rPE$_{basic/advanced}$ (y-axis) across different recycling plant processing capacities (x-axis, from 2,500 tonne/year up to 20,500 tonne/year capacity). These graphs exclude gate fees. Collection costs are included in Figure 5.9.

5.3.4.3 Dependency of mechanical recycling performance on source separation efficiency

Figure 5.8 shows the results of the sensitivity analysis toward the economic balance (i.e., net benefit or cost, in €/tonne rPE$_{basic/advanced}$) of the basic recycling plant (Figure 5.8a) and advanced recycling plant (Figure 5.8b) when the residue content (in %) in the incoming waste increases. Sensitivity analysis results (Figure 5.8) suggest that the net economic balance of basic and advanced recycling plants can drop up to -€559/tonne rPE$_{basic}$ and -€826/tonne rPE$_{advanced}$, respectively, when the residue content reaches 50%, and regranulates are sold at low prices (€600/tonne rPE$_{basic}$ and €900/tonne rPE$_{advanced}$, red line in Figure 5.8). A similar trend can be observed in the recycling yield, which can drop to 41% and 32% (blue dot in Figure 5.8), when the residue content is high (50%) and the price of regranulates drops simultaneously. We can also observe that rPE$_{basic/advanced}$ should be sold at higher prices (€1,000/tonne rPE$_{basic}$ and €1,500/tonne rPE$_{advanced}$) when the residue content exceeds 30–35%, otherwise mechanical recycling non-household end-use plastic waste is economically unfeasible, even without selective collection cost. Jacobsen et al. (2022) highlight the importance of having well-established waste management systems and waste producers' engagements to improve the purity of source separated plastic waste. Thus, this study can serve as a tool to set the maximum allowable residue content from an economic perspective.



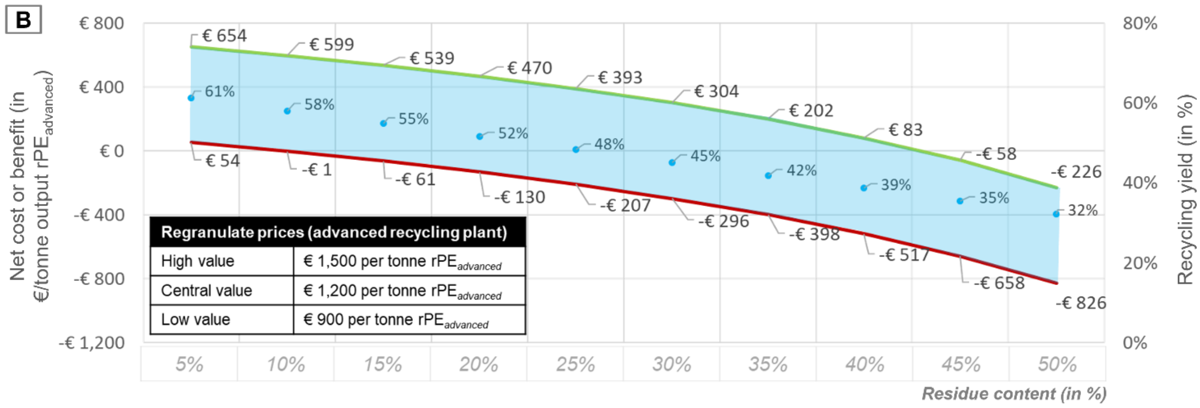


Figure 5.8 Sensitivity analysis towards recycling yield (blue dots) and net cost or benefit of non-household end-use plastic film waste recycling: A) basic recycling plant and B) advanced recycling plant. The green line shows the net cost/benefit of high regranulate prices while red line shows the net cost/benefit of low regranulate prices. In this figure, the recycling plant capacity is fixed at 10,500 tonne/year equals to the non-household end-use plastic waste collected from the urban areas considered in this study.

5.3.5 Cost benefit analysis of collecting and mechanical recycling plastic film waste from urban areas

The estimated annual costs of non-household end-use plastic film waste selective collection (in different frequencies: weekly, fortnightly, or monthly) and mechanical recycling from urban areas in this study (10,500 tonne/year capacity) per tonne rPE$_{basic/advanced}$ is shown in Figure 5.9. Next to that, the revenue and net benefit or cost of producing rPE from non-household end-use plastic film waste in urban areas in this study are also presented in Figure 5.9. Note that the revenue (green bars) and net benefit or cost (blue bars) reflect the central regranulate price, which is €800/tonne rPE$_{basic}$ and €1,200/tonne rPE$_{advanced}$. The error bars shown in Figure 5.9 indicate the potential net benefit or cost changes if the rPE$_{basic/advanced}$ price drops or rises, as elaborated in section 5.2.6 and in Larrain et al. (2021) study.

As seen in Figure 5.9, viable business case for selective collection and mechanical recycling of non-household end-use plastic film waste from urban areas can be profitable only in a few cases, when assuming no fees are applied to the actors generating the waste. First, waste management can only be profitable when waste is selectively collected fortnightly or monthly, and not weekly, as presented in Figure 5.9. The estimated fortnightly and monthly collection costs range from €90/tonne rPE$_{basic}$ (S1, monthly) to €340/tonne rPE$_{advanced}$ (S4, fortnightly), while the estimated costs or recycling range from €545/tonne rPE_{basic} (S1) to €1,100/tonne $rPE_{advanced}$ (S4). Second, the $rPE_{basic/advanced}$ should be sold at central (€800/tonne rPE_{basic} and €1,200/tonne $rPE_{advanced}$) or higher prices (€1,000/tonne rPE*basic* and €1,500/tonne $rPE_{advanced}$), and high-quality feedstock should be maintained Figure 5.9). When the waste composition for the waste collection worsens (S2 and S4 in Figure 5.9), selective collection and recycling non-household end-use plastic film waste is economically feasible only when the rPE is sold at a higher price (€1,000/tonne rPE_{basic} and €1,500/tonne $rPE_{advanced}$). Overall, the total costs of selective collection (fortnightly or monthly) and mechanical recycling on non-household end-use plastic film from urban areas are estimated to range from €635/tonne rPE_{basic} (S1, monthly) to €1,445 per tonne $rPE_{advanced}$ (S4, fortnightly), while the net benefit ranging from €5/tonne rPE_{basic} to €537/tonne $rPE_{advanced}$.

Furthermore, related to the business case for non-household end-use plastic film waste, the CBA result suggest that it is economically unfeasible to make profit from weekly waste collection, even when the $rPE_{basic/advanced}$ is sold at higher price (€1,000/tonne rPE_{basic} or €1,500/tonne $rPE_{advanced}$), as shown in Figure 5.9. However, Figure D.28 in the Appendix D indicates that mechanical recycling plant becomes more cost-effective as more waste is processed (capacity increases) with an overall a cost reduction of about 41–43%. The annual cost per tonne rPE_{basic} in S1 drops from -€544/tonne to -€308/tonne as the waste processed increases from 10,500 to 20,500 tonne/year. Similarly, the annual cost per tonne $rPE_{advanced}$ in S3 drops from -€846/tonne to -€492/tonne as the capacity increases (Figure D.28). Further research is needed to develop a business case for weekly collection depending on the total plant capacity and gate fees. As the capacity increases garbage trucks need to travel more distance and collect more waste to supply waste feedstock for recycling, in which the increase of additional collection cost would mainly depend on (i) type of business activity (NACE sector), (ii) business density, (iii) waste composition, and (iv) waste quantity in the new municipality or region(s). Next to this, the collection scheme would also depend on the desire and general behavior of the businesses to agree on a less frequent collection, which would mean they have to store the waste longer to increase the economic feasibility of the whole system. These behavioral aspects are subjected to future research.

The CBA of selective collection and recycling waste from urban areas suggests that financial instruments are needed in many scenarios to support the recycling chain. For example, a positive economic balance and viable business case can only be achieved when the

$rPE_{basic/advanced}$ is sold at higher price if the residue content gets higher (25w%), as shown in S2 and S4 (Figure 5.9). This can be achieved when the market is 'forced' to use recycled content (e.g., by minimum recycled content target; European Commission, 2022a), and non-household waste can play a crucial role because of its homogenous composition, at least per type of business activity (NACE code classification) (Horodytska et al., 2020; 2018; Huysman et al., 2017). However, as a precautionary action, especially when regranulate (or plastic in a broader sense) price drops, financial support for waste operators (e.g., recyclers) should be established, for example by applying gate fees or EPR scheme (fees) (Bening et al., 2021). Furthermore, the CBA results (Figure 5.9) also indicate that viable business case of recycling non-household end-use plastic film waste rely upon good source separation by actors generating the waste. In this sense, giving financial incentives to companies can be used as an interesting option to ensure a proper separate waste collection at source (e.g., €30/tonne as done by Valipac, 2022). Several studies also suggest that financial incentive is one of the enablers of stakeholders' participation to do a source separation by companies in urban areas (Jacobsen et al., 2022; Klotz et al., 2022; Kleinhans et al., 2021c; Marques et al., 2014; Da Cruz et al., 2014; 2012). This way, the feedstock quality and the required (minimum) quantity can be achieved to ensure viable business case. Yet, appropriate measurements should be sought to analyze (and monitor) the waste quality (as feedstock to recycling facility) per actor generating waste, in which artificial intelligence technology could play a role here in the future.

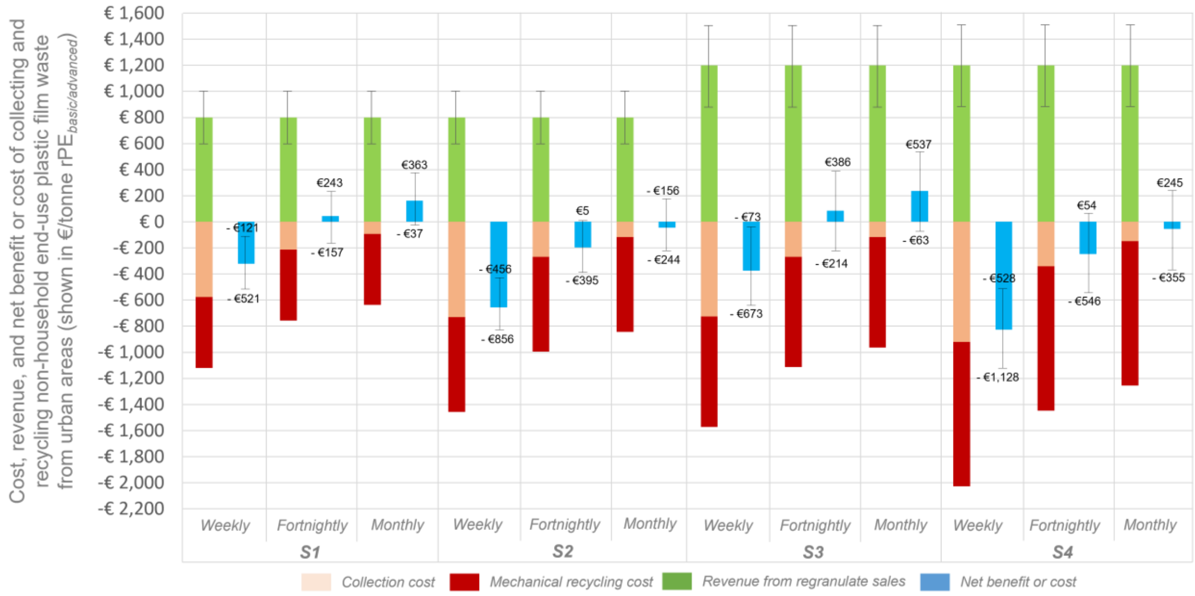
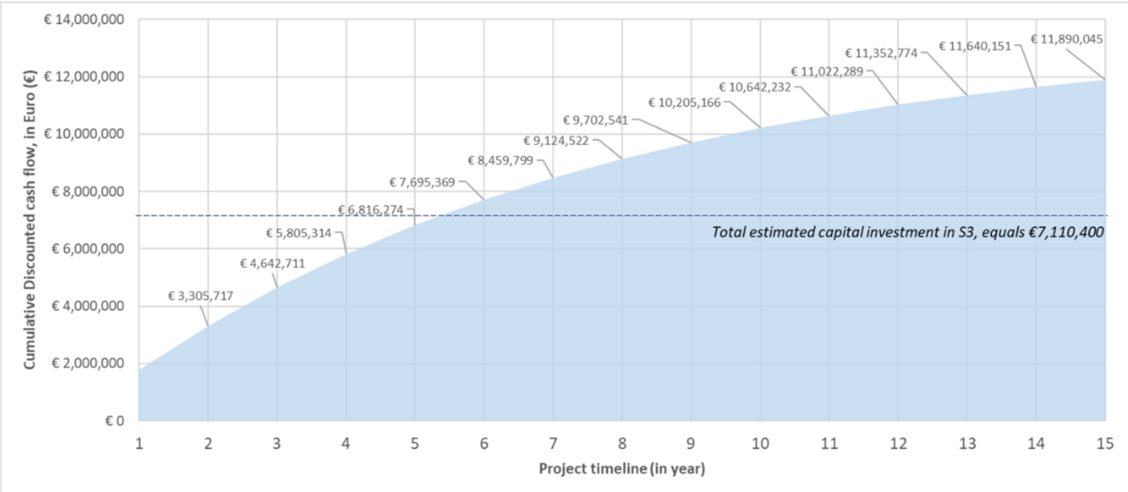


Figure 5.9 Cost, revenue, and net benefit or cost of collecting (weekly, fortnightly, or monthly collection) and mechanical recycling (10,500 tonne/year, in S1–S4) of non-household end-use plastic film waste from urban areas, shown in €/tonne rPE$_{basic/advanced}$. The blue bar reflects the net benefit or cost from selling rPE$_{basic/advanced}$ at central prices. The error bars indicate potential net benefit or cost changes when rPE$_{basic/advanced}$ is sold at lower or higher prices, as elaborated in section 5.2.6.

5.3.6 Future cash flow modeling: Case study of S3 – an advanced recycling plant with higher feedstock quality

Based on the calculation of repayment of loan interest, the annual costs increases by 18% from around €5.5 million (Table D.11) to around €5.7 million, in which the repayment of loan interest (15% interest rate) is estimated to be €218,312 per year. The projected cumulative cash flow summary over 15 years project lifetime is presented in Figure 5.10, assuming a 15% discount rate. The annual discounted net cash flow (in € per year) is also presented in a table below Figure 5.10. From Figure 5.10 it can be observed that an estimated cumulative discounted cash flow of over €11.8 million could be achieved. Based on the results of this future cash flow modeling, the non-household end-use plastic film recycling operation in S3 is predicted to be economically attractive. Moreover, based on the future cash flow modeling, it can be observed that the payback period could be achieved after 5.5 years, i.e., when the cumulative net discounted cash flow reaches the total capital investment of €7,110,400 for S3 (Figure 5.10). However, the presented discounted net cash flow in Figure

5.10 assumes constant total annual costs (around €5.7 million) and revenue (around €7.7 million) for 15 years. To improve the discounted cash flow modeling results, future research needs to perform future projections on several key parameters such as regranulate price, market demand, energy prices, etc. as demonstrated by Larrain et al. (2021). Moreover, as shown in the sensitivity analyses in Chapter 4 (section 4.3.7) and literature (Larrain et al., 2021; Lubongo et al., 2022; Fivga and Dimitriou, 2018), regranulate prices and energy prices are amongst the most sensitive parameters towards economic model results. Thus, more detailed analyses of these modeling parameters towards future cash flows need to be further investigated.



	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10	Year 11	Year 12	Year 13	Year 14	Year 15
Discounted net cash flow (in € per year)	1,768,174	1,537,543	1,336,994	1,162,603	1,010,959	879,095	764,431	664,722	578,019	502,625	437,066	380,057	330,484	287,378	249,894

Figure 5.10 Cumulative net discounted cash flow over 15 years project timeline for S3 – an advanced mechanical recycling with higher feedstock quality, incl. the annual discounted net cash flow (shown in € per year).

5.3.7 GHG emissions from collecting and mechanical recycling of plastic film waste from urban areas

As visualized in Figure 5.11, the GHG emissions of producing one tonne rPE$_{basic}$ (S1–S2) ranges 1,089–1,433 kg CO2-eq. mainly depending on the selective collection scheme. For every one tonne rPE$_{advanced}$ (S3–S4), the GHG emissions ranges from 2,289–2,761 kg CO2-eq., also depending on the selective collection scheme (Figure 5.10). It can be observed from Figure 5.10 that producing rPE$_{basic/advanced}$ results in 74–79% and 49–56% less GHG emissions compared to virgin PE granulate production plus incineration (5,048 kg CO2-eq/tonne rPE), respectively. Figure 5.11 also presents the breakdown of GHG emissions during waste collection, from the energy consumption, NaOH consumption (during hot washing), and residual treatment. It can be observed that the GHG emissions mainly come from residual treatment (60–70% of the total carbon footprint), followed by energy consumption (23–28%) and the waste collection phase (2–9%). The environmental performance of mechanical recycling of plastic film waste from urban areas through advanced recycling plant can still be improved by minimizing the residue. As shown in Figure 5.5 and discussed in section 5.3.3, the mechanical recycling yields in S3 (61%) and S4 (48%) are relatively low compared to S1 (77%) and S2 (61%).

Finally, when comparing the GHG emissions of different collection frequencies only, it can be observed that GHG emissions of monthly collection is 3–4% lower than weekly and fortnightly collection. When the feedstock quality gets lower (in S2 and S4), it can be observed that the GHG emissions increases by 15–21% (compared to S1 and S3). In S2 and S4, a higher GHG emission is mainly caused the increase of residual treatment by 42% and 25% compared to S1 and S3, respectively (as visualized in Figure 5.5). As illustrated in Figure 5.11, the overall GHG emission from advanced recycling plant (in S3 and S4) is 48–52% higher compared to basic recycling plant (in S1 and S2). However, further research should be performed to assess the substitution rate (and environmental saving) of rPE$_{basic/advanced}$, which have different quality as indicated in Figure D.26. To date, different methods have been investigated in previous studies (Gracia-Gutierrez et al., 2023; Tonini et al., 2022), which requires further analysis on the technical properties (e.g., melt flow index, viscosity, etc.) of rPE$_{basic/advanced}$ (Demets et al., 2021; Huysveld et al., 2022; Uekert et al., 2023).

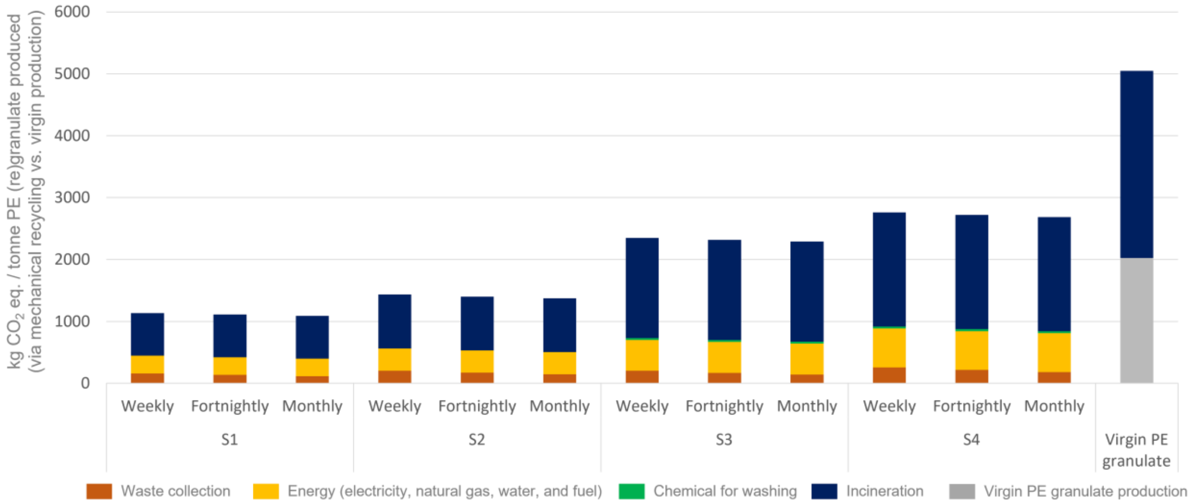


Figure 5.11 Greenhouse gas estimation of collecting and recycling non-household end-use plastic film waste from urban areas considered in this study to produce 1 tonne rPE$_{basic/advanced}$ (in S1–S4) compared to 1 tonne virgin PE granulate production. S1: basic recycling plant with higher feedstock quality, S2: basic recycling plant with lower feedstock quality, S3: advanced recycling plant with higher feedstock quality, and S4: advanced recycling plant with lower feedstock quality.

5.4 CONCLUSION

This study uses the cost-benefit analysis model to develop potential business cases for selective collection and mechanical recycling of non-household end-use plastic film from urban areas. The City of Ghent in Belgium and twelve municipalities nearby are chosen as the case study. This study also analyzes the waste composition and quantity based on real waste sampling combined with data from literature. The logistic simulation results indicate that fortnightly and monthly selective collection is most favorable in terms of costs. The material flow analysis results indicate that the recycling yield ranges from 61% to 77% depending on the plant layouts (i.e., basic vs. advanced recycling plant with extra NIR sorting and hot washing steps). When the residue content is increased up to 25%, the recycling yield can drop to 48–61%.

It is estimated that around €4–€7 million is needed to build the recycling plants, depending on the configurations. Given the economic parameters (adjusted to the Belgian market), the annual costs are expected to be around €4–€6.5 million per year. The costs-benefits analysis shows a positive net economic balance ranging from €5/tonne rPE$_{basic}$ to

€537/tonne $rPE_{advanced}$ (i.e., the recycling chains generate profit) when around 10,500 tonne/year of waste is collected recycled, indicating processing capacity related to the economy of scale. In the positive scenarios, annual costs from waste collection (fortnightly or monthly) range from €90/tonne rPE_{basic} to €340/tonne $rPE_{advanced}$, while mechanical recycling costs range from €545/tonne rPE_{basic} to €1,100/tonne $rPE_{advanced}$. The positive net economic balance can be achieved when the regraulates are sold at €800/tonne rPE_{basic} and €1,500/tonne $rPE_{advanced}$ (depending on the recycling plant layouts and regranulate quality). The modeling results indicate a positive economic balance of selective collection and mechanical recycling non-household end-use plastic film waste from urban areas when i) the high-quality feedstock is maintained and ii) the waste is collected fortnightly or monthly.

Furthermore, the greenhouse gas emissions calculation suggests that minimizing residual streams and maintaining high-quality feedstock from the waste collection are keys to lowering the carbon footprint. Results indicate that the carbon footprint from mechanical recycling non-household end-use plastic film waste can be 49–79% less than current linear economic model of using virgin polyethylene granulate and waste incineration.

Concluding, selective collection and recycling non-household end-use plastic film waste from urban areas can be economically attractive when a few operating conditions are met. To realize this, waste producers, waste operators, and regulators must establish effective waste management systems in the future. Targets and extended producer responsibilities schemes should be established to incentivize non-household end-use plastic waste treatment, especially to sustain plastic recycling operations when regranulate price drop (e.g., due to low oil price). Financial incentives for waste producers to properly separate waste at source can be promoted to ensure feedstock quality and quantity. Nevertheless, given the large quantity of plastic films in non-household waste, society will need this feedstock to achieve its recycling targets. Thus, the developed method in this study can be used in broader European regions (and beyond) to improve plastic circularity, especially in commercial and industrial sectors.

CHAPTER 6: PRELIMINARY ASSESSMENT OF RECYCLED CONTENT AVAILABILITY FOR FLEXIBLE PACKAGING IN EUROPE

Abstract

The new proposed Packaging and Packaging Waste Regulations (PPWR) consists of new mandatory minimum recycled content targets for flexible packaging. According to PPWR, a 35% recycled content for non-contact-sensitive and 10% recycled content for contact-sensitive flexible packaging shall be achieved by 2030. As a response to the proposed PPWR, a preliminary assessment of recycled content availability for flexible packaging (household and non-household) in Europe is conducted. For this purpose, a material flow analysis (MFA) model is developed to trace the fate of flexible packaging throughout its end-of-life treatment in 2030, assuming that the flexible packaging design will be improved, more selective collection for flexible packaging, a better performance of sorting techniques, and improved mechanical and pyrolysis yield. Five scenarios are developed and investigated, consisting of different combinations of mechanical recycling and pyrolysis to reach recycled content targets. Moreover, the capital investment associated with achieving the recycled content targets are estimated, which is estimated based on the MFA model results and economic factors (in € per tonne input). The MFA results suggest that the recycled content targets can be achieved by using mechanical recycling and pyrolysis as complementary techniques to deal with flexible packaging waste. In the most positive scenarios, €7.7 – 8.8 billion of capital investment would be needed to build mechanical recycling and pyrolysis infrastructure, including pretreatment and hydrotreatment for pyrolysis. The MFA results also indicate a trade-off between achieving higher-quality of regranulates to meet 10% recycled content target for flexible packaging (assuming pyrolysis would become a more dominant technique to achieve the target), and annual regranulates production (i.e., quantity of secondary materials).

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Chapter 6

Preliminary Assessment of Recycled Content Availability for Flexible Packaging in Europe

6.1 INTRODUCTION

The plastic (converters) demand in EU 27+3 reached 50.3 Mt in 2021, in which packaging sector represents the largest end-use market for plastic with 19.7 Mt (i.e., 39% of total plastic demand) (Plastics Europe, 2022). Flexible packaging (FP) such as pouches, wrappers, collation shrink, sachets, etc. accounts for half of all plastic packaging demand in EU 27+3 (Hestin et al., 2017; Flexible Packaging Europe, 2021; KIDV, 2020). Around 80% of polymers used in FP are polyolefins (PO), such as (linear) low-density polyethylene ((L)LDPE) and polypropylene (PP). The remaining ~20% of FP is based on other materials including paper, aluminum, polyamide (PA), Polethylene Terephthalate (PET) films. (CEFLEX, 2020; KIDV, 2020; Lase et al., 2023a). Next to household, also non-household FP is a crucial waste stream, accounting for up to half of the FP market share (Hestin et al., 2017). The non-household FP refers to plastic film generated by end-users from commercial, industrial, and institutional activities such as restaurants, retailers, manufacturers, offices, etc. (ISO, 2016; Kleinhans et al., 2021b). Thus, the recycled plastic can be grouped based on its origins, such as Post Consumer Recycled (PCR) plastic or Post Industrial Recycled (PIR) plastic (Ragaert et al., 2017).

To enable a circular economy for both household and non-household FP, several *production-oriented solutions* are favored in the EU 27+3 through for example FP design guidelines (CEFLEX, 2020; RecyClass, 2023). To allow new FP items such as collation shrink films, pouches, wrappers, etc. to be made (partially) of recycled plastic, FP designs should follow both the 'Design from Recycling' and 'Design for Recycling' principles (Vyncke et al., 2018; Ragaert et al., 2018; Ragaert et al., 2020; Berwald et al., 2021; Bashirgonbadi et al., 2022; CEFLEX, 2021). An example is the change from multi- to mono-material multilayer FP applications (e.g., pouches), as demonstrated by Bashirgonbadi et al. (2022), Borealis (2019), Amcor (2021). Some studies also indicate that mono-material FP recycling rate is higher compared to the multi-material (multilayer) FP counterparts (Horodytska et al., 2020; Lase et al., 2022). Moreover, a circular economy for FP can also be enhanced by improving the *end-of-life* (EoL) *treatment-oriented solutions* such as promoting separate waste collection, sorting, and advancing recycling technologies (Ellen MacArthur Foundation, 2016; PRI, 2019). Amongst other options, the EoL solutions include 'improved' mechanical recycling (MR) and chemical recycling options (e.g., pyrolysis) to deal with FP waste. Several improvements to MR include an extensive washing process (e.g., hot washing with chemicals), deinking–delamination, deodorization, improved extrusion (e.g., double melt filtration), or even further polymer dissolution recycling, which can improve the (technical) quality of recycled plastic (Denolf et al., 2023; Lase et al., 2022; Roosen et al., 2021; Kol et al., 2021; Demets et al., 2020). Next to MR, chemical recycling technology such as pyrolysis can also be chosen to recycle FP. Several studies indicate that pyrolysis can be used as an alternative option to deal with FP waste to produce high-quality recycled plastic (Civancik-Uslu et al., 2021; Larrain et al., 2020; Kusenberg et al., 2022d; Huysveld et al., 2022).

To reinforce a circular economy (for FP), the European Commission (EC) has set out several regulations, such as the Packaging and Packaging Waste Directive (PPWD). Recently, a new Packaging and Packaging Waste Regulation (PPWR) proposal was drafted, which is intended to replace PPWD upon approval by the European Parliament (European Commission, 2018; 2022). The EU 27+3 also put a tax on plastic waste that is not recycled (€800/tonne) (European Commission, 2020). The PPWD mandates that 55% of plastic packaging waste to be recycled by 2030, while the PPWR proposed new recycled content targets for new flexible packaging applications. Article 7 of the PPWR proposal set out a minimum percentage of 10% recycled content for contact-sensitive packaging and 35% for non-contact-sensitive packaging by 2030, including FP applications (European Commission, 2022).

The new targets in the PPWR proposal raise questions (in terms of feedstock availability) about the potential future scenarios to achieve the minimum recycled content targets for FP, including the mix of recycling technologies needed to produce recycled plastic (PCR and PIR plastic) for contact and non-contact-sensitive FP applications. Therefore, this study aims to investigate potential scenarios to meet the new proposed recycled content targets in 2030 by performing a prospective material flow analysis (MFA) model, which is elaborated in section 6.1. The MFA model following mass balance principles is selected because it provides transparency of the material flows throughout the defined systems and the claim of recycled content attainments (Tabrizi et al., 2021; Broeren et al., 2022; Lase et al., 2023b). The MFA model is based on the methodologies used in previous studies (Antonopoulos et al., 2021; Eriksen et al., 2020; Kawecki et al., 2018; Lase et al., 2023b). This study considers two recycling options: mechanical recycling (MR) and pyrolysis (incl. pretreatment and coupled with hydrotreatment and steam cracking). Five future scenarios in 2030 are considered in this study, consisting of two pessimistic scenarios and three optimistic scenarios of future EoL treatment for FP. The pessimistic scenarios assume a slower implementation of DfR principles, slower increase of selective collection rates, lower sorting yield for multi-material FP, and lower yields of MR and pyrolysis yield-to-monomers. The optimistic scenarios assume a faster implementation/increase of the abovementioned parameters. These assumptions are discussed in section 6.2.2. To assess and compare the scenarios, five evaluation indicators are selected (in section 6.2.3), namely *end-of-life recycling rate*, *plastic-to-plastic* rate, *plastic-to-chemicals* rate, *plastic-to-fuel* rate, and *recycled content availability* (for contact-sensitive and non-contact sensitive FP), as elaborated by Caro et al. (2023), Lase et al. (2023b), UNEP, 2011 and Perio et al., 2018. In this study, the *recycled content availability* in 2030 is described as the ratio between recycled plastic production from two recycling options (MR and pyrolysis) over the plastic demand for FP in 2030 (Tabrizi et al., 2021; Lase et al., 2023b). Next to modeling and tracing the flows of FP throughout the end-of-life treatment in Europe, the capital investment to build the infrastructure is quantified, focusing on the difference between MR and pyrolysis of FP waste in different scenarios (further elaborated in section 6.2.4).

6.2 MATERIALS AND METHODS

6.2.1 Overall modelling approach

The system boundary for material flow analysis (MFA) of flexible packaging (FP) waste in this study is the European Union (EU) 27+3 (Norway, Switzerland, and the United Kingdom) in 2030, starting from FP waste generation to recycling to recycled polymer (via MR and pyrolysis). The prospective MFA model is built by following four steps: (i) estimation of FP waste quantities (in kilotonne, kt) as input to the MFA model, (ii) building process flow diagrams of potential FP waste management systems in EU 27+3 (Figure 6.1), (iii) applying transfer coefficients (TCs, in %) of each process, and (iv) uncertainty calculations. The polymers considered in the scope of this study are (L)LDPE, PP, and a mix of other multi-material flexible packaging such as PET, PA, paper, and aluminum laminates (Lase et al., 2022). It is assumed that the waste management systems such as the selective collection rates, sorting yield (e.g., at material recovery facilities (MRFs), and MR yield will improve in 2030. The data points of improved TCs in 2030 are obtained from literature. TCs are described as the partitioning of mass input(s) and output(s) (in %) for each process within the defined system boundary. More information about the MFA model is elaborated in section 6.2.

To interpret and compare the MFA results, five indicators are applied, namely (i) recycled content availability, (ii) end-of-life recycling rates (EoL-RR), (iii) plastic-to-plastic (P2P) rate, (iv) plastic-to-chemical (P2C) rate, and (v) plastic-to-fuel (P2F) rate, as suggested in previous studies (Lase et al., 2023b; Broeren et al., 2022; Arena and Ardolino, 2022; Perio et al., 2018; UNEP, 2011) and elaborated in section 6.3.

Finally, the capital investment needed to build new MR and pyrolysis plants to achieve the targets is estimated based on the economic factors found in literature (in €/tonne input) (section 6.2.4). Next, the carbon footprint and saving (in kgCO_2-eq/tonne input) associated with different recycling options (MR and pyrolysis) is estimated based on literature and databases, i.e., Ecoinvent 3.8 (section 6.2.5). Lastly, an uncertainty analysis is calculated and applied to the MFA model results, capital investment calculation, and carbon footprint/saving, which is elaborated in section 6.2.6. The standard deviation of the model results is calculated by assuming a Triangular Distribution (TD) of the TCs and Monte Carlo simulations, as used in Lase et al. (2023b) and Bisinella et al. (2016) studies. Triangular Distribution is selected for this study, following Bisinella et al. (2016) study, mainly because i) statistical analysis and sampling of the selected parameters are not carried out, hence probability is assigned based on data variability found in literature, and ii) expert opinions are involved in determining TCs used in the model (i.e., preferred min, max, and more values).

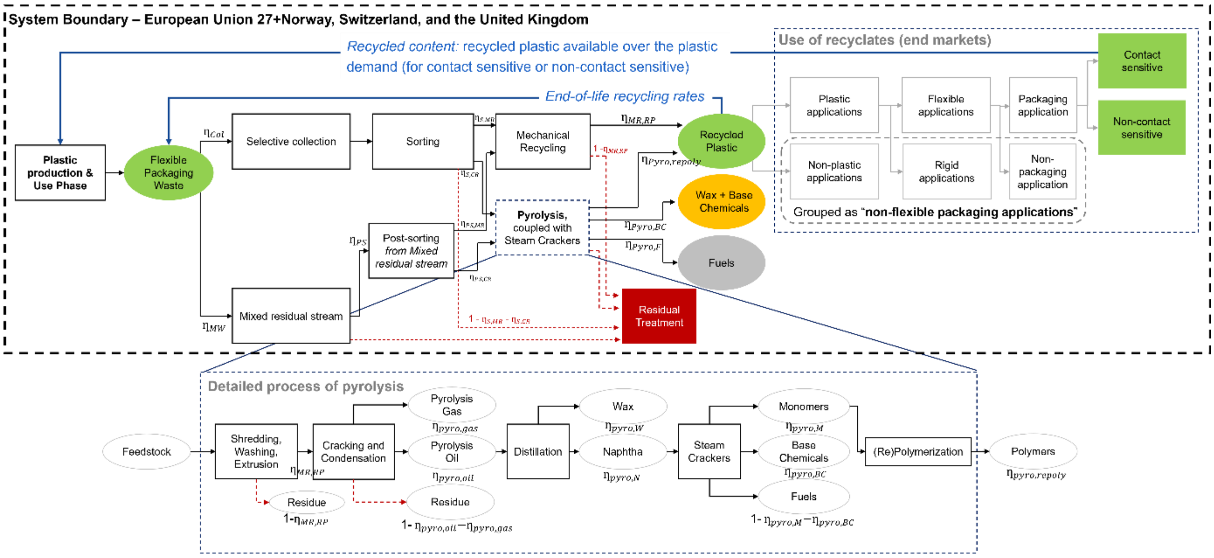


FIGURE 6.1 Process flow diagram of the life cycle of flexible packaging in EU 27+3, including plastic demand in the production and use phase, followed by waste management systems: collection, sorting, and recycling. The potential end market for recycled plastic includes non-flexible packaging and flexible packaging applications (contact- and non-contact-sensitive applications), further elaborated in section 6.2.2.5.

6.2.2 Material flow analysis (MFA) model

The following sub-sections describe the MFA methodology used in this study, including (i) system boundary, (ii) scenarios, (ii) the FP waste composition, (iii) the transfer coefficients, and (iv) end-market applications for recycled plastic from MR and pyrolysis.

6.2.2.1 System boundary

The process flow diagram and system boundary of this study can be found in Figure 6.1. The system boundary considers the plastic demand for FP in EU 27+3 (household and non-household waste), which at the EoL is collected, (post-)sorted, and recycled. The functional unit (FU) used in this study corresponds to the 10,000 kt of mixed FP (incl. moisture and residue content) used in Europe (equals plastic demand in EU 27+3), which is elaborated in section 6.2.2.3. It is assumed that the 10,000 kt of FP use will be discarded after being used by consumers and none of them becomes 'stock' due to the short lifetime of FP (Hestin et al., 2017). Moreover, this study assumes that FP from household is sorted after selective collection as well as post-sorting of household and non-household FP from (mixed) residual waste streams (Figure 6.1). This study assumes that MR and pyrolysis can accept waste from the selective collection or from the post-sorting of household and non-household waste (Kleinhans et al., 2021a; Bashirgonbadi et al., 2022), followed by pretreatment (e.g., washing and sink-float separation) to improve the feedstock quality for MR and pyrolysis recycling (Roosen et al., 2022; Kusenberg et al., 2022e). The transfer coefficients refer to the partitioning of mass input(s) and output(s) for each process in the defined system boundary (Figure 6.1), which are used to model the mass balances (elaborated in section 6.2.2.4).

A typical EoL treatment for household FP waste in EU 27+3 starts with selective waste collection to be sent into MRFs for mechanical sorting (Kleinhans et al., 2021a; Roosen et al., 2022; Antonopoulos et al., 2021). For example, FP waste from households can be selectively collected via P+MD system in Belgium (Roosen et al., 2022) and dual-stream commingled collection in Germany, Denmark, and The Netherlands (Brouwer et al., 2019; Cimpan et al., 2015; Eriksen et al., 2019). At MRFs (i.e., sorting box in Figure 6.1), household FP waste is sorted using a series of sorting steps using drum screens, windshifters, ballistic separators, near-infrared (NIRs), and to some extent manual sorting (Kleinhans et al., 2021a; Cimpan et al., 2016; Picuno et al., 2021). The sorted FP waste is typically baled (e.g., bale rich in PE, PP or PO; Lase et al., 2022) and forwarded into the recycling facility for further processing. A similar sorting strategy can be applied to the collected (mixed) residual waste, in which MFRs can apply so-called post-sorting to recuperate recyclables (incl. FP) from residual waste (Brouwer et al., 2019; 2018). Next to household FP, this study includes non-household FP. As it has received less attention in regulations and (EPR) policies, the non-household FP waste treatment is typically done by commercial or voluntary agreements between the business (i.e., waste producers) and waste management operators (Lase et al., 2023a). For example, waste operators can selectively 'pick' certain high-value items to be recycled in their facilities. FP waste can either be sorted at the location where it is generated (Bendix et al., 2021; Gardner, 2020) (e.g., at retail stores, restaurants, manufacturing facilities, farms, etc.) or at the recycling facility by the waste management company (Bauer et al., 2019; Lase et al., 2023a). Moreover, post-sorting of non-household FP from residual waste stream is commonly done, typically using sensor-based sorting technology as investigated by Kleinhans et al (2022). After the collection and post-sorting (as shown in Figure 6.1), the non-household FP waste can be forwarded directly to MR or pyrolysis plants.

It is assumed that all (post)sorted FP waste will be processed in EU27+3 in 2030 and the extra-EU waste export is assumed to be zero, as suggested by Antonopoulos et al. (2021)

and Lase et al. (2023b). At MR facilities (MR box in Figure 6.1), the sorted bales rich in FP waste from households and selectively collected FP from non-household are shredded, (extra) sorted, washed, and extruded into recycled plastics (Bashirgonbadi et al., 2022; Lase et al., 2022; 2023a; Civancik-Uslu et al., 2021). At pyrolysis plants, the collected and sorted FP is (pre-) treated by means of shredding, (extra) sorting, washing, and extrusion before being fed into the pyrolysis reactor to be recycled back into monomers (Civancik-Uslu et al., 2021; Kusenberg et al., 2022b; 2022c; 2022d; 2022e; Larrain et al., 2021; 2020). This study uses pyrolysis with hydrotreatment and steam cracking followed by polymerization to produce recycled plastic, mainly rPE and rPP, which can be used back for FP applications (Figure 6.1). The 'r' is typically added to the nomenclature referring to different plastic types to indicate it is a recycled plastic (Bashirgonbadi et al., 2022). Finally, the recycled plastic produced from MR and pyrolysis can be used for flexible packaging and non-flexible packaging applications (end market application), which is further elaborated in section 6.2.2.5.

6.2.2.2 Scenarios

Two explorative pessimistic scenarios (**S1*pessimistic*** and **S2*pessimistic***) and three optimistic scenarios (**S3*optimistic***, **S4*optimistic***, and **S5*optimistic***) of future EoL treatment for FP in 2030 are considered in this study. The main differences between pessimistic and optimistic scenarios are summarized in Table 6.1, which is reflected in the key modeling parameters (in %), such as FP waste composition (in section 6.2.2.3), transfer coefficients (in section 6.2.2.4), and potential end market applications (in section 6.2.2.5). The pessimistic scenarios assume a slower implementation of DfR for FP, lower selective collection rate, lower sorting yield for multi-material FP, utilization of conventional MR technology and lower pyrolysis yield-to-monomers. In the optimistic scenarios, the same modeling parameters are assumed to be higher (or better) compared to pessimistic scenarios. Moreover, in pessimistic scenarios, it is assumed that mechanically recycled plastics are only suitable for non-contact sensitive applications (e.g., collation shrink). However, in optimistic scenarios, it is assumed that 35% of the recycled content for contact-sensitive applications (e.g., food packaging) can come from MR (Table 6.1). Lastly, note that in all scenarios (optimistic and pessimistic), the chemically recycled plastic is assumed to be suitable for both non-contact-sensitive and contact-sensitive applications.

S1$_{pessimistic}$ illustrates the improvement of FP waste composition, increased selective collection rate, sorting yield, and conventional MR technology. In this scenario, (post-)sorted FP is treated through MR only, assuming zero capacity for pyrolysis. S2$_{pessimistic}$ explores the waste management systems in which conventional MR and pyrolysis are complementary to reach the recycled content targets by balancing the feedstocks (post-sorted and sorted FP waste) into conventional MR and pyrolysis. Moreover, in S1$_{pessimistic}$ and S2$_{pessimistic}$, it is assumed that sorting yield for multi-material FP is limited, as suggested by Lase et al. (2022) and Kleinhans et al. (2021). It is also assumed that conventional MR in S1$_{pessimistic}$ and S2$_{pessimistic}$ corresponds to shredding, (additional) sorting, cold washing (25–40^{o}), and extrusion with single melt filter (90 – 110 µm mesh size), as suggested by Civancik-Uslu et al. (2021) and Larrain et al., 2021. Lastly, the conventional pyrolysis in S1$_{pessimistic}$ and S2$_{pessimistic}$ is preceded by conventional MR process, as pretreatment, followed by cracking, condensation and distillation (without adding catalyst), as described by Larrain et al. (2020) and Gracia-Gutierrez et al. (2023).

S3$_{optimistic}$ can be perceived as a better waste management system for FP compared to S1$_{pessimistic}$ in 2030. S3$_{optimistic}$ illustrates the improvement DfR for FP, better selective collection rate, higher sorting yield for multi-material FP (compared to pessimistic scenarios), and utilization of advanced MR technologies. In S3$_{optimistic}$, it is assumed that the sorted FP (from MRFs and post-sorting residual waste) is treated through advanced MR only, assuming zero capacity for pyrolysis. S4$_{optimistic}$ and S5$_{optimistic}$ can be perceived as a better waste management systems in 2030 (compared to S2$_{pessimistic}$), in which advanced MR and pyrolysis options can be used to treat (post)sorted FP waste. In S4$_{optimistic}$ and S5$_{optimistic}$, the DfR for FP, selective collection rate, sorting yield for multi-material FP, and advanced pyrolysis yield-to-monomers are further improved (compared to pessimistic scenarios). The sorting yield for multi-material FP in optimistic scenarios is further improved by the implementation of ‘smart’ sorting and packaging such as digital watermarks, chemical tracers or artificial intelligence-based sorting, as elaborated by Soares et al. (2022), NTCP (2023), and Alliance to End Plastic Waste (2023). The MR is also assumed to be improved by hot washing with detergent (> 80^{o}), extrusion with double melt filter (90 – 110 µm, plus 125 µm mesh size), degassing, and deodorization steps, as elaborated by Lase et al. (2022). Similar to the pessimistic scenarios, it is assumed that advanced pyrolysis of FP in optimistic scenarios is preceded by conventional MR process.

Furthermore, it is assumed that in $S4_{optimistic}$, advanced MR option is maximized to meet recycled content targets first, while advanced pyrolysis is used only after advanced MR reaches its technological (and legal) limit. On the other hand, it is assumed that in $S5_{optimistic}$, advanced pyrolysis option is maximized to meet recycled content targets first, while advanced MR is used only after advanced pyrolysis reaches its technological (and legal) limit. Finally, in all scenarios, the mixed residual waste, rejects (from sorting and recycling) are sent for residual treatment in EU 27+3; landfill and incineration (Plastics Europe, 2022).

Table 6.1. Key differences (assumptions) between pessimistic and optimistic scenarios, which is reflected in the FP waste composition (in %, section 6.2.2.3), transfer coefficients (in %, section 6.2.2.4) and potential end market application (section 6.2.2.5) used in MFA model. Acronyms: DfR (Design for Recycling), FP (flexible packaging), MR (mechanical recycling).

Scenario	**DfR for flexible packaging**	**Selective collection rate**	**Sorting yield for multi-material**	**Mechanical recycling technology**	**Yield-to-monomers from pyrolysis**	**[4]Application for mechanically recycled plastic**
Pessimistic: S1*pessimistic* ; MR only S2*pessimistic ; MR and pyrolysis options, maximizing MR*	Slower, **+**	Lower	Lower	[1]Conventional mechanical recycling	Conventional pyrolysis, *lower yield-to-monomers*	• Non-contact-sensitive FP application • Non-flexible packaging application
Optimistic: *S3*optimistic ; *MR only* *S4*optimistic ; *MR and pyrolysis options, maximizing MR* *S5*optimistic ; *MR and pyrolysis options, maximizing Pyrolysis*	Faster, **+++**	Higher	Higher, *same as mono-material*	[2]Advanced mechanical recycling	[3]Advanced pyrolysis, *higher yield-to-monomers*	• Contact sensitive FP application, *35% of recycled content for contact-sensitive application comes from MR* • Non-contact-sensitive FP application • Non-flexible packaging application

[1]*Conventional mechanical recycling refers a typical cold washing and extrusion with single melt filter (90 – 110 µm) (Civancik-Uslu et al., 2021; Larrain et al., 2021).*

[2]*Advanced mechanical recycling refers a typical conventional mechanical recycling with an addition of hot washing (> 80°), extrusion with double melt filter (90 – 110 µm, plus 125 µm mesh size), degassing, and deodorization steps (Bashirgonbadi et al., 2022; Lase et al., 2022).*

[3]*Higher yield-to-monomer from advanced pyrolysis because of the introduction of catalysts or hydrothermal pyrolysis, etc. (Gracia-Gutierrez et al., 2023; Ozoemena and Coles, 2023; Kusenberg et al., 2022e; Eschenbacher et al., 2022).*

[4]*More detail information on the end market application can be found in section 6.2.2.5.*

6.2.2.3 Flexible packaging composition

Table 6.2 provides information on the FP market share considered in this research. Based on Hestin et al. (2017), AMI (2019) and KIDV (2020), it is estimated that the market share of household and non-household flexible packaging is 50–60% and 50–40% of the total FP plastic demand in EU 27+3, respectively. Based on expert judgment, it is estimated that 60–80% of the FP is used for contact-sensitive applications (e.g., food packaging), whilst the remaining 40–20% is used for non-contact-sensitive applications (e.g., collation shrink film, stretch film, etc.).

Table 6.3 provides information on the FP waste composition in pessimistic ($S1_{pessimistic}$ and $S2_{pessimistic}$) and optimistic scenarios ($S3_{optimistic}$, $S4_{optimistic}$, and $S5_{optimistic}$). It is assumed that the DfR guidelines for FP applications will be widely implemented in 2030 in EU 27+3, for example the DfR guidelines proposed by CEFLEX (2020) or RecyClass (2023), which means that more FP is designed as mono-material (e.g., mono-material PP pouch), as demonstrated by Soares et al. (2022), Borealis (2019), and Amcor (2021). In pessimistic scenarios, it is assumed that 75–90% of FP is made following DfR guidelines, while in the optimistic scenario 85–95% of FP is produced following DfR guidelines. Hence, the FP composition in the optimistic scenarios assumes a 50% reduction of multi-material multilayer FP in 2030, i.e., from ~20% in 2019 (Lase et al., 2022; CEFLEX, 2020; KIDV, 2020) to ~10% (mode value) in 2030. The detailed datasets of FP compositions assuming a Triangular Distribution (TD) used as input to the MFA model are provided in Table 6.3. Lastly, It is important to note that multimaterial FP in this study refers to FP that is made of more than one material type, such as PE/PET/PE animal food bag. Moreover, multilayer FP is not always made of multimaterial, for example, multilayer FP that is made of monomaterial as demonstrated by Amcor (2023) for example. However, the adhesive or barrier can still be made of non-polymeric material (e.g., SiO_x), but this is not considered when determining multi- vs. mono-material.

Table 6.2. Summary of the flexible packaging market share used in the MFA model with 10,000 kt as the functional unit (FU). A Triangular Distribution (TD) is assumed in the datasets.

Modeling parameters	**Min.**	**Mode**	**Max.**
Market share			
Household flexible packaging	50%	57%	60%
Non-household flexible packaging	50%	43%	40%
Contact sensitive flexible packaging applications	60%	70%	80%
Non-contact sensitive flexible packaging applications	40%	30%	20%

Table 6.3. Flexible packaging composition (household and non-household) in pessimistic and optimistic scenarios. A Triangular Distribution (TD) is assumed in the datasets. The mono-material PE/PP refers to PE- and PP-based flexible packaging design suitable for recycling following design guidelines (e.g., from CEFLEX, 2020; RecyClass, 2023).

	Pessimistic (S1 and S2)			**Optimistic** (S3, S4, S5)		
	Min.	*Mode*	*Max.*	*Min.*	*Mode*	*Max.*
Mono-material PE	68%	70%	78%	75%	78%	88%
Mono-material PP	7%	10%	12%	10%	12%	7%
Multi-material FP	25%	20%	10%	15%	10%	5%
Total	*100%*	*100%*	*100%*	*100%*	*100%*	*100%*

6.2.2.4 Transfer coefficient

The transfer coefficients (TCs) used to model the flow of selective collection rate, sorting yield, conventional/advanced MR yield (or as pretreatment for pyrolysis), and pyrolysis of FP waste in 2030 are obtained from literature. The full list of TCs used in this study for modeling pessimistic ($S1_{pessimistic}$ and $S2_{pessimistic}$) and optimistic scenarios ($S3_{optimistic}$, $S4_{optimistic}$, and $S5_{optimistic}$) is shown in Table 6.4 and Table 6.5.

In all scenarios, it is assumed that the selective collection rate of FP (μ_{Col}, Figure 6.1) will improve and post-sorting from the (mixed) residual waste occur (η_{PS}, Figure 6.1) (Roosen et al., 2022; Antonopoulos et al., 2021; Brouwer et al., 2018; Picuno et al., 2021). Sorting yield ($\mu_{S,MR/CR}$, Figure 6.1) also improves, assuming that PE- and PP-based sorting occurs, as suggested by Lase et al. (2022) and Bashirgonbadi et al. (2022). The sorting yield for multi-material FP in optimistic scenarios is further improved by the implementation of 'smart' sorting (Soares et al., 2022; NTCP, 2023; Alliance to End Plastic Waste, 2023). The conventional and advanced MR yield (wet weight, $\mu_{MR,RP}$, Figure 6.1) of FP waste is improved, assuming that best practices of MR will be achieved by 2030, as suggested by Antonopoulos et al. (2021). Note that the TCs for conventional and advanced mechanical recycling are assumed to be similar (but different recycled plastic quality), as suggested by Bashirgonbadi et al. (2022) and Lase et al. (2022). However, the MR yield for multi-material multilayer FP is still be lower than the mono-material FP in all scenarios (Horodytska et al., 2019; Horodytska et al., 2018; Lase et al., 2022).

The TCs for conventional and advanced pyrolysis used in the MFA model is provided in Table 6.5. For pyrolysis, the first steps start with conventional MR (as pretreatment) in all pessimistic and optimistic scenarios. The pretreatment stage aims to remove contamination (organic and inorganic residue), potentially some hetero elements (e.g., Chlorine), and non-PO materials (e.g., PET, or PVC) from the waste stream (Kusenberg et al., 2022e). Thereafter, the FP waste is fed into cracking and condensation reactor to produce pyrolysis oil (as main product), in which the TCs in the pessimistic scenarios ($\mu_{Pyro\ Oil}$, Figure 6.1) are obtained from Civancik-Uslu et al. (2021), Larrain et al. (2020), Kusenberg et al. (2022c), and Jeswani et al. (2021). The pyrolysis oil is distilled into naphtha (C_1–$C_{16\text{-}24}$), in which the naphtha is fed into steam crackers, after upgrading techniques such as hydrotreatment, to produce monomers. The TCs for pyrolysis oil distillation in pessimistic scenario ($\mu_{Pyro,N}$, Figure 6.1) and steam cracking of naphtha into monomer ($\mu_{Pyro,M}$, Figure 6.1) are obtained from literature (Civancik-Uslu et al., 2021; Larrain et al., 2020; Zhao et al., 2021; Gholami et al., 2021; Kusenberg et al., 2022a; 2022b; 2022c; 2022d; 2022e). Lastly, the TCs for (re)polymerization of the monomers produced from pyrolysis process ($\mu_{Pyro,repoly}$, Figure 6.1) are obtained from Jeswani et al. (2021) and Lase et al. (2023b) studies. In the optimistic scenario, the yield-to-monomers from advanced pyrolysis is assumed to be further increased because of further improvement in pyrolysis of FP (e.g., by adding catalyst or hydrothermal pyrolysis, etc.) (Gracia-Gutierrez et al., 2023; Ozoemena and Coles, 2023; Kusenberg et al., 2022e; Eschenbacher et al., 2022; Arena and Ardolino et al., 2022; Cardamone et al., 2022). As results, in optimistic scenarios, the yield-to-monomers from advanced pyrolysis of is assumed to increase up to 32–47%, from 15–30% in pessimistic scenarios (Table 6.5)

Table 6.4. The transfer coefficient from selective collection to mechanical recycling (shown in %) used in material flow analysis (MFA) model, assuming a Triangular Distribution (TD) for uncertainty analysis using Monte Carlo simulation with 1,000 iterations.

Stages in waste management systems	**Symbols** (in Figure 6.1)	**Pessimistic scenarios (S1, S2)**			**Optimistic scenarios (S3, S4, S5)**			**Sources**
		Min.	*Mode*	*Max.*	*Min.*	*Mode*	*Max.*	
Collection stage								
Household								
Selective collection	η_{Col}	50%	60%	70%	60%	70%	80%	Hestin et al. (2017); Lase et al. (2023b); Brouwer et al. (2019); Roosen et al. (2022); Antonopoulos et al. (2021)
Residual waste	η_{MW}	50%	40%	30%	40%	30%	20%	
Non-household								
Selective collection	η_{Col}	50%	60%	70%	60%	70%	80%	
Residual waste	η_{MW}	50%	40%	30%	40%	30%	20%	
[1]**Sorting stage**								
Sorting at MRFs for household flexible packaging								
Mono-material								
Sorting yield	$\eta_{S,MR\ or\ CR}$	70%	80%	90%	70%	80%	90%	Kleinhans et al., 2021; Lase et al. (2023b); Lase et al. (2022); Antonopoulos et al. (2021)
Sorting residue	$1-\eta_{S,MR\ or\ CR}$	30%	20%	10%	30%	20%	10%	
Multi-material								
Sorting yield	$\eta_{S,MR\ or\ CR}$	55%	60%	70%	70%	80%	90%	
Sorting residue	$1-\eta_{S,MR\ or\ CR}$	50%	40%	30%	30%	20%	10%	
Post-sorting flexible packaging from mixed residual waste stream								
Post-sorting yield		8%	10%	15%	8%	10%	15%	Brouwer et al. (2018); Picuno et al. (2021); Lase et al. (2023b)
Residual waste to final treatment		92%	90%	85%	92%	90%	85%	
[2]**Mechanical recycling,** conventional and advanced technologies								
Mono-material								
Recycling yield	$\eta_{MR,RP}$	60%	80%	89%	60%	80%	89%	Antonopoulos et al. (2021); Lase et al. (2022); Lase et al. (2023b)
Recycling residue	$1-\eta_{MR,RP}$	40%	20%	11%	40%	20%	11%	
Multi-material								
Recycling yield	$\eta_{MR,RP}$	50%	58%	63%	50%	58%	63%	
Recycling residue	$1-\eta_{MR,RP}$	50%	42%	37%	50%	42%	37%	

[1]Assuming advanced sorting techniques such as digital watermark or artificial intelligence for multi-material FP in the optimistic scenarios (S3, S4, S5) in 2030

[2]Assuming mechanical recycling yield for household and non-household flexible packaging waste (wet weight) in 2030. The yield of conventional and advanced mechanical recycling technologies is assumed to be identical (but with better recycled plastic quality), as suggested by Lase et al. (2022) and Bashirgonbadi et al. (2022).

Table 6.5. The transfer coefficient of conventional and advanced pyrolysis (shown in %) used in material flow analysis (MFA) model, assuming a Triangular Distribution (TD) for uncertainty analysis using Monte Carlo simulation with 1,000 iterations.

Pyrolysis, coupled with steam cracking	**Symbols** (in Figure 6.1)	**Pessimistic scenarios (S1, S2)** *Min.*	*Mode*	*Max.*	**Optimistic scenarios (S3, S4, S5)** *Min.*	*Mode*	*Max.*	**Sources,** for conventional pyrolysis
Pyrolysis – Cracking and condensation		*Conventional pyrolysis*			*[1]Advanced pyrolysis*			
Yield to pyrolysis oil	$\eta_{Pyro,oil}$	70%	80%	89%	75%	80%	89%	Civancik-Uslu et al. (2021); Larrain et al. (2020); Kusenberg et al. (2022a); Jeswani et al. (2021)
Yield to pyrolysis gas	$\eta_{Pyro,gas}$	10%	10%	10%	10%	10%	10%	
Yield to pyrolysis solid residue	$1-\eta_{Pyro,oil}-\eta_{Pyro,gas}$	20%	10%	1%	20%	10%	1%	
Pyrolysis – Distillation								
Yield to naphtha	$\eta_{Pyro,N}$	50%	56%	65%	85%	90%	95%	Civancik-Uslu et al. (2021); Larrain et al. (2020); Kusenberg et al. (2022a)
Yield to wax	$\eta_{Pyro,W}$	50%	44%	35%	15%	10%	5%	
Pyrolysis – Steam cracking								
Yield to monomer	$\eta_{Pyro,M}$	44%	49%	51%	50%	55%	56%	Kusenberg et al. (2022b); Kusenberg et al. (2022c); Zhao et al. (2021); Gholami et al. (2021)
Yield to base chemicals	$\eta_{Pyro,C}$	26%	29%	31%	20%	23%	26%	
Yield to fuel/energy	$1-\eta_{Pyro,M}-\eta_{Pyro,C}$	30%	22%	18%	30%	22%	18%	
Pyrolysis – repolymerization								
Yield to polymer	$\eta_{Pyro,Repoly}$	95%	98%	99%	95%	98%	99%	Lase et al. (2023b); Jeswani et al. (2021)
Yield to residue	$1-\eta_{Pyro,Repoly}$	5%	2%	1%	5%	2%	1%	

[1]The transfer coefficients for advanced pyrolysis is own elaboration of the authors and based on expert judgment

6.2.2.5 Potential end market applications for recycled plastic from MR and pyrolysis

There are two groups of potential end market applications for recycled plastic produced from recycling FP waste that are considered in this study, namely *flexible packaging* and *non-flexible packaging* applications. Within non-flexible packaging options, the recycled plastic can be used to substitute virgin plastic for rigid application (e.g., injection molding; Bashirgonbadi et al., 2022), flexible non-packaging application (e.g., agricultural films; Watkins et al., 2020), or to substitute non-plastic applications (e.g., replacing wooden street bench or road pavement; Huysman et al., 2017) (Figure 6.1). Alternatively, recycled plastic (from MR and pyrolysis) can be used for contact-sensitive FP (e.g., food packaging) or non-contact-sensitive FP (e.g., collation shrink or stretch film) (Figure 6.1), as demonstrated by Bashirgonbadi et al. (2022) and Roosen et al. (2023a).

In the pessimistic scenarios ($S1_{pessimistic}$ and $S2_{pessimistic}$), it is assumed that the recycled plastic produced from conventional MR is only suitable for non-contact-sensitive applications because of the legislative limitations in Europe (De Tandt et al., 2021). However, in the optimistic scenarios ($S3_{optimistic}$, $S4_{optimistic}$ and $S5_{optimistic}$), it is assumed that advanced MR can produce up to 35% of recycled plastic demand for contact-sensitive applications. This can be for example when the recycled plastic is used in between layers (Soares et al., 2022). On the other hand, in all scenarios, the recycled plastic from pyrolysis (conventional and advanced) is suitable for contact and non-contact-sensitive applications because of their assumed virgin-like quality (Kusenberg et al., 2022a; Huysveld et al., 2022).

In all scenarios (see section 6.2.2.2), depending on the assumption towards the possible end market application (as described in previous paragraph), the recycled plastics produced from (conventional and advanced) MR and pyrolysis are used to first meet the 35% recycled content target for non-contact-sensitive applications (as first priority), followed by meeting 10% recycled content target for contact-sensitive applications stated by the PPWR (as second priority) set out by the European Commission, 2022. This is assumption is made because it is relatively easier to meet the (technical and legal) requirements of non-contact-sensitive applications compared to contact-sensitive applications. Lastly, in some scenarios, there is extra mass of recycled plastic available (*surplus material*, if any) after the recycled content targets are met. This the surplus material can be used for either i) flexible packaging applications (boost recycled content beyond the targets) or ii) non-flexible packaging applications.

6.2.3 Evaluation indicators

The five indicators used in this study to compare the MFA model results can be found in Table 6.6. The *end-of-life recycling rate* (EoL-RR, in %) measures the amount (in kt) of waste recycled into recycled plastic ($\mu_{recycled\ plastic}$) and/or base chemicals ($\mu_{base\ chemicals}$) over the amount of waste generated ($\mu_{waste\ generated}$) (UNEP, 2011; Perio et al., 2018). Only recycled plastic and base chemicals are considered on the numerator (i.e., fuel is exempted) to conform to the definition of ‘recycling’ by the European Commission (2018; 2008). The *plastic-to-plastic rate* (P2P), *plastic-to-chemicals rate* (P2C), and *plastic-to-fuels rate* (P2F) (measured in %) measure the amount (in kt) of recycled plastic ($\mu_{recycled\ plastic}$), base chemicals ($\mu_{base\ chemicals}$), and fuels (μ_{fuel}) production over the waste generated ($\mu_{waste\ generation}$), respectively (Broeren et al., 2022; Arena and Ardolino, 2022). Lastly, the recycled content availability (in %) calculates the amount of recycled plastics ($\mu_{recycled\ plastic}$) that are used for contact-sensitive and non-contact sensitive applications over their respective plastic demand (Lase et al., 2023b).

Table 6.6. Five selected evaluation indicators applied to MFA results and their corresponding definitions and formulas, which are also elaborated in previous studies (Lase et al., 2023b; UNEP, 2011; Perio et al., 2018; Broeren et al., 2022; Arena and Ardolino, 2022). Acronyms: CS (contact-sensitive applications), NCS (non-contact-sensitive applications), RC (recycled content availability).

Indicators	Definition	Equation
End-of-life recycling rates (EoL-RR)	The total mass of plastic waste that is converted into secondary materials (recycled plastic and base chemicals) over total plastic waste generation under the definition of '[1]recycling' from the European Commission (2018a; 2008), excluding plastic waste-to-energy (e.g., hydrocarbons)	$EoL - RR = \frac{\mu_{recycled\ plastic} + \mu_{base\ chemicals}}{\mu_{waste\ generated}}$ (Equation 6.1)
Plastic-to-plastic rate (P2P)	The total of plastic waste that is converted into recycled plastic over the total plastic waste generation	$P2P = \frac{\mu_{recycled\ plastic}}{\mu_{waste\ generated}}$ (Equation 6.2)
Plastic-to-chemicals rate (P2C)	The total of plastic waste that is converted into base chemicals (incl. wax) over the total plastic waste generation	$P2C = \frac{\mu_{base\ chemicals}}{\mu_{waste\ generated}}$ (Equation 6.3)
Plastic-to-fuels rate (P2F)	The total of plastic waste that is converted into fuels for energy use over the total plastic waste generation	$P2F = \frac{\mu_{fuel}}{\mu_{reported\ waste}}$ (Equation 6.4)
Recycled content availability, for contact-sensitive applications	The share of recycled plastic uptake to contact-sensitive flexible packaging applications over the plastic demand for contact - sensitive applications (as elaborated in section 6.2.2.5)	$RC_{for\ CS} = \frac{Uptake\ of\ \mu_{recycled\ plastic\ for\ CS}}{\mu_{Plastic\ Demand}\ for\ CS}$ (Equation 6.5)
Recycled content availability, for non-contact-sensitive applications	The share of recycled plastic uptake to non-contact-sensitive flexible packaging applications over the plastic demand for non-contact-sensitive applications (as elaborated in section 6.2.2.5)	$RC_{for\ NCS} = \frac{Uptake\ of\ \mu_{recycled\ plastic\ for\ NCS}}{\mu_{Plastic\ Demand\ for\ NCS}}$ (Equation 6.6)

[1]The definition of 'recycling' as stated in European Commission (2018a; 2008) reports are '*any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations*'. Hence, it (mainly) includes plastic waste recycling back into plastic from mechanical recycling in 2030. When chemical recycling is implemented in 2030, 'recycling' can include plastic waste recycling back into plastic or other materials for other purposes (e.g., base chemicals from pyrolysis for petrochemical industry such as cosmetics, fertilizers, pharmaceutical, etc.), excluding fuel or energy use.

6.2.4 Estimation of capital investment of mechanical recycling and pyrolysis

Next to mass balance and secondary materials availability to meet the recycled content targets, the capital investment to reach the (input) capacity needed for mechanical recycling pyrolysis of FP in all scenarios ($S1_{pessimistic}$ – $S5_{optimistic}$) is estimated in this study. The estimation of capital investment focuses on the mechanical recycling and pyrolysis (incl. pre-treatment) only, i.e., excluding the capital investment for selective collection and sorting, as well as steam cracking and (re)polymerization (because these infrastructure already exist). The economic factors used to calculate the capital investment are presented in Table 6.7, which is shown in €/tonne input to MR or pyrolysis, taken previous studies. The economic factors correspond to the construction of a typical MR facility with 20 kt/year capacity (Larrain et al., 2021; Bashirgonbadi et al., 2022) and pyrolysis facility with 80 – 100 kt/year capacity (Yadav et al., 2023; Larrain et al., 2020). Moreover, the economic factors for conventional and advanced pyrolysis are assumed to be similar (with a relatively broader range of values compared to MR) because of limited data availability. Note that the analysis focuses only on the capital investment to build recycling infrastructure (MR and Pyrolysis) in 2030, while the annual costs associated with the investment (e.g., 10 years depreciation for recycling infrastructure; Larrain et al., 2020) are excluded from the calculation.

Table 6.7. Economic factors to estimate the capital investment needed to build mechanical recycling and pyrolysis in all scenarios, shown in €/tonne input to MR or pyrolysis. A Triangular Distribution (TD) is assumed in the calculation. Acronyms: MR (mechanical recycling), QRP (quality recycling process).

	Economic factor (in €/tonne input)			**Sources**
	Min.	**Mode**	**Max.**	
Conventional MR, or as pretreatment for pyrolysis	500	600	650	Larrain et al. (2021); Bashirgonbadi et al. (2022)
Advanced MR	800	1,000	1,200	Estimated from QRP, Bashirgonbadi et al. (2022)
*Conventional pyrolysis	650	850	1,200	Riedewald et al. (2021);
*Advanced pyrolysis	650	850	1,200	Yadav et al. (2023); Larrain et al. (2020) ; Pryme (2022)

**The capital investment includes product upgrading (e.g., hydrotreatment), excluding steam cracking and polymerization process.*

6.2.5 Uncertainty analysis

In Chapter 6, the MFA model is developed based on large datasets of TCs from literature, in which the selected parameters are subjected to variability that can influence the model results. In this respect, uncertainty analysis is conducted to quantify the uncertainty (error) around the model results due to combined effects of inherent modelling parameters variability or modelling assumptions (Claverul et al., 2012). The uncertainty analysis assumes a Triangular Distribution (TD) of input parameters (in Table 6.2 – Table 6.7). The uncertainty is calculated by systematically propagating the input(s) and/or output(s) uncertainties of the MFA model, i.e., the mass of FP flows (FU: 10,000 kt) and selected indicators (section 6.2.3), which is used in previous studies (Bisinella et al., 2016; Lase et al., 2023b). Uncertainty analysis is also used to calculate the standard deviation (error) of capital investment. The Monte Carlo simulation with 1,000 iterations is used to perform the uncertainty analysis using RiskAMP add-in of Microsoft Excel®, in which the simulation randomly samples a value within each uncertainty distribution and calculates the standard deviation. The standard deviation (error) is shown in kt for the mass flows, percentage (%) for the indicators, and € for capital investment.

6.3 RESULTS AND DISCUSSION

6.3.1 Material flow analysis of flexible packaging waste treatment in 2030

In this study it is estimated that, in 2030, 7,000 ± 230 kt of FP placed on the market is used for contact-sensitive and 3,000 ± 137 kt of FP for non-contact sensitive applications (building from 10,000 ± 298 kt FU, in section 6.2.2.1). To meet 10% recycled content target for contact-sensitive FP and 35% recycled content target for non-contact-sensitive FP, 700 ± 65 kt and 1,050 ± 81 kt of recycled plastic is needed in 2030, respectively. The material flow analyses of FP waste treatment in $S1_{pessimistic}$ – $S5_{optimistic}$ are visualized in Figure 6.2. The summary of evaluation indicators in $S1_{pessimistic}$ – $S5_{optimistic}$ is available in Table 6.8.

In $S1_{pessimistic}$, it is estimated that 4,144 ± 268 kt of recycled plastic will be produced from conventional MR, whilst 5,856 ± 242 kt of FP waste is sent for residual treatment (Figure 6.2A). The estimated EoL-RR in $S1_{pessimistic}$ is 41% ± 3% attributed to recycled plastic production (P2P) from conventional MR (Table 6.8). The recycled plastic production in $S2_{pessimistic}$ is estimated to be 1,707 ± 130 kt, in which 1,050 ± 77 kt comes from conventional MR and 656

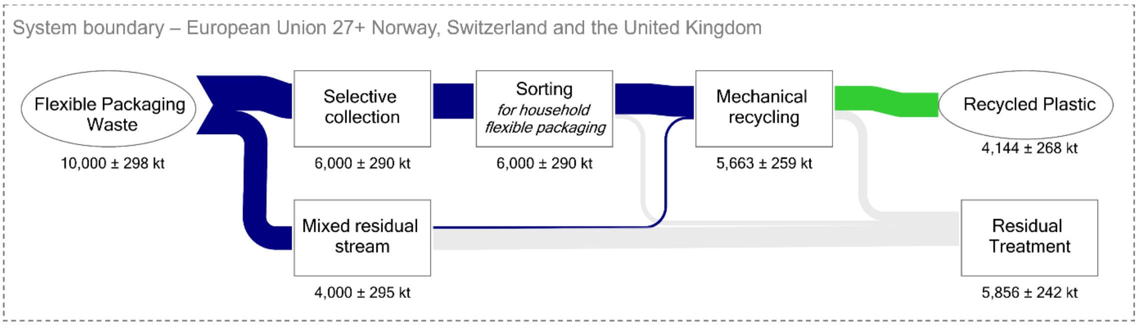
± 70 kt from conventional pyrolysis. Next to recycled plastic, it is estimated that 1,463 ± 147 kt of base chemicals and 637 ± 64 kt of fuels will be produced from the conventional pyrolysis process (Figure 6.2B). As shown in Table 6.8, the EoL-RR of FP in $S2_{pessimistic}$ is estimated to be 32% ± 2%, in which 17% ± 1% is P2P (from conventional MR and pyrolysis) and 15% ± 1% is P2C (from conventional pyrolysis). The P2F in $S2_{pessimistic}$ is estimated to be 6% ± 1% (Table 6.8). Note that around 49% of the fuel production in S2pessimistic (equals 309 ± 31 kt) is auto-consumed as energy source for pyrolysis process (Figure 6.2B). Furthermore, a substantial quantity of waste feedstock (4,228 ± 382 kt) is forwarded from conventional MR to conventional pyrolysis. This can be explained by the fact that the yield-to-monomer from conventional pyrolysis is relatively low (15–30%, Table 6.4), hence more waste feedstock is needed to meet the recycled content target for contact-sensitive FP.

In the optimistic scenario, the amount of recycled plastic production from advanced MR in $S3_{optimistic}$ is estimated to be 4,877 ± 325 kt, which is around 15% higher than $S1_{pessimistic}$. The amount of FP waste sent for residual treatment is 5,123 ± 314 kt in $S3_{optimistic}$, i.e., around 13% lower than in $S1_{pessimistic}$ (Figure 6.2A). As seen in Table 6.8, the EoL-RR in $S3_{optimistic}$ is 49% ± 3% (only from P2P), around 8% higher than in $S1_{pessimistic}$ (41% ± 3%).

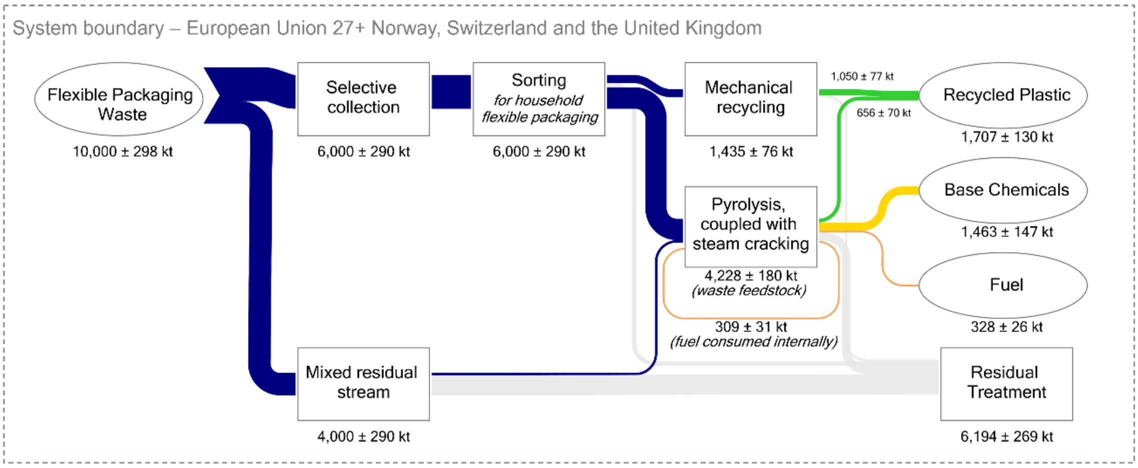
The recycled plastic production in $S4_{optimistic}$ and $S5_{optimistic}$ is estimated to be 2,664 ± 204 kt and 2,051 ± 204 kt, respectively (Figure 6.2D and Figure 6.2E). Out of 2,664 ± 204 kt recycled plastic in $S4_{optimistic}$, 1,295 ± 86 kt comes from advanced MR and 1,370 ± 126 kt from advanced pyrolysis (Figure 6.2D). In $S5_{optimistic}$, 301 ± 21 kt is produced from advanced MR and 1,750 ± 152 kt from advanced pyrolysis (Figure 6.2E). Compared to $S2_{pessimistic}$, the recycled plastic production in $S4_{optimistic}$ and $S5_{optimistic}$ is approximately 56% and 20% higher, respectively. Next to recycled plastic, 894 ± 119 kt and 1,143 ± 157 kt of base chemicals are produced in $S4_{optimistic}$ and $S5_{optimistic}$, respectively, which is approximately 17–38% lower than in $S2_{pessimistic}$. It is estimated that 970 ± 105 kt and 1,240 ± 142 kt of fuel are produced in $S4_{optimistic}$ and $S5_{optimistic}$, respectively. The EoL-RR in $S4_{optimistic}$ and $S5_{optimistic}$ is estimated to be 36% ± 3% and 32% ± 3% (Table 6.8). In $S4_{optimistic}$, 27% ± 2% is P2P (from advanced MR and pyrolysis) and 9% ± 1% is P2C from advanced pyrolysis, while in $S5_{optimistic}$, 21% ± 2% is P2P and 11% ± 2% is P2C from advanced pyrolysis. Compared to the EoL-RR in $S2_{pessimistic}$ (32% ± 2%), the EoL-RR in $S4_{optimistic}$ is approximately 4% higher, while the EoL-RR in $S5_{optimistic}$ is comparable to $S2_{pessimistic}$ (Table 6.8). The P2F in S4optimistic and S5optimistic is estimated to be 10% ± 1%

and 13% ± 1%, respectively (Table 6.8), in which around 37% is auto-consumed as energy source for pyrolysis process (Figure 6.2D and Figure 6.2E).

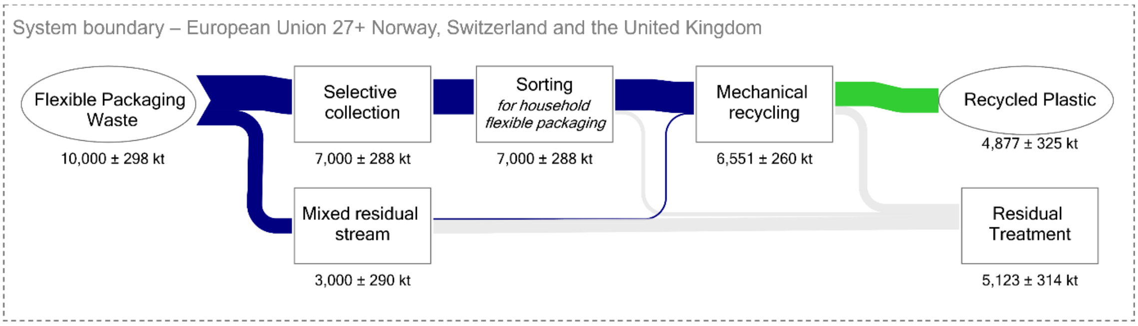
[**A**] Aggregated flows of flexible packaging waste throughout waste management systems in S1$_{pessimistic}$



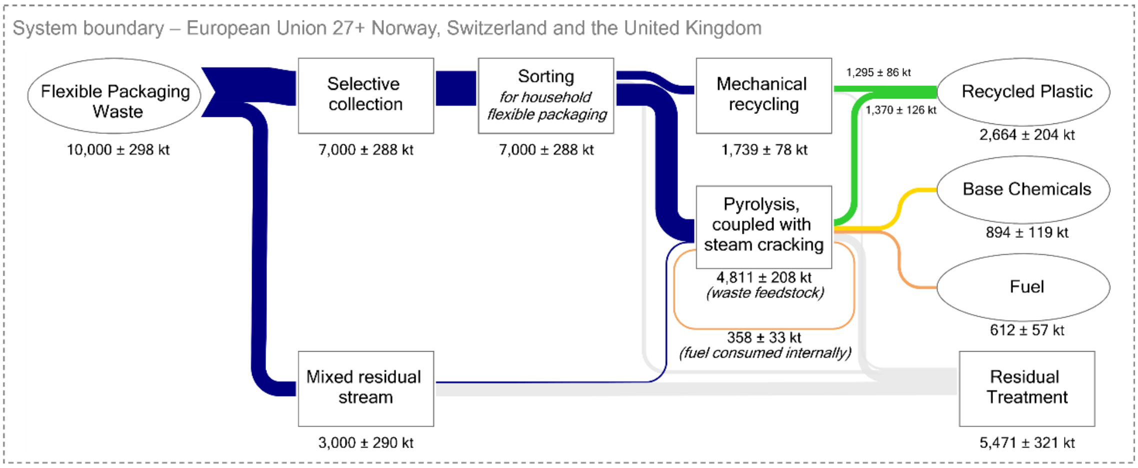
[**B**] Aggregated flows of flexible packaging waste throughout waste management systems in S2$_{pessimistic}$



[**C**] Aggregated flows of flexible packaging waste throughout waste management systems in S3$_{optimistic}$



[**D**] Aggregated flows of flexible packaging waste throughout waste management systems in S4$_{optimistic}$



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[**E**] Aggregated flows of flexible packaging waste throughout waste management systems in S5$_{optimistic}$

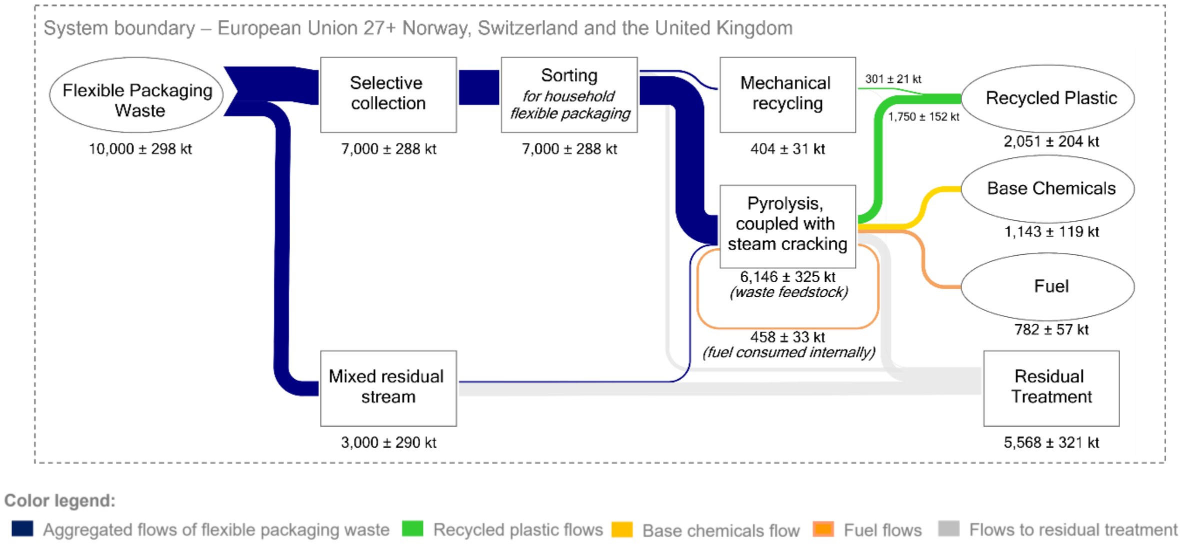


FIGURE 6.2 Material flow analysis of flexible packaging waste treatment in 2030 in S1$_{pessimistic}$ (A), S2$_{pessimistic}$ (B), S3$_{optimistic}$ (C), S4$_{optimistic}$ (D), and S5$_{optimistic}$ (E).

Table 6.8. Summary of the evaluation indicators applied to compare the MFA results in this study. Abbreviations: EoL-RR (end-of-life recycling rate), P2C (plastic-to-chemical), P2F (plastic-to-fuel), P2P (plastic-to-plastic).

Indicators	S1*pessimistic*	S2*pessimistic*	S3*optimistic*	S4*optimistic*	S5*optimistic*
P2P	41% ± 3%	17% ± 1%	49% ± 3%	27% ± 2%	21% ± 2%
P2P from MR	41% ± 3%	11% ± 1%-	49% ± 3%	13% ± 2%	3% ± 1%
P2P from CR	-	7% ± 1%	-	14% ± 2%	17% ± 2%
P2C	-	15% ± 2%	-	9% ± 1%	11% ± 2%
P2F	-	6% ± 1%	-	10% ± 1%	13% ± 1%
[1]EoL-RR	41% ± 3%	32% ± 3%	49% ± 3%	36% ± 3%	32% ± 3%

[1] EoL-RR considers only P2P and P2C because P2F recycling does not conform to the definition of 'recycling' in WFD (European Commission, 2018; 2008).

6.3.2 Recycled content availability for flexible packaging, and surplus materials, in 2030

The potential use (end market application) of recycled plastic produced in S1*pessimistic* – S5*optimistic* can be found in Figure 6.3, while the summary of recycled content availability (attainment) for contact-sensitive and non-contact-sensitive applications is provided in Table 6.9. Figure 6.3 shows the plastic demand for contact sensitive (7,000 ± 293 kt, dark brown bar) and non-contact sensitive FP (3,000 ± 288 kt, light brown bar), assuming total 10,000 kt ± 478 kt of FP demand in 2030. Figure 6.3 also shows the quantity of recycled plastic (from MR and pyrolysis) that is used in different applications: non-contact-sensitive FP (light and dark blue bars), contact-sensitive FP (light and dark green bars) as well as the surplus materials (light and dark grey bars). Note that the in this research, the recycled plastic (from MR and pyrolysis) is used only to meet recycled content targets: 10% for contact sensitive and 35% for non-contact sensitive FP applications (European Commission, 2022). The surplus material (extra mass) can be used to i) boost the recycled content target (following the technical and legal limitation, section 6.2.2.5) or ii) non-flexible packaging applications (e.g., rigid application, wooden park bench, agriculture film, etc.).

It can be observed that the 35% recycled content target for non-contact-sensitive FP can be met in S1*pessimistic* (Table 6.9) by using 1,050 ± 71 kt of recycled plastic from conventional MR, while the contact-sensitive target is not met because of the absence of pyrolysis technology and the assumption that recycled plastic from conventional MR in S1*pessimistic* is unsuitable for contact-sensitive applications (as elaborated in Table 6.1 and section 6.2.2.5

and). Still, 3,094 ± 210 kt of surplus recycled plastic is available for non-flexible packaging applications or to increase recycled content of non-contact-sensitive FP (Figure 6.3). From Figure 6.3, it is also estimated that achieving 100% recycled content for non-contact-sensitive FP is possible by using 3,000 kt of recycled plastic from conventional MR in $S1_{pessimistic}$.

Figure 6.3 also indicates that, in $S2_{pessimistic}$, 35% recycled content target for non-contact-sensitive FP can be achieved (by using 1,050 ± 76 kt recycled plastic from conventional MR), while the recycled content for contact-sensitive FP is slightly below the 10% target, i.e., it is estimated that 656 ± 68 kt recycled plastic from conventional pyrolysis can reach 9% ± 1% recycled content for contact-sensitive FP (Figure 6.3). However, as indicated in Figure 6.3, no surplus recycled plastic can be used for non-flexible packaging applications, which implies that a considerable amount of surplus recycled plastic from conventional MR in $S1_{pessimistic}$ (3,084 ± 210 kt) must be supplied by virgin plastic production (or non-plastic, if the recycled plastic replaces non-plastic application such as wooden bench park).

In $S3_{optimistic}$ (Figure 6.3), it can be observed that 1,050 ± 69 kt of recycled plastic from advanced MR can be used to meet 35% recycled content target (for non-contact-sensitive FP). Next to that, 246 ± 16 kt of recycled plastic from advanced MR can be used for contact-sensitive FP, which leads to 4% ± 1% recycled content for contact-sensitive FP (Figure 6.3, Table 6.9). In $S3_{optimistic}$, it is estimated that 3,582 ± 238 kt of surplus recycled plastic is available for non-flexible packaging applications or to further increase recycled content for non-contact-sensitive FP (up to 100% recycled content is possible). The MFA results in $S3_{optimistic}$ indicate that the surplus recycled plastic increase by approximately 14%, from 3,054 ± 210 kt in $S1_{pessimistic}$ to 3,582 ± 238 kt in $S3_{optimistic}$ (Table 6.9).

The 35% recycled content target for non-contact-sensitive FP and 10% recycled content target for contact-sensitive FP could be achieved in $S4_{optimistic}$ and $S5_{optimistic}$ (Table 6.9). In $S4_{optimistic}$, the 35% recycled content target is achieved by using 1,050 ± 74 kt of recycled plastic from advanced MR, while the 10% recycled content target is achieved by using 246 ± 16 kt of recycled plastic from advanced MR and 455 ± 43 kt of recycled plastic from advanced pyrolysis (Figure 6.3). In $S5_{optimistic}$, 1,050 ± 74 kt of recycled plastic from advanced pyrolysis is used to meet non-contact-sensitive target, while 700 ± 59 kt recycled plastic from advanced pyrolysis is used to meet contact-sensitive target. Lastly, in $S4_{optimistic}$ and $S5_{optimistic}$, it is estimated that 914 ± 87 kt and 301 ± 10 kt of surplus recycled plastic from advanced pyrolysis

(Figure 6.3) can be used for either i) non-flexible packaging application, ii) contact-sensitive FP, or non-contact-sensitive FP.

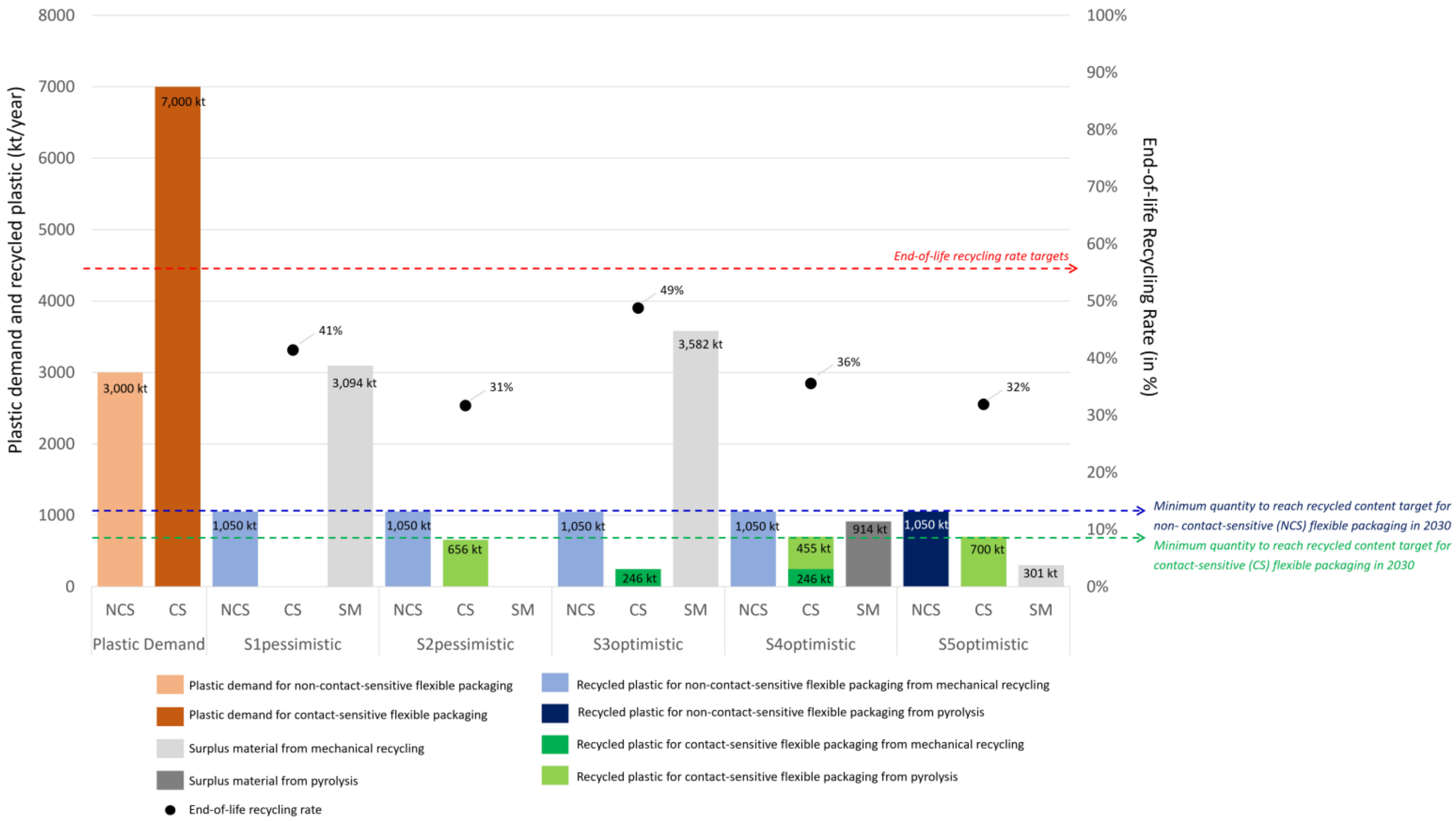


FIGURE 6.3 Potential use of recycled plastic from mechanical recycling and pyrolysis in 2030: S1$_{pessimistic}$, S2$_{pessimistic}$, S3$_{optimistic}$, S4$_{optimistic}$, and S5$_{optimistic}$, including the end-of-life recycling rates (black dots). The values are rounded in kilotonne, including the standard deviations. The blue dashed line corresponds to the minimum quantity to reach recycled content target for non-contact-sensitive flexible packaging. The green dashed line corresponds to the minimum quantity to reach contact-sensitive flexible packaging. The red dashed line corresponds to the end-of-life recycling rates targets. Acronyms: NCS (non-contact-sensitive), CS (contact-sensitive), SM (surplus material), kt (kilotonne).

Table 6.9. Summary of the recycled content availability and surplus recycled plastic in $S1_{pessimistic}$ – $S5_{optimistic}$. The end market application for surplus recycled plastic (in kt/year) is elaborated in section 6.2.2.5. Values are rounded, including the standard deviations (in %).

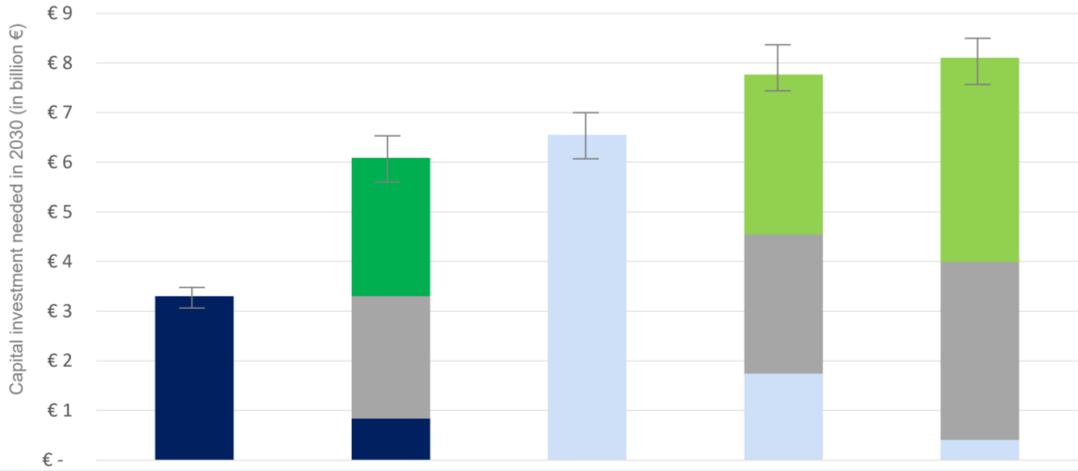
	*$S1_{pessimistic}$	$S2_{pessimistic}$	*$S3_{optimistic}$	**$S4_{optimistic}$	**$S5_{optimistic}$
Non-contact sensitive	35% ± 4%	35% ± 4%	35% ± 4%	35% ± 4%	35% ± 4%
Contact sensitive	-	9% ±1%	4% ±1%	10% ±1%	4% ±1%
Surplus recycled plastic (in kt/year)	3,094 ± 210	-	3,582 ± 238	914 ± 87	301 ± 10

** The surplus recycled plastic produced from (conventional and advanced) MR, only suitable for non-flexible packaging and non-contact sensitive flexible packaging applications (section 6.2.2.5).*
*** The surplus recycled plastic produced from advanced pyrolysis, suitable for non-flexible packaging, non-contact sensitive flexible packaging, and contact-sensitive flexible packaging applications (section 6.2.2.5).*

6.3.3 Capital investment needed to build mechanical recycling and pyrolysis plant to deal with flexible packaging in 2030

Figure 6.4 summarizes the estimated capital investment needed in 2030 in $S1_{pessimistic}$ – $S5_{optimistic}$. As seen in Figure 6.4, it is estimated that €3.3 ± 0.2 billion would be needed to build conventional MR to process 5,663 ± 259 kt of FP waste in $S1_{pessimistic}$. The estimated capital investment in $S2_{pessimistic}$ where conventional MR and pyrolysis is needed reach total €6.1 ± 0.4 billion in 2030, which is approximately 1.8 times higher than in $S1_{pessimistic}$. Out of €6.1 ± 0.4 capital investment needed in $S2_{pessimistic}$, €3.3 ± 0.2 billion is used to build conventional MR with 1,435 ± 76 kt capacity and pretreatment for pyrolysis with 4,288 ± 180 kt capacity. The remaining €2.8 ± 0.4 billion is used to build conventional pyrolysis reactor with 3,094 ± 235 kt capacity (Figure 6.4).

In the more $S3_{optimistic}$ scenario, it is estimated that total €6.6 ± 0.5 billion capital investment is needed to build advanced MR with 6,551 ± 260 kt capacity. The capital investment in $S3_{optimistic}$ (€6.6 ± 0.5 billion) is approximately 2.0 times higher compared to $S1_{pessimistic}$ (€3.3 ± 0.2 billion), while the input capacity increases from 5,663 ± 259 kt to 6,551 ± 260 kt (i.e., approximately 16% more capacity needed in $S3_{optimistic}$ compared to $S1_{pessimistic}$) (Figure 6.4). The capital investment in $S4_{optimistic}$ is estimated to reach total €7.7 ± 0.4 billion, in which €1.7 ± 0.1 billion is used to build advanced MR (1,739 ± 78 kt capacity), €2.8 ± 0.2 billion for pretreatment for waste feedstock to advanced pyrolysis (4,811 ± 208 kt capacity), and €3.2 ± 0.3 billon for advanced pyrolysis reactor (3,582 ± 288 kt capacity). In $S5_{optimistic}$, it is estimated that total €8.2 ± 0.5 billion capital investment is needed. In $S5_{optimistic}$, €3.5 ± 0.2 billion would be needed to build the pretreatment facility for FP prior to advanced pyrolysis, while €4.2 ± 0.4 billion would be needed to build the advanced pyrolysis reactor. In S5optimistic, around €0.4 ± 0.1 billion is needed for advanced MR infrastructure (Figure 6.4). Compared to S2*pessimistic*, it is estimated that 27–35% more capital investment would be needed in S4*optimistic* and S5*optimistic* (Figure 6.4).



Input capacity (in kt)	$S1_{pessimistic}$	$S2_{pessimistic}$	$S3_{optimistic}$	$S4_{optimistic}$	$S5_{optimistic}$
Mechanical recycling	5,663 ± 259	1,435 ± 76	6,551 ± 260	1,739 ± 78	404 ± 31
Pre-treatment pyrolysis	-	4,228 ± 180	-	4,811 ± 208	6,146 ± 325
Pyrolysis reactor	-	3,094 ± 235	-	3,582 ± 288	4,576 ± 374

Color Legend
Conventional mechanical recycling; Conventional pyrolysis; Pretreatment for pyrolysis
Advanced mechanical recycling; Advanced pyrolysis

FIGURE 6.4 Estimated capital investment of conventional/advanced mechanical recycling and pyrolysis in S1*pessimistic*, S2*pessimistic*, S3*optimistic*, S4*optimistic*, and S5*optimistic*, as elaborated in section 6.2.2.2 and Table 6.1. The capital investment for pyrolysis includes product upgrading (e.g., hydrotreatment), excluding steam cracking and polymerization. Values are rounded, including the standard deviation (error bars).

6.3.4 Circularity of flexible packaging in Europe in the future

This study focuses on the material flows of FP waste treatment throughout waste management systems in 2030. A few crucial assumptions made in the model are: better FP designs (e.g., more mono-material FP), increased sorting capabilities using digital watermarks, chemical tracers or artificial intelligence (e.g., PP film sorted as a separate fraction; Lase et al., 2022), large-scale implementation of 'improved' MR and pyrolysis (as illustrated by Lase et al., 2022; 2023a; 2023b), and high-quality recycled plastic from MR and pyrolysis to be used for FP applications. However, the quality aspects such as the feedstock quality (i.e., sorted PE, PP, or PO bales; Lase et al., 2022) and the quality of recycled plastic (e.g., technical properties; Bashirgonbadi et al., 2022) need to be considered in future research. Coupling quantity-based (as shown in this study) and quality-based modeling would improve the assessment of potential end-market applications, including the technical feasibility of processing different bales' quality and the use of different recycled plastic quality (to meet specific technical properties required by the end market). Furthermore, quality evaluation following legal aspects of using recycled plastic in contact sensitive applications (e.g., food contact material) should be included, as illustrated by Demets et al. (2020) and De Tandt et al. (2021). From the end market side, it is difficult to predict the market uptake of recycled plastic, given the breadth of technical requirements of various applications, which is subjected to further research (Tonini et al., 2022; Demets et al., 2020). Thus, the MFA results in this study should be seen as the preliminary assessment in which maximum uptake (of recycled plastic) to meet recycled content targets for FP under optimal conditions, whereas the actual uptake might be lower.

Next to technical evaluation of the waste feedstock availability, recycling technologies, and recycled plastic quality from FP waste treatment, the MFA results show that there is a trade off between producing higher (technical) quality recycled plastic via pyrolysis and higher quantity of recycled plastic via MR. Figure 6.3 shows that a considerable quantity of recycled plastic that can be used for non-flexible packaging (e.g., 3.582 ± 238 kt in $S3_{optimistic}$) would be reduced (e.g., 914 ± 87 kt in $S4_{optimistic}$) as we try to meet the proposed recycled content target (assuming pyrolysis will become more dominant to help achieve the target). The proposed recycled content targets in EU 27+3 could mean that less recycled plastic would be produced, which can lead to a lower EoL-RR (as demonstrated in $S2_{pessimistic}$ vs $S1_{pessimistic}$, Table 6.8), but with more higher recycled plastic (technical) quality to reach recycled content target (e.g., in $S2_{pessimistic}$ vs. $S1_{pessimistic}$). This is likely to be a political discussion if ambition is to achieve closed-loop of plastic, or to substitute as much crude oil as possible. An important aspect here is also the production of base chemicals and fuels, which substitute oil, but do not count for the plastic substitution. Further market research on the demand for base chemicals from petrochemical industry should be conducted to investigate the potential substitution from plastic waste recycling. It is also important to note that pyrolysis and advance MR technologies are still under development, making the learning curve of these technologies not so easy to predict. This is especially relevant given the complexity of waste composition (as feedstock) and the interplay other sustainability strategies such as DfR, sorting techniques, pretreatment strategies, etc.

6.4 CONCLUSION

This study investigates the feasibility of reaching the proposed recycled content targets for contact sensitive and non-contact sensitive flexible packaging applications set out by the European Commission (2022) in Europe by 2030. Material flow analysis is used to investigate the potential combination of mechanical recycling and pyrolysis technologies to produce recycled plastic to be used as recycled content for flexible packaging applications.

The results suggest that the proposed targets can be reached in 2030 when mechanical recycling and pyrolysis are used (as complementary technology) to deal with FP waste, assuming that pyrolysis will become a more dominant technology to meet the recycled content for contact-sensitive FP application. Depending on the selection of recycling technology, a considerable amount of FP-based recycled plastic currently going to non-flexible packaging applications as recyclates (e.g., wooden park bench, agriculture films, etc.) could be reduced, which implies that more virgin plastic materials (incl. non-plastic materials like wood), or other sources of material (e.g., recyclates from rigid packaging), should be used for these end market applications. Moreover, implementing mechanical recycling and pyrolysis also reduces the overall end-of-life recycling rate of flexible packaging, which illustrates the trade-off between achieving higher-quality recycled plastic (through pyrolysis) and the annual recycled plastic production. From economic perspective, it can be observed that €7.7 – 8.2 billion of capital investment is needed to build mechanical recycling and pyrolysis (incl. pretreatment and hydrotreatment) infrastructure to reach recycled content targets.

For future research, more robust modeling parameters should be gathered to improve the model results and analysis, for example monitoring the development of state-of-the-art sorting and recycling techniques to increase sorting and recycling yield (e.g., ‘smart’ sorting using digital watermarks, catalytic pyrolysis, etc.). The MFA results indicate that improvement in the sorting and recycling techniques can considerably improve the quantity of recycled plastic production to meet the targets. Moreover, the recycling system can still be optimized by balancing the waste feedstock processed through mechanical recycling or pyrolysis to find the most economically (and environmentally) beneficial system. This optimization study can provide information on the full circularity potential of flexible packaging in Europe to reach recycled content targets and achieve the highest recycling rate possible.

CHAPTER 7: CONCLUSIONS AND FUTURE PERSPECTIVES

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Chapter 7

Conclusions and future perspectives

7.1 GENERAL CONCLUSIONS

The plastic recycling rate in Europe is relatively low due to various factors such as complex material compositions (e.g., multimaterial structure, presence of additives, etc.), limitation of state-of-the-art sorting or recycling technologies, and unfavorable socio-economic circumstances (e.g., poor source separation by individuals or low economic value of recycled plastic). On the other hand, the European Commission sets out very ambitious recycling targets (e.g., recycling rate or recycled content targets), alongside voluntary pledges by the European plastic industry to improve plastic circularity in Europe (e.g., to boost recycled plastic production to 10 million tonne by 2025). Thus, it is evident that the European government and industry are looking for effective strategies to improve the status quo of plastic waste treatment soon. Hence, this study investigates various improvements within the European plastic waste management systems by developing and applying a prospective material flow analysis model. The model allows us to trace the fate of plastic waste throughout defined waste management pathways (or future scenarios) and asses the associated recycling performances using several evaluation indicators.

The overarching conclusions of this study are centered around the following four key elements:

- Combining material flow analysis (MFA) with other assessment methods such as MIOA and CBA, and applying these methods to analyze potential future scenarios to improve beyond the current *status quo* of plastic waste treatment in Europe. This allows assessing circular economy and sustainability strategies especially around new emerging recycling technologies (as presented in Chapters 2, 3, and 6).

- MFA as a decision-making tool to support the development of new technologies and providing quantitative evidence from technical, economic, and environmental perspectives (as presented Chapters 4 and 5).
- MFA as a monitoring tool to ensure the attainment of recycling targets, such as end-of-life recycling rates and recycled content targets (as presented in Chapters 2 and 6).
- Future outlook of MFA to be used in the center of circularity strategies in combination with life cycle assessment, extended producers responsibility schemes, design-for-recycling, etc. (as presented in future outlook in Chapter 7).

Focusing on each chapter in this thesis, in the first instance, different plastic recycling options can be used to deal with certain waste streams (as feedstocks to plastic recycling plants) in the future (in 2030 as the evaluation year in this study) in Europe, as discussed in **Chapter 2**. Mechanical recycling is, to date, the most ubiquitous option used to recycle plastic waste in Europe. However, emerging technologies such as chemical recycling (e.g., pyrolysis coupled with hydrotreatment and steam cracking) and solvent-based recycling (e.g., dissolution–precipitation) plants are expected to be built in Europe, and become alternative options for plastic recycling. Nevertheless, there is limited study on the potential contribution of chemical and solvent-based recycling to plastic circularity in Europe. The material flow analysis results show that plastic waste treatment through chemical recycling (e.g., pyrolysis and gasification) produces not only recycled plastic, but also base chemicals (as feedstock for the petrochemical industry) and hydrocarbon (as new energy sources). The multi-outputs of chemical recycling certainly contribute positively to the overall plastic recycling system and plastic circularity, yet pose legal challenges because fuel and energy alike outputs are not considered as 'recycling' under the Waste Framework Directive in Europe. Hence, the results of this study provide quantitative evidence of the contribution of chemical and solvent-based recycling by providing granular material flow analyses, which can be used to measure the attainment of recycling targets in compliance with European legislation. For policy makers, the approach can also be used to monitor the annual advancement of plastic circularity as well as the attainment of recycling targets in Europe.

Next to providing granular evidence of various outputs from plastic recycling technologies, this study provides quantitative evidence of how much the contribution of chemical recycling and solvent-based recycling might be. The material flow analysis results suggest that the highest achievable end-of-life plastic recycling rate through robust plastic recycling treatments is 80%. In these most positive scenarios, chemical recycling is seen as complementary to mechanical recycling (and not competing for feedstock) to deal with plastic waste streams that would otherwise be landfilled or incinerated (due to the limitation of current state-of-the-art mechanical recycling technology). The 80% end-of-life plastic recycling rate is calculated based on the amount of plastic waste recycling into new recycled plastic (i.e., 61% plastic-to-plastic recycling rate) and into new valuable base chemicals (i.e., 19% plastic-to-chemical recycling rate), excluding the plastic-to-fuel recycling of around 3%.

Furthermore, the estimation of recycled plastic production from mechanical, chemical, and solvent-based recycling allows the quantification of potential recycled (plastic) content availability. The material flow analysis results (at sector level) suggest that chemical recycling of plastic waste can help achieve recycled content targets (e.g., 25–30% recycled plastic in new electronic devices) set by stakeholders within the European plastic industry. Next to a large-scale implementation of chemical recycling, the recycled content targets can be met when closed-loop recycling and processing the ‘missing plastic’ (i.e., plastic waste that is not accounted for in the European statistical databases) are realized simultaneously.

Next to investigating the potential technological advancement of plastic recycling options at the European level, this study also explores more specific improvements needed in two important sectors in Europe, namely the electronic sector and the packaging sector. In **Chapter 3**, a multivariate input-output analysis model is developed to better estimate the e-waste generated in the Belgian and Dutch markets (as a case study). The multivariate input-output analysis model is selected because it considers the dynamic interplay between electronic products’ sales (placed on the market), stocks (as product accumulation in the market), and lifespan. This approach improves the estimation of e-waste generation based on the products’ lifespan distribution profile. After that, the estimated e-waste generation (and its disposal age compositional data) is combined with the material flow analysis model to evaluate the recycling performance in the electronic sector. Results of this modeling approach suggest that the lifespan of electronic product gets shorter nowadays in the Belgian and Dutch markets. This conclusion can be drawn by observing the lifespan distribution profiles of the selected small household appliance products (i.e., vacuum cleaners, coffee machines, and electric shavers). The current bottlenecks within e-waste recycling chain to achieve the recycling targets in Europe (e.g., recycling rate and recycled content targets) are low separate collection rate and pre-processing efficiency (i.e., manual dismantling and sorting) for plastic in e-waste. In the future (2030 is selected as the evaluation year in this study), it is estimated that a considerable amount of electronic products purchased in the past years (e.g., 1990 – 2010) can still be found (become e-waste) in 2030. These old electronic products are expected to take up to 5–10% of the total e-waste age composition in the evaluation year 2030. This phenomenon can be explained by the fact that the lifespan distribution of the selected electronic products can range up to 25–30 years, mainly because many of them are "hoarded" in the consumers' possession before disposal. Subsequently, some of the e-waste disposed of in 2030 would still contain high-risk legacy chemicals (e.g., brominated flame retardants) because the restriction of hazardous substances in electronic devices was only put into force effectively in 2006. Therefore, the multivariate input-output analysis model results suggest that a separate treatment for plastic containing hazardous substances from e-waste would still be relevant in 2030. Thus, the combination of multivariate input-output analysis and material flow analysis models gives a better estimation of plastic-containing waste products (e.g., e-waste, as studied in this research). It also provides insights on the potential resources and inherited legacy chemicals therein, relevant for recyclers and policymakers to reinforce their recycling strategies and measurements.

The second case study in this PhD examines waste management systems for household and non-household flexible packaging waste. Flexible packaging waste recycling rates relatively low in Europe because of complex material composition (multimaterial multilayer structure), low separate collection rates, and low recycled plastic quality. This leads to the low economic value of recycled plastic from flexible packaging. For example, recycled plastic from flexible packaging waste is typically used for less demanding applications such as garbage bags and horticulture products (e.g., garden pots), and not as new flexible packaging again. Three chapters are dedicated to flexible packaging recycling.

Chapter 4 investigates the recycling performance and economic balance of an improved mechanical recycling process for flexible packaging waste from households. The improved mechanical recycling process, called *Quality Recycling Process* (QRP), is compared with a typical conventional mechanical recycling process for flexible packaging waste from households by developing and applying material flow analysis and cost benefit analysis models. QRP is perceived as a more elaborated recycling process because it adds additional sorting to bales rich in PE film (e.g., DSD 310-1) and bales rich in mix PO film (e.g., DSD 323-2), followed by either Tier 1 or Tier 2 recycling of QRP. Tier 1 recycling of QRP is comparable to a typical conventional mechanical recycling process, while Tier 2 recycling of QRP adds hot washing, extrusion with a double melt filter, and deodorization steps to clean the incoming waste streams further. The material flow analysis results show that the process yield of QRP (64–66%) is comparable to conventional mechanical recycling (66%) for flexible packaging waste from households. However, QRP creates two additional outputs that are not created from conventional mechanical recycling, namely rPE Film Natural and rPP Film (an 'r' to the nomenclature referring to different regranulate types). These regranulates have considerably higher polymer grades (99% for rPE Film Natural and 96% for rPP Film) compared to regranulates from conventional mechanical recycling. The rPE Film Natural also has the highest transparency grade (i.e., 83%), which correlates to a higher regranulates quality and potentially higher market values. The other two outputs of QRP are (Tier 1- and Tier 2-) rPE Flex and rPO Ne, which have similar qualities to regranulates produced by conventional mechanical recycling, allowing the same end-market applications. The granular material flow analysis model also provides insights into the net recovery of different flexible packaging types processed via QRP and conventional mechanical recycling. The net recovery results indicate that monomaterial transparent flexible packaging (e.g., PE Film transparent) has better net recovery than multimaterial flexible packaging (incl. black and heavily printed films). This result suggests that following several "design for recycling" guidelines can improve the recyclability of household flexible packaging waste.

The cost benefit analysis results conclude that the capital investment of QRP can be up to 2 times higher than the conventional mechanical recycling, while the annual costs (OPEX and CAPEX) of QRP operation can be 1.7 times higher. However, the higher capital investment and annual costs of QRP can be compensated by delivering higher quality regranulates (with higher market values) that can be used in more demanding applications such as shrink film, sealable pouches, and standing pouches. Compared to conventional mechanical recycling, the QRP can improve the economic value of flexible packaging waste recycling from household by 5–38%. The higher capital investment of QRP is attributed to the QRP additional sorting (6 – 7% of total capital investment), hot washing (10 – 14% of total capital investment), and improved extrusion (5 – 8% of total capital investment). With extra equipment needed for QRP, the capital investment on the extra spaces or areas in the QRP recycling plant takes up to 22 – 23% of the total capital investment. Similarly, the annual costs associated with QRP additional sorting, hot washing, and improved extrusion process account for 3 – 4 %, 9 – 12 %, and 4 – 8% of the total OPEX in QRP, respectively. Concluding, QRP has the potential to improve the current recycling of household flexible packaging waste by producing regranulates with better quality, which can fulfill a larger market segment (such as flexible packaging again). Moreover, results show that such improved mechanical recycling process for flexible packaging waste from households can be economically viable. Furthermore, external financial support is still needed to sustain QRP (e.g., through gate fees or green fees from extended producer responsibility systems), as is also needed in conventional recycling.

In **Chapter 5**, the technical and economic viability of collecting and mechanical recycling of flexible packaging waste from non-households are investigated by performing logistic simulations (i.e., waste collection of non-household plastic waste) and developing material flow analysis combined with cost benefit analysis model (i.e., mechanical recycling for non-household plastic waste). The compositional analyses from actual waste sampling in the urban areas of Ghent in Belgium suggest that non-household plastic film waste consisting of mainly PE transparent (50% by weight), PE colored (36% by weight), PP transparent (3% by weight), and PP colored (3% by weight). The material flow analysis results suggest that the mechanical recycling yield of non-household plastic film waste ranges from 61 – 77% with regranulates (i.e., rPE) consisting of 89 – 95 % PE. The sensitivity analyses show that a higher level or residue content (up to 50% by weight) can drop the mechanical recycling yield to 48 – 61%.

It is estimated that 10,400 tonne per year of non-household plastic film waste will be generated (and can be collected) from the urban areas of Ghent and its 12 neighboring municipalities considered in this study. The estimated capital investment needed to build the recycling plants is €4 – €7 million, depending on the plant layouts. Given the economic modeling parameters adjusted to the Belgian market, the annual costs are expected to range from €4 – €6.5 million per year. The model results also indicate that a positive net economic balance of €5 – €537 per tonne regranulate output (i.e., the recycling chains generate profit) is achieved when around 10,500 tonne per year of non-household plastic film waste from urban areas. Moreover, a positive economic balance is achieved when waste is collected fortnightly or monthly, and the feedstock is maintained at higher quality (i.e., collected non-household plastic film waste with 5% residue content). In this positive scenario, the collection cost is estimated to be around €90 – €340 per tonne rPE with mechanical recycling cost of

€545 – €725 per tonne regranulate (via basic recycling plant) or €845 – €1,100 per tonne regranulate (via advanced recycling plant). Furthermore, the regranulates prices are sold at higher price to establish a self-sustaining non-household plastic film waste collection and mechanical recycling, i.e., €800 – €1,000 per tonne regranulate (via basic recycling plant) and €1,200 – €1,500 per tonne regranulate (via advanced recycling plant). Overall, the net economic benefit of collecting and mechanical recycling of non-household plastic film waste ranges from €5 – €537 per tonne output, depending on the collection frequencies and mechanical recycling plant layouts. The sensitivity analysis results suggest that the economic balance of collecting and mechanical recycling drops when the residue content reaches 25%, but can still generate profit if the regranulate is sold at highest price of €1,000 – €1,500 per tonne regranulate. However, collecting and recycling non-household plastic film waste becomes economically unfeasible when the residue content reaches 30 – 35%, even when the regranulate is sold at highest price (€1,000 – €1,500 per tonne regranulate).

Finally, the greenhouse gas emission accounting suggests that the production of rPE from non-household plastic film waste is significantly lower (49–79%) than the current linear economy of virgin PE granulate production and incineration (assumed baseline for current plastic waste treatment). The environmental performance of collecting and mechanical recycling non-household plastic film can still be improved by minimizing residual streams and maintaining higher-quality feedstock from waste collection. Thus, establishing a sustainable waste management system for non-household end-use plastic film waste (at least on par with the household counterparts) can be an important step to increase plastic recycling rates, and towards a more circular economy of non-household plastic waste.

Chapter 6 investigates the potential recycled content availability for flexible packaging from household and non-household sector, specifically to meet the recycled content targets set out in the new proposed Packaging and Packaging Waste Regulations (PPWR) (European Commission, 2022a). Article 7 of PPWR states that minimum 35% recycled content for non-contact-sensitive and 10% recycled content for contact-sensitive applications should be achieved. Starting from the closed-loop assumption, in which the regranulates produced from recycling flexible packaging waste (from household and non-household), a material flow analysis model is developed to trace the fate of flexible packaging waste throughout the end-of-life system. Particularly for this preliminary assessment, two recycling technologies are considered, namely (conventional/advanced) mechanical recycling and pyrolysis. Five scenarios are developed, which depicts further improvements in 2030 related to the flexible packaging design (e.g., from multi- to mono-material), more selective collection (e.g., flexible packaging collecting in P+MD system), better sorting techniques to sort mono- and multi-material flexible packaging (e.g., 'smart' sorting based on digital watermarks or artificial intelligence), and better mechanical recycling or pyrolysis technologies (e.g., advanced mechanical recycling like QRP or catalytic pyrolysis). Based on the developed material flow analysis model, the capital investment associated with achieving the recycled content targets are estimated, which is based on the economic factor (in €/tonne input to mechanical recycling or pyrolysis) found in literature.

The material flow analysis results in Chapter 6 indicate that 35% recycled content target for non-contact-sensitive flexible packaging and 10% recycled content for contact-sensitive flexible packaging can be achieved when mechanical recycling and pyrolysis are used simultaneously to deal with flexible packaging waste (assuming 100% closed-loop recycling, from flexible packaging waste to new flexible packaging in Europe). In the most positive scenarios, advanced mechanical recycling or advanced pyrolysis technologies can be used to meet recycled content targets for contact-sensitive and non-contact-sensitive flexible packaging in 2030. In order to realize this full closed-loop recycling, it is estimated that €7.7 – €8.2 billion of capital investment would be needed to build the mechanical recycling and pyrolysis (incl. pretreatment and hydrotreatment) infrastructure. The material flow analysis results also indicate an important trade-off between achieving higher-quality of regranulates to meet 10% recycled content target for flexible packaging (assuming pyrolysis would become a more dominant technique to achieve the target), and annual regranulates production (i.e., quantity of secondary materials). As results, implementation of recycled content targets through mechanical recycling and pyrolysis (as complementary technique) could reduce the overall end-of-life recycling rate for flexible packaging, in which in this case the end-of-life recycling rate is defined as the ratio between the flexible packaging waste converted into secondary materials (i.e., regranulates and base chemicals, fuel exempted) of the total flexible packaging waste generated in 2030.

Concluding, recycling (and collecting) household and non-household plastic film waste (from urban areas) can be economically attractive when a few operating conditions are met. To realize this, waste producers, waste operators, and regulators must establish effective waste management systems in the future. Targets and extended producer responsibilities schemes should be established to incentivize non-household end-use plastic waste treatment, especially to sustain plastic recycling operations when regranulate price drop (e.g., due to low oil price). Financial incentives for waste generators to properly separate waste at source can be promoted to ensure feedstock quality and quantity. Nevertheless, given the large quantity of plastic films in household and non-household sectors (around 9.0 million tonnes in Europe), society and industry will need this feedstock to achieve its recycling targets.

7.2 SCIENTIFIC IMPACTS AND CONTRIBUTIONS

Evidently, plastic is widely used in the economy and positively impacts humanity and society. However, plastic waste pollution causes adverse environmental impact and economic losses, hence, enabling a circular economy for plastic is crucial. Plastic recycling can be used as one of the solutions to deal with plastic waste pollution, in which plastic waste is collected for extensive sorting and recycling to allow the production of secondary materials. This thesis focuses on assessing potential improvements within the European plastic waste management systems. For this purpose, different methods are used, including material flow analysis (MFA) models in combination with other assessment tools such as cost-benefit analysis (CBA) and carbon footprint calculations. The results in this thesis could contribute to assessing potential improvements within the plastic waste recycling systems by providing thorough assessments on the fate of plastic waste throughout the recycling chains as well as the associated economic and environmental aspects, focusing on the European recycling chains (as case studies). Moreover, this thesis also provides insights into the attainment of recycling rates and recycled content targets in Europe.

In first instance, the presented results in this thesis could contribute to improve active participation from a broader group of stakeholders to enable a circular economy for plastic and material circularity. Our results for example show that consumers play a crucial role in determining the fate of plastic waste, especially in applying plastic separation at source (e.g., at households, schools, offices, etc.). Some plastic waste (e.g., plastic in electronic waste) can ‘hibernate’ for a relatively longer period of time in consumers’ possession before being disposed of, which could delay material recycling and circulation from the waste products. Moreover, inefficient waste separation at source by the consumers could lead to plastic waste ending up in incineration facilities or landfills, in which plastic waste is not necessarily recycled back into the economy. Inefficient waste separation at source also impacts the performance of plastic recycling operations. For example, more residue is generated, which hampers the economic competitiveness of recycling plants as shown in Chapter 5. Indeed, the economic balance is affected because recyclers need to pay more for residual treatment and get less revenue from a lower recycled plastic quality. The environmental performance of plastic recycling is also affected, particularly from dealing with residual treatment, typically via incineration or landfill. Furthermore, when the waste contains more residue, a lower recycled plastic quality could be expected, limiting the potential use of the secondary materials (e.g., only suitable for less demanding applications such as trash bags). Hence, societal and behavioral changes could be detrimental to ensuring a successful and sustainable plastic waste management systems. This PhD thesis offers quantitative evidence for the role of stakeholders such as the consumers.

Furthermore, the results presented in this study emphasize (and could contribute) the urgent need to improve plastic waste compositional datasets. We need to gain more information on the waste feedstock compositions because it determines the extent to which recycling technology can deal with the waste and predict recycled plastic quality, which is important in plastic waste management. The collected plastic waste (from households, schools, offices, etc.) is typically contaminated with other non-plastic materials (e.g., paper, organic waste, etc.), partly because of inefficient separation at source by the consumers. Even after an elaborated sorting process at material recovery facilities, a considerable amount of non-plastic material can still be found, making recycling more challenging. With gained knowledge around waste compositions, the industry could choose the most appropriate sorting and recycling techniques to improve the waste streams, hence, plastic waste can be valorised into high-quality recycled materials (both from technical and economic perspectives). The prediction of plastic waste flows throughout plastic recycling chain and the quality of secondary materials could also be improved by gathering more realistic waste compositional data.

This thesis contributes to discussing plastic waste management from policy perspective. The presented case studies and results in this thesis could be used as the basis to formulate recycling targets based on quantitative projections. Policy makers could adopt the modeling approach or results to assess the impact of ambitious recycled content targets on the overall plastic recycling rate, and vice versa. With current market conditions, mechanical recycling is still used as the main plastic recycling option, however, new emerging recycling technologies like chemical and solvent-based recycling are expected to enter the European market in the near future. Hence, policymakers would need to revisit the European recycling strategies to ensure that these technologies positively contribute to increasing plastic recycling rates and provide recycled content. In this context, paying more attention to the nature of each recycling technique becomes imperative, for example, in the case of multi-output chemical recycling technologies. Pyrolysis and gasification technologies produce recycled polymers and valuable base chemicals but also fuels, which does not conform with 'recycling' definition in Europe. Thus, policymakers need to come up with a harmonized set of rules on how to quantify recycling rates for all recycling technologies in the near future (especially for the multi-output recycling process). Furthermore, with current legislation, the use of mechanically recycled plastic for contact-sensitive applications (e.g., food packaging) is limited partly because of concerns related to legacy chemicals. On the other hand, it is currently assumed that chemical and solvent-based recycling could produce virgin-like recycled plastic, which is expected to be the dominant technology to supply contact-sensitive applications and reach the recycled content targets. In this context, policymakers should ensure that legislation does not discriminate or aid certain recycling option(s). Lastly, this topic is especially relevant because the model results suggest that the recycling rates and recycled content targets could change considerably depending on the preferred recycling technologies used to deal with plastic waste.

7.3 FUTURE PERSPECTIVES

Findings from this research have provided more knowledge on the potential improvement of plastic waste management systems in Europe to achieve the ambitious recycling targets set out by the European Commission and pledges by the European plastic industry. In particular, quantity-based material flow analyses modeling has been used at different levels such as regional level (e.g., flows of plastic waste in Europe 27+3: in Chapter 2), national level (e.g., flows of WEEE in Belgium and The Netherlands: in Chapter 3), urban area level (e.g., flows of flexible packaging in the City of Ghent and its neighboring municipalities: in Chapter 5), and recycled content availability in Europe 27+3 (for EEE in chapter 3 and flexible packaging in Chapter 6). Based on the findings of this study, a number of potential improvements have been noted as suggestions for further research, as elaborated in the following sections:

7.3.1 Expanding the scope of material flow analysis modelling

Notably, this study focuses on improving end-of-life plastic waste management options, such as recycling options to deal with plastic waste in Europe. In the future, the end-of-life-oriented solutions analysis should be coupled with production- and use-oriented solutions such as waste reduction, “design for recycling” implementation, and new circular business models like repair, remanufacture, re-use or re-use or refill strategies. Thus, further research on the more systematical investigation of production- and use-oriented solutions can extend this field of study. This research trajectory can comprehensively analyze the potential full plastic circularity in Europe.

Moreover, the attainment of recycled content targets in Europe has been investigated in this study by modeling aggregated flows of plastic waste from various sectors. A case study on recycled content availability from three small household appliances in the electronic sector has also concluded that there is plenty of room for improvement to achieve the recycled content targets through closed-loop recycling. However, it is difficult to predict the future market uptake of recycled plastic. The quantification of recycled content availability only shows the potential at the sector level, while some pledges are very specific to application or product level, such as recycled content in new passenger cars or electronic devices. In this case, the findings can be perceived as indications or a proxy towards the average recycled content availability in the respective sector. Thus, further research should be conducted to assess the feasibility of recycled content targets at the application or product level. This research at application or product level can also be combined with research at system level to investigate the required actions and impacts to achieve the recycled content targets from system level perspective (e.g., capital investment to build the infrastructure).

For further research, a more detailed material composition characterization should be carried out for better resource estimation, risk assessment, and recycling target attainment within the European plastic recycling chain. For example, multivariate input-output analysis can determine the e-waste age composition, including their potential resources (e.g., plastic) and hazardous substances (e.g., brominated flame retardants). Hence, a detailed material composition analysis is highly relevant to formulate adequate measurements, especially to comply with European health and safety regulations. Furthermore, some multimaterial waste behaves differently throughout the plastic recycling chains (e.g., during washing and extrusion), and a more detailed compositional characterization of the waste will improve the material flow modeling results and analyses. This work can provide a good foundation for further research on substances flow analysis (e.g., volatile organic compounds in packaging waste) in real plastic recycling systems to meet the end market requirements (e.g., maximum odour content in recycled plastic).

Finally, this study is limited to the selected case studies and scenarios, which can be further extended in the future (based on new technological development and insights). The developed model has the capacity to be expanded to other regions (e.g., Asia Pacific region), sectors (e.g., automotive sector) and product categories (e.g., other e-waste types, packaging formats, etc.). Furthermore, it can be applied more specifically on certain polymer types or other materials/substances in certain sectors or regions. Expanding the research further is highly relevant because the model's modularity allows the parameters to be easily adjusted.

7.3.2 Linking the quantity-based modeling with the quality aspects

This study focuses on the quantity-based material flow modeling of different plastic waste streams through different recycling options, such as mechanical, chemical, or solvent-based recycling. However, further studies on the bale and regranulate quality should be carried out to meet certain technical specifications by recyclers. The evaluation of waste composition is important for mechanical recycling and pyrolysis including the level of contamination or impurities, such as chlorine and polymeric contaminations. A more detailed characterization of the waste feedstock quality will also help predict the quality of the recycled plastic. Thus, the robustness of the model should be improved by analyzing the quality aspects in recycling by predicting the output quality from the model input (as waste feedstock).

Evaluation of recycled plastic's quality also includes legal aspects to use recycled plastics in certain application (e.g., as food contact material) and technical characteristics of recycled plastic to meet market specification (e.g., processability of recycled plastics). This is important to predict the potential market uptake (i.e., to avoid market saturation that will reduce the uptake of recycled plastic). Furthermore, there is a possibility that a considerable amount of mechanically-recycled plastic has to be exported outside Europe because the European market that can deal with certain mechanical recycling qualities gets saturated. According to the current perception, potential market applications that can use recycled plastic might be less of a concern for chemically-recycled plastic because of a higher recycled plastic quality. Thus, further research linking quantity-based modeling with quality aspects of recycled plastic needs to be further investigated.

7.3.3 Integrating material flow analysis with other sustainability assessment tools

The MFA model can be used to analyze one aspect of the sustainability performance of different alternative scenarios toward meeting the circular economy targets. Future research should provide information on the sustainability aspects (e.g., social, economic, and environmental impacts) concerning mechanical, chemical and solvent-based recycling.

To date, several studies have indicated that the environmental performance of chemical recycling is better than landfill and incineration, but not outcompeting mechanical recycling. However, depending on the substitution rate of virgin plastic (i.e., the quality aspects), chemical recycling can perform better than mechanical recycling (i.e., beyond 1:1 substitution rate of virgin plastic). There is also a need to further investigate the environmental benefit of producing base chemicals (e.g., aromatic fractions) from chemical recycling.

Furthermore, the model can also be coupled with estimations on capital investment, operational cost and revenue of chemical recycling and solvent-based recycling options. More importantly, future research should assess the economic feasibility of achieving higher recycled plastic quality through full chemical or solvent-based recycling chains, compared to mechanical recycling. This includes not only the pyrolysis of plastic to produce naphtha, but also naphtha upgrading (e.g., through hydrotreatment) and steam cracking steps. For solvent-based recycling, the economic assessment includes the potential recovery of the chemical agents used in the pilot or commercial scale operations. Finally, a lot of research opportunities are available in this area to realize a full circularity of plastic in Europe and beyond.

7.3.4 Plastic in a circular economy and the role of material flow analysis in the future

Two leading solutions have been investigated to tackle plastic pollution and improve the circularity of plastic, namely *production-* and *use-oriented* solutions and *EoL treatment-oriented* solutions (Lase et al., 2023) (Figure 7.1). The production/use-oriented solutions correspond to a more sustainable plastic production (e.g., by using secondary resource), eliminating plastic use, substitute plastic with other materials (e.g., by using biodegradable plastic), improve product design (e.g., increasing design-for-recycling efforts), and implementing reuse/repair systems. The EoL treatment-oriented solutions correspond to improving waste management infrastructure, such as expanding separate waste collection, advancing sorting and pretreatment technologies, and increasing recycling capacity (mechanical, chemical, and solvent-based).

Figure 7.1 illustrates the future of plastic circularity, sustainable targets, and options, including material flow analysis in the center of a circular economy for plastic. In this sense, MFA can be used to trace the fate of plastic throughout the life cycle of plastic and bring transparency to ensure for example trustworthy product sustainability claims (e.g., product recyclability or recycled content) (Rizos et al., 2023; Tabrizi et al., 2021). When fully optimized, MFA can also be used to monitor the attainment of recycling targets (e.g., recycling rates or recycled content) as well as can enable eco-modulation for EPR fees and harmonize claim of products' sustainability performance (Figure 7.1, white boxes) (Laubinger et al., 2021; Broeren et al., 2022; Tabrizi et al., 2021). MFA can also be coupled with quality aspects and other sustainability tools (e.g., LCA) to quantitatively measure the 'quality of recycling', as shown by Roosen et al. (2023b). In this research, a quality of recycling framework is developed based on three fundamental pillars: achieving the lowest environmental impact, highest displacement potential of recycled plastic (from virgin counterparts), and longest in-use lifetime, which is defined as the durability of material to stay in the economy (Roosen et al., 2023b).

Choices regarding production/use-oriented solutions start with sustainable material sourcing, following design-from-recycling principles. Recycled content targets can also be used to boost recycled plastic production and ensure sustainable business for recycling operations and market for secondary materials. *Elimination* strategy, in which unnecessary production and use of plastic are significantly reduced, can be used to prevent plastic waste production at the end-of-life. A few practical implementations of this strategy is the elimination of unnecessary packaging (e.g., secondary packaging), excess unused space for marketing purposes, elimination of primary packaging for some products (e.g., 'naked' cosmetic products) (SYSTEMIQ, 2022; Fortunati et al., 2020). Next is *material substitution* strategies, which focuses on replacing hard-to-recycle plastic with other materials (e.g., paper or compostable packaging) (SYSTEMIQ, 2022). For this purposes, thorough assessments are needed such as to investigate the material properties (e.g., barrier properties), suitability for the EoL treatment (e.g., with the existing recycling options), and environmental impacts (typically through LCA studies). As an example, Maaskant et al. (2023) study highlights key aspects to be assessed when substituting thermoplastic materials with biobased alternatives of 17 products including food packaging, beverage bottles, textiles, mattresses, etc. Material substitution can also be seen as replacing material used in a product (e.g., adhesive for labels) so that it is suitable for the end-of-life treatment (e.g., water-soluble/releasable adhesive), which is also linked to the next strategy of *Design for Recycling*. Particularly for DfR, specific industrial guidelines can be followed, such as RecyClass or CEFLEX packaging guidelines (RecyClass, 2023; CEFLEX, 2020). DfR also exists for EEEs, in which the guidelines suggest that new electronic products should be made free of hazardous substances (following the WEEE Directive), easy to dismantle from hazardous components (e.g., battery or copper wire; Lase et al., 2021), and use connections that allow easy liberation/dismantling (PolyCE, 2021). Similarly, in the automotive industry, DfR can be implemented so that ELV parts can be easily dismantled, suitable for subsequent recycling infrastructure, and avoid the use of hazardous substances (Maury et al., 2022; SYSTEMIQ, 2022). At the use phase, the implementation of *reuse and repair systems* should be widely implemented, such as in-store refill stations, returnable/reusable food service items (e.g., cups or containers), and repair and maintenance services for electronic and automotive products (Feber et al., 2020). A study from Ellen MacArthur Foundation (2019) suggests four business-to-consumer reuse models, namely (i) refill at home, when consumers refill their reusable container at home, (ii) refill to go, when consumers refill their reusable container away from home such as at in-store refill station, (iii) return from home, when packaging is picked up at home by pick-up service, and (iv) return on the go, when consumers return packaging at the store or drop-off point to get product replacement.

Once discarded by the users, EoL treatment-oriented solutions should be applied to ensure adequate waste management treatments for plastic-containing products. To date, source segregation of plastic waste has been reported to increase plastic recycling rates and recycled plastic quality (Plastics Europe, 2022; Cimpan et al., 2015). Amongst other studies, MFA modeling by Roosen et al. (2021) shows that *expanding the collection* system for household plastic packaging can help accomplish the ambitious recycling targets. In the automotive and electronic sectors, Maury et al. (2022) and Wang (2014) indicate that separate collection of ELVs and WEEE via authorized waste operators (e.g., WEEE take-back scheme) is crucial to ensure the recovery and recycling of plastic waste. The next stage of plastic waste management is *sorting and pretreatment*, which is crucial to determine the extent to which we can produce high-quality materials (waste feedstock) for further recycling (Van Eygen et al., 2016; Parajuly et al., 2016). State-of-the-art mechanical (and automated) sorting processes (e.g., using NIRs) can be complemented by new emerging technologies such as 'smart sorting' based on artificial intelligence (NTCP, 2023), digital watermarks (Alliance to End Plastic Waste, 2023), and chemical tracers (Soares et al., 2022) so that plastic material can be sorted as much as possible for recycling. Another example from the automotive sector is scaling up advanced post shredder technologies (PST) more widely to further recover plastic from shredder residue (Maury et al., 2022; SYSTEMIQ, 2022). Once correctly sorted, the mechanical and chemical pretreatment steps can be employed to further remove impurities from the sorted waste streams (Kol et al., 2021). Next to typical pretreatment technologies like cold washing, float-sink, and drying (Civancik-Uslu et al., 2021; Larrain et al., 2021), advanced pretreatment technologies can be used, including hot washing with detergent (i.e., washing at elevated temperature; Lase et al., 2022), deodorization (Roosen et al., 2022), and dehalogenation processes (Kusenberg et al., 2022e). Finally, once recovered and pretreated, plastic waste can be recycled using mechanical, chemical, or solvent-based recycling techniques. Several studies have shown that we need all recycling technologies to enable a circular economy for plastic (SYSTEMIQ, 2022; SYSTEMIQ, 2023; Simon and Martin, 2019; Hann and Connock, 2020; Crippa et al., 2019), although further research needs to be carried out to identify the required capacity to ensure the most efficient and optimal mix of recycling infrastructure in the future.

MFA can be used at the EoL treatment stage as a monitoring tool for the attainment of recycling goals such as recycling targets (i.e., recycling rates and recycled content) (Brunner and Rechberger, 2005; Caro et al., 2023), enable eco-modulation for EPR fees (Laubinger et al., 2021), and claim products' sustainability claims (e.g., products' recyclability) (Tabrizi et al., 2021). Caro et al. (2023) shows that mass balance accounting allows transparent monitoring of different recycling technologies, including emerging multi-outputs recycling process (e.g., plastic-to-fuel from pyrolysis). Rizos et al. (2023) also show that different allocation methods can be used to claim attainment of recycled content targets, for example, polymer-only, fuel-exempt, or auto-fuel exemption methods. Modulated EPR fees (eco-modulation) can be fully implemented if granular and transparent EoL cost and material flows are known, following strict criteria on recyclability, recycling rates, and the presence of hazardous substances throughout the EoL system (Laubinger et al., 2021). Through MFA, modulated EPR fees can be implemented by putting lower fees or higher fees, depending on the product design and flow at the EoL management systems (e.g., flows of multimaterial flexible packaging at MRFs). The most significant benefits of such modulated system would theoretically strengthen the DfR implementation, increase recycling rate, and improve recycled plastic quality (Laubinger et al., 2021). Within the entire life cycle, MFA can be used as the foundation for further sustainability assessment such as LCA, circularity evaluation, cost-benefit analysis, techno-economic assessment, social-LCA, etc. Ultimately, through such a circular system (Figure 7.1), more secondary resources (mainly as recycled plastic, but also base chemicals and fuel from chemical recycling) could be produced in the future, which would decouple the reliance on primary resources (virgin plastic) and improve the sustainability performance of the entire plastic industry.

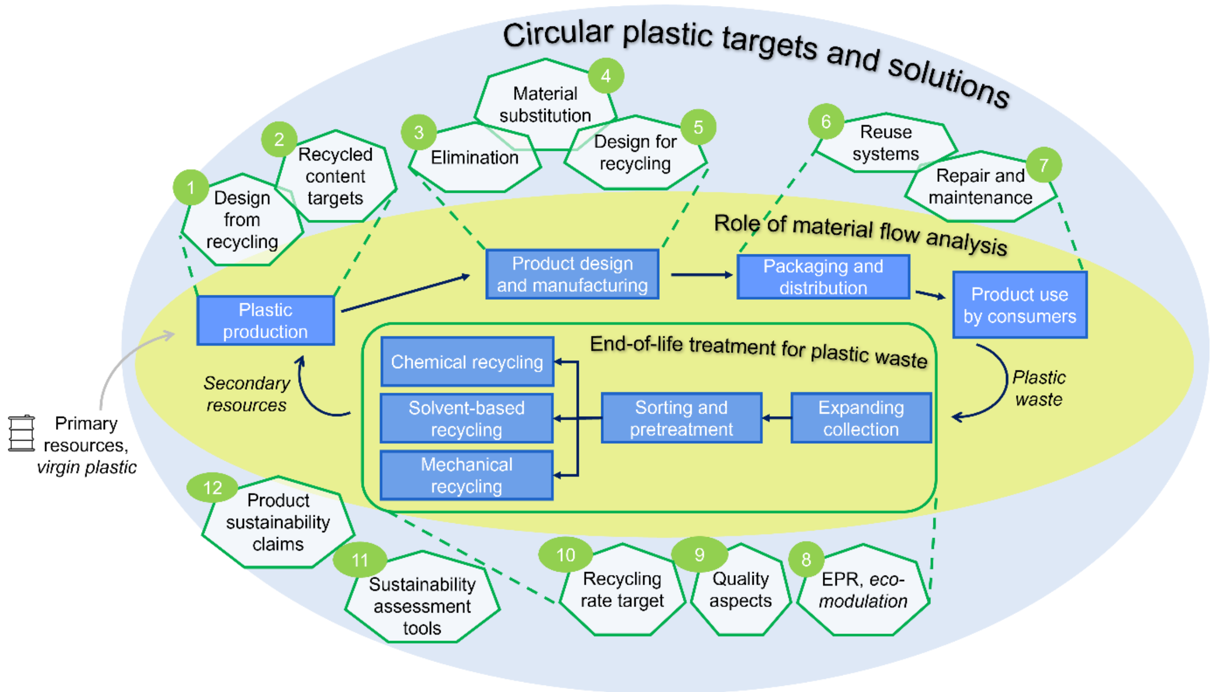

FIGURE 7. 1 Visualization of plastic circularity and role of material flow analysis in the center of a circular economy for plastic in the future. The blue boxes depict the life cycle of plastic. The white boxes correspond to options to improve circularity of plastic at the production, use, and the end-of-life stages.

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A

APPENDIX A. CURRENT AND FUTURE FLOWS OF PLASTIC WASTE RECYCLING IN EUROPE

SECTION 1: DESCRIPTION OF DIFFERENT CHEMICAL RECYCLING AND SOLVENT-BASED RECYCLING PROCESS

The following descriptions refer to the four different chemical recycling (CR) and solvent-based recycling (SBR) processes considered in this study. Each technology produces either polymer, base chemicals, fuels or a combination of these outputs. The residues of all CR technologies are sent for residual treatment (e.g., incineration, landfilling, or chemical treatment of hazardous waste).

Pyrolysis, coupled with *distillation, steam crackers* and *hydrotreatment* (Figure A.1) is a process applied to plastic waste to thermally break the polymer chains in an oxygen-free environment at temperatures between 350°C and 500°C, creating pyrolysis oil as the main product (Ragaert et al., 2017, Vollmer et al., 2020, Kusenberg et al., 2022a). Before being fed into the pyrolysis reactor, the plastic waste is (pre-)treated by means of shredding, washing and extrusion to remove materials that cause operational issues such as PET, PVC, metals, and organic compounds (Kusenberg et al., 2022a). Thereafter the pre-treated waste is fed into the cracking and condensation reactor to produce pyrolysis oil, gas, and char. Later, the pyrolysis oil is distilled to obtain the naphtha and wax. The naphtha is fed into commercial steam crackers producing base chemicals, including monomer for repolymerization, while the wax is used to replace slack wax (Kusenberg et al., 2022a; 2022b; 2022c; 2022d; 2022e; Civancik-Uslu et al., 2021; Larrain et al., 2020; Jeswani et al, 2021; Genuino et al., 2022).

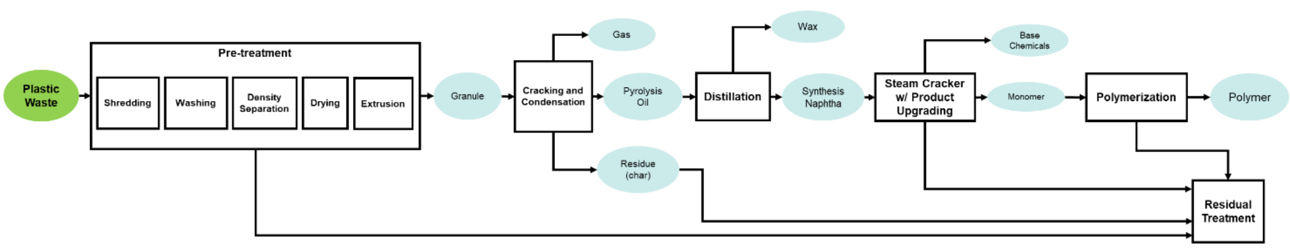

Figure A.1 Process flow diagram of pyrolysis. Adapted from literature (Arena and Ardolino, 2022; Civancik-Uslu et al., 2021; Larrain et al., 2022; Kusenberg et al., 2022a; 2022b; 2022c; 2022d; 2022e).

Gasification, coupled with *Fischer-Tropsch Synthesis* (Figure A.2) involves partial oxidation of plastic waste at temperatures between 700°C and 1,500°C to produce syngas, which is a mix of predominantly hydrogen and carbon monoxide (Hann and Connock, 2020; Crippa et al., 2018; Manžuch et al. 2021; Lopez et al., 2018; Cossu et al., 2017; Khoo et al., 2019). The syngas can then be used to produce valuable base chemicals, including monomer for repolymerization, via Fischer Tropsch Process (FTP) (or Methanol-to-Olefin (MTO) process) (SYSTEMIQ, 2012, Crippa et al., 2019, Gholami et al., 2021, Mastellone et al., 2019).

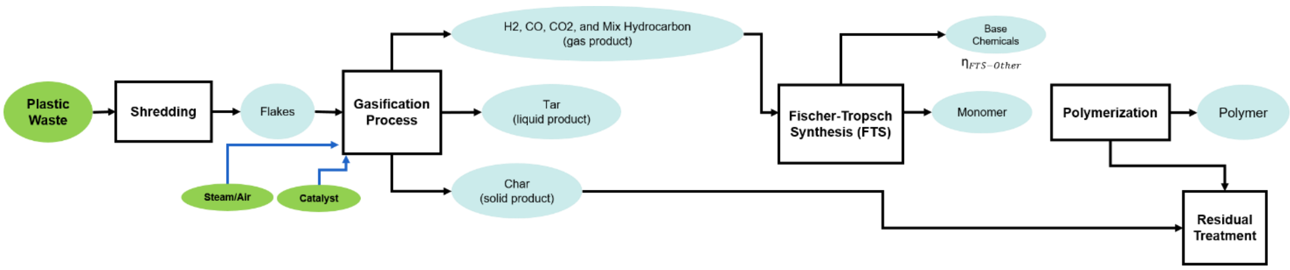

Figure A.2 Process flow diagram of gasification. *Source*: Hann and Connock (2020), Crippa et al. (2018), Manžuch et al. (2021), Lopez et al. (2018), Cossu et al. (2017), Khoo et al. (2019), Gholami et al. (2021), and Mastellone et al. (2019).

Chemical depolymerization (Figure A.3) is a process by which the polymer chain is broken down using chemical agents, which is also known as chemolysis and solvolysis. Depending on the chemical agents involved, this process is also called methanolysis, glycolysis, hydrolysis, aminolysis, etc. (Ragaert et al., 2017; Manžuch et al. 2021; Vollmer et al., 2020). Chemical agents break down the polymer into the shorter chain (either oligomers or monomers) from which it is originally formed. Thereafter, the oligomer or monomer is recovered from the mixture using distillation, precipitation and/or crystallization techniques (Hann and Connock, 2020; Crippa et al., 2019; Manžuch et al. 2021). The recovered monomers or oligomers are used for repolymerization (e.g., the Bis-HydroxyEthyl-Terephthalate (BHET) that is recovered by glycolysis is used for the production of PET) (Vollmer et al., 2020).

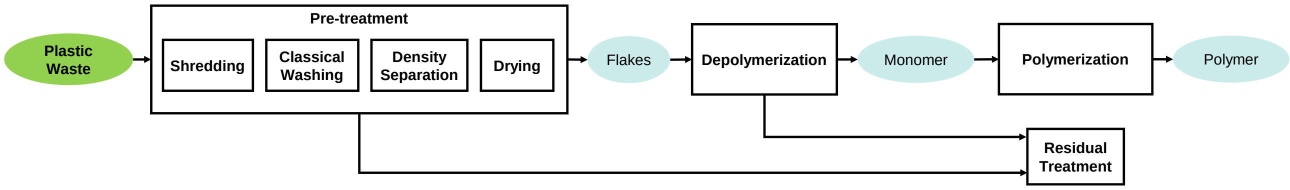

Figure A.3 Process flow diagram of chemical depolymerization. *Source*: Vollmer et al. (2020), Hann and Connock (2020), Simon and Martin (2019), and Crippa et al. (2019).

Solvent-based recycling or *purification* (Figure A.4) is used to dissolve the polymer in plastic waste by using a specific solvent(s) followed by the removal of the additives through filtration or phase extraction. Thereafter, the dissolved polymer is recovered by precipitation using an anti-solvent (Kol et al., 2021; Crippa et al., 2019). The product of solvent-based purification is a 'near-virgin' quality polymer, which can be reformulated into different applications (Crippa et al., 2019). Furthermore, deinking and delamination techniques are also considered as solvent-based purification, in which certain solvents are used to dissolve certain polymer layer (in multilayer packaging products) or colorants from the plastic waste (Kol et al., 2021; Ügdüler et al., 2020; Horodytska et al., 2018; Walker et al., 2020).

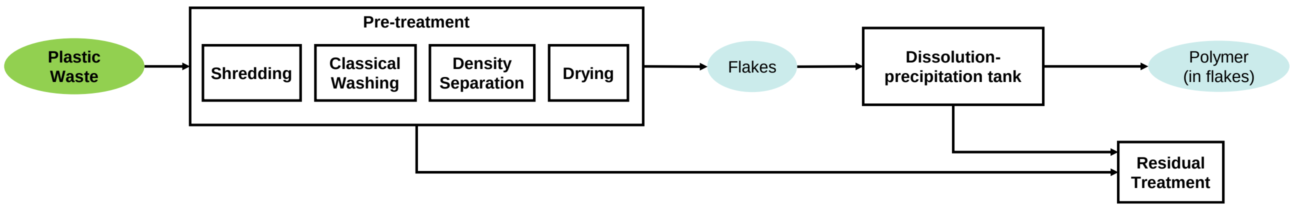

Figure A.4 Process flow diagram of solvent-based purification. *Source*: Kol et al. (2021), Ügdüler et al. (2020), Horodytska et al. (2018), Walker et al. (2020).

SECTION 2: PLASTIC WASTE MANAGEMENT INFRASTRUCTURE ACROSS DIFFERENT SECTOR IN 2018

Section A.2.1 Description of process flow diagram at process level per sector in 2018

Packaging sector: The management of plastic packaging wastes begins with either separate collection or mixed waste collection systems, as shown in Figure A.5 with the efficiency (in %) of $\eta_{collection}$ and $1 - \eta_{collection}$, respectively. In this study, the separate collection system refers to plastic waste collected separately (either as a single waste stream or commingled one) through kerbside, drop-off, or deposit systems. In Figure 2.1 in the main text, the separate collection rate is reflected by $TCs_{collection}$. These collection systems can differ from one country to another, or even within different municipalities; however, it is essentially a separate collection of plastic waste at source to be then sorted at sorting facilities, also known as material recovery facilities (MRFs) (Antonopoulos et al., 2021; Lase et al., 2022). At MRFs, a series of mechanical sorting processes are employed such as optical sorters, wind shifters, drum screens, to create different bales as shown in Figure A.5 by the different efficiencies (in %), i.e., $\eta_{primary\ bales}$, $\eta_{secondary\ bales}$, $\eta_{mixed\ bales}$ of the respective sorted plastic bales (Antonopoulos et al., 2021; Kleinhans et al., 2021). Thereafter, the sorted bales are sent to mechanical recycling (MR) facilities to be washed and reprocessed into recyclates ($\eta_{recycling}$ in Figure A.5) (Lase et al., 2022; Faraca et al., 2019). In Figure 2.1 in the main text, the yield of plastic packaging waste sorting is reflected by $TCs_{sorting}$, which refers to the sorting rate (Antonopoulos et al., 2021) or sorting recovery (Kleinhans et al., 2021) of different plastic packaging waste at MRFs. The $TCs_{recycling}$ in Figure 2.1 in the main text refers to the recycling rates (Antonopoulos et al., 2021) or recycling yield (Lase et al., 2022) of the respective sorted bales.

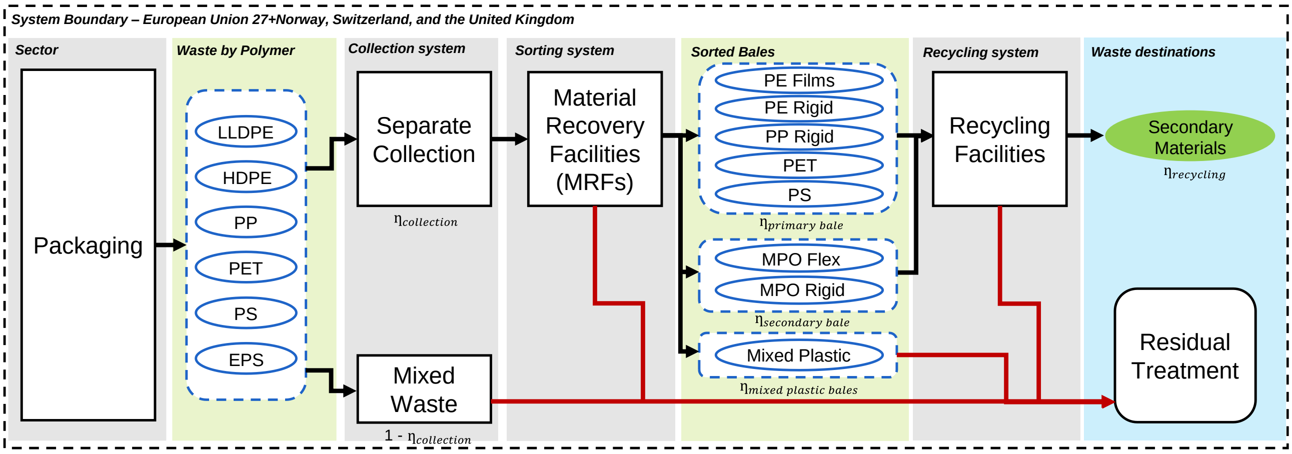

Figure A.5 Process diagram of plastic waste management in packaging sector. ɳ denotes the efficiency (in %) of each process.

Electronic sector: The waste generated from the electronic sector, known as waste electronic and electrical equipment (WEEE) (De Meester et al., 2019), can be classified into different groups such as small household appliances (SHA), large household appliances (LHA), cooling and freezing appliances (CFAs) (European Commission, 2018d). The management of WEEE starts with separate collection by the consumers, i.e., separate collection efficiencies (in %) $TCs_{collection}$ in Figure 2.1 in the main text and $\eta_{collection}$ in Figure A.6. The WEEE separate collection is facilitated by a formal management system through authorized WEEE collectors, repair centers or brokers/scraps dealers on behalf of the electronic producers or authorized WEEE collectors (Huisman et al., 2012; Wang et al., 2013). Otherwise, the WEEE are reported to be informally collected, which can be shipped to other countries as legal and illegal waste trading too (represented in Figure A.6 by $1 - \eta_{collection}$) (Huisman et al., 2012). Once separately collected, the WEEE undergoes manual depollution and disassembly, shredding, and mechanical sorting, which all together is called 'pre-processing step' (the efficiency is represented by $\eta_{sorting}$ in Figure A.6). After shredding, the mixed shredded plastic is mechanically sorted and sent to plastic recyclers. At the MR facilities, the sorted plastic is washed and extruded into recyclates (the efficiency is represented by $\eta_{recycling}$ in Figure A.6) (De Meester et al., 2019; Van Eygen et al., 2016). In Figure 2.1 in the main text, the $TCs_{sorting}$ for WEEE refer to the yield of pre-processing step (De Meester et al., 2019; Van Eygen et al., 2016) while the $TCs_{recycling}$ refer to the yield of washing and extrusion (Lase et al., 2021).

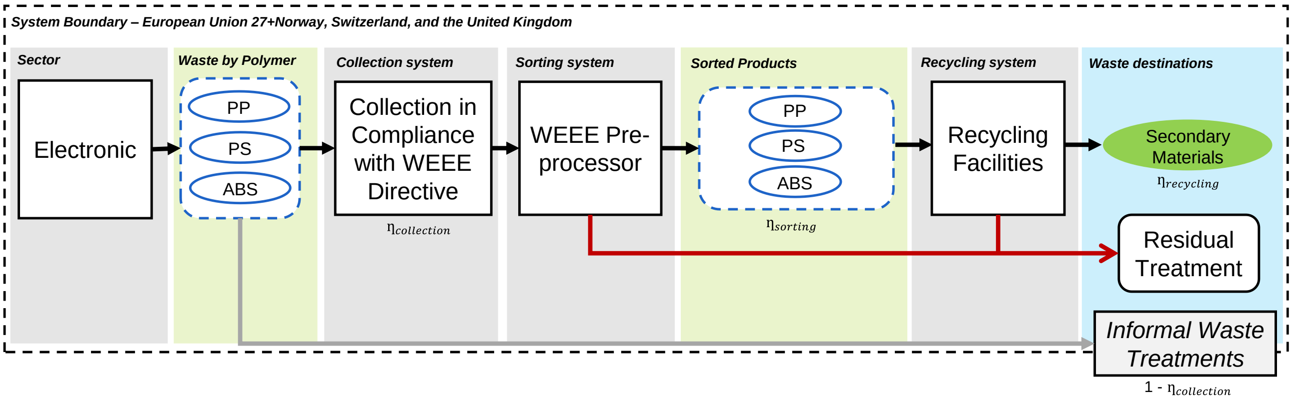

Figure A.6 Process diagram of plastic waste management in electronic sector. η denotes the efficiency (in %) of each process.

Automotive sector: Plastics can be found in different parts of passenger cars, such as bumpers, dashboards, cushions, 'under-the-hood' parts (Maury et al., 2022). End-of-life vehicles (ELVs) are formally deregistered (the efficiency is represented by $\eta_{collection}$ in Figure A.7) and sent to authorized treatment facilities (ATFs), which serve as the formal channel for ELVs recycling in Europe, i.e., $TCs_{collection}$ in Figure 2.1 in the main text (Maury et al., 2022; Aigner, 2020). Otherwise, the ELVs can be sent for legal or illegal exports or end up in unknown whereabouts (the efficiency is represented by $1 - \eta_{collection}$ in Figure A.7) (Maury et al., 2022). The typical treatments of plastic waste from ELVs at the ATFs are depollution and dismantling, shredding, post-shredding treatment (PST), and MR (Maury et al., 2022; Aigner, 2020; Baldassarre et al., 2022). At ELVs dismantlers, some vehicle parts (e.g., bumper and seats) are manually dismantled for recycling (dismantling efficiency is represented by $\eta_{dismantling}$ in Figure A.7). The remaining ELVs parts, known as the 'hulks', are baled and transferred to the shredding plants, resulting in automotive shredder residues (ASR). The ASR from the ELV treatment can be divided into two streams: light fluffs and heavy fluffs (Maury et al., 2022; Buekens and Letcher, 2020), in which more than 85% of plastics are mostly found in the light fluffs (shredding efficiency to light fluffs is denoted as $\eta_{shredding}$ in Figure A.7). Thereafter, the light fluffs are sent to the PST to mechanically sort plastics into distinct polymer streams using magnetic, eddy current, and density-based separation techniques, i.e., the PST efficiency is denoted as η_{PST} in Figure A.7 (Maury et al., 2022; Aigner, 2020; Buekens and Letcher, 2020). After sorting, plastics are sent to MR facilities to be washed and extruded into recyclates (recycling efficiency is represented by $\eta_{recycling}$ in Figure A.7) (Maury et al., 2022; Baldassarre et al., 2022). As for the PUR, the rebonded flexible foam technique is applied, in which the polymer is shredded, coated with adhesive binder and converted into rebonded foam parts (ISOPA, 2021; Europur, 2016; ISOPA, n.d., Datta and Wloch, 2017). In Figure 2.1 in the main text, the $TCs_{sorting}$ for ELVs refer to the processes from dismantler, shredder, and PST that create sorted plastic streams, while the $TCs_{recycling}$ refer to the MR with washing and extrusion.

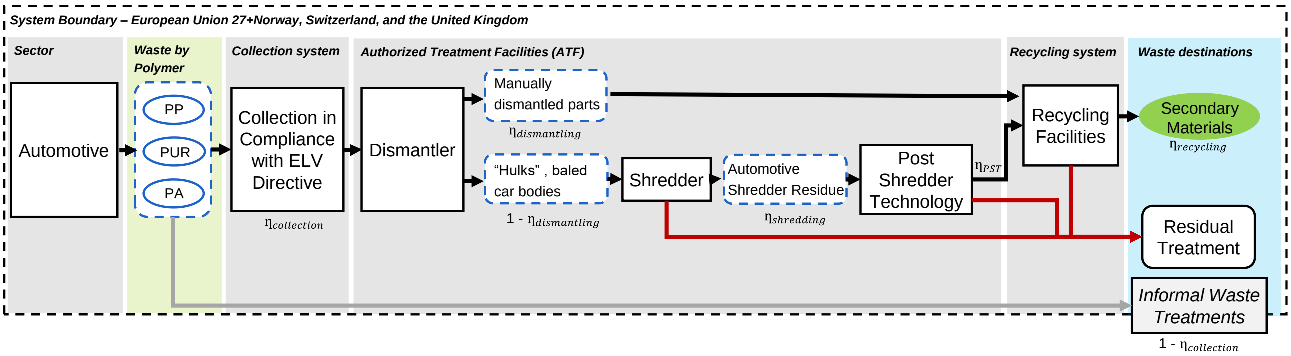

Figure A.7 Process diagram of plastic waste management in automotive sector. η denotes the efficiency (in %) of each process.

Building and construction sector: Plastic waste in construction and demolition waste (CDW) is typically collected alongside with other mixed materials (e.g., EPS for insulation is collected together with the glass wool). In practice, separate collection of CDW is done by waste management companies based on commercial agreements or specific take-back schemes and reverse logistic on the sites (collection efficiency is denoted as $\eta_{collection}$ in Figure A.8). Collection and (on-site) sorting occur at the same time, where CDW (incl. plastic waste) is either separated in a container as mixed construction waste, as recyclable, or per polymer type by the recyclers (Bendix et al., 2021; Gardner, 2020). When the CDW is collected as mixed construction waste in a container (represented by $1 - \eta_{collection}$ in Figure A.8), additional manual and mechanical separation of the materials can be done at the transfer station. Easy-to-identify objects such as window or door profiles, pipes, cables, and foam insulation are sorted by the 'pickers' for recycling, i.e., the efficiency is denoted as $\eta_{on-site\ sorting}$ in Figure A.8 (Gardner, 2020). At recycling facilities, the sorted plastics are shredded, washed, and regranulated by means of extrusion with melt filters (recycling efficiency is represented by $\eta_{recycling}$ in Figure A.8) (Bendix et al., 2021). In Figure 2.1 in the main text, the $TCs_{collection}$ and $TCs_{sorting}$ refer to the separate collection rate in a container for CDW, including on-site sorting and separation at transfer station. The $TCs_{recycling}$ refer to the MR yield by means of shredding, washing and extrusion.

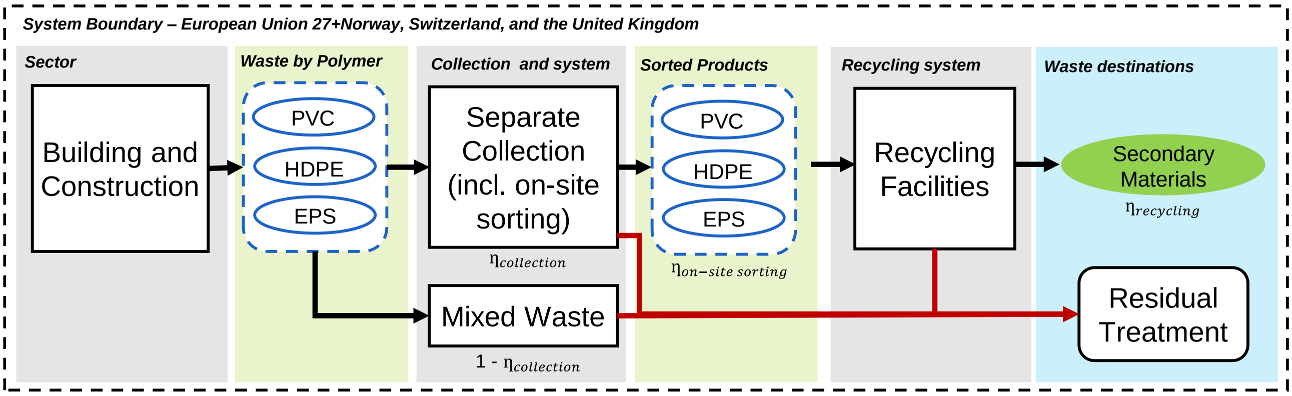

Figure A.8 Process diagram of plastic waste management in building and construction sector. η denotes the efficiency (in %) of each process.

Agriculture sector: In the absence of relevant legislation and waste management system, agriculture plastic waste (APW) are typically managed by a collective and voluntary approach between the farmers or growers and recyclers (European Commission, 2020b; Agriculture Plastic Environment, 2021). In practice, collection and sorting by polymer or product types of APW occur simultaneously and are prepared on the site (sometimes are even baled before the delivery to the waste operators), i.e., the efficiency is denoted as $\eta_{collection}$ and $\eta_{on-site\ sorting}$ in Figure A.9. Most of them are collected by the waste management operator for recycling by means of 'a bring' or 'a pickup' system depending on the region and farm sizes (Bauer, 2019). Here, the business-to-business relationships principle between the farmers and the waste operators is key in the overall APW management (European Commission, 2020b; Agriculture Plastic Environment, 2021). At the recycling facility, APW are shredded, washed, and extruded into recyclates, i.e., recycling efficiency is denoted as $\eta_{recycling}$ in Figure A.9 (European Commission, 2020b). In Figure 2.1 in the main text, the $TCs_{collection}$ and $TCs_{sorting}$ refer to the yield of on-site sorting by the farmers, while the $TCs_{recycling}$ refer to the yield of recycling processes (incl. shredding, washing, and extrusion).

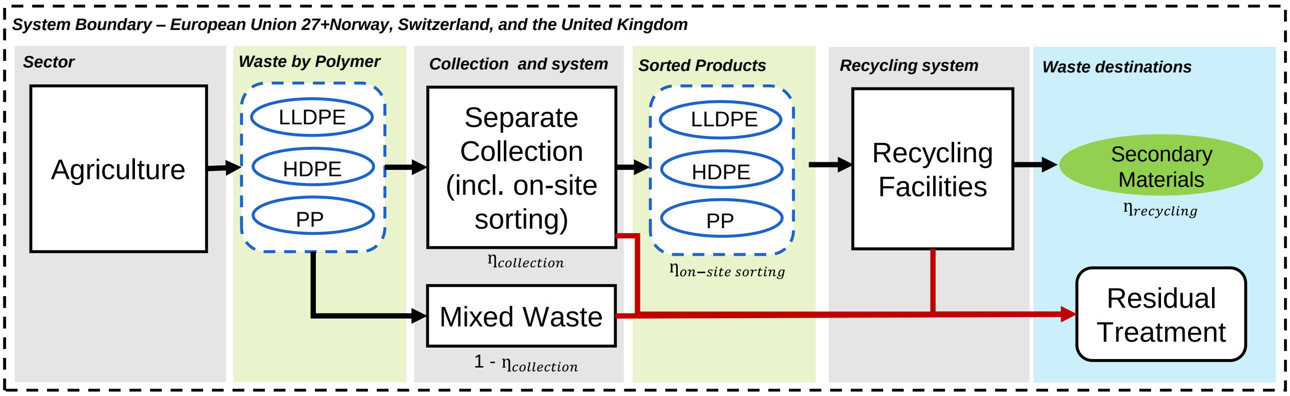

Figure A.9 Process diagram of plastic waste management in agriculture sector. η denotes the efficiency of each process.

Section A.2.2 Description of improvements in the waste management system in 2030

Packaging sector: It is projected that collection will increase and more plastic packaging waste sorting by polymer types would occur. For example, PP flex might be separated from the mixed film bales as shown by Bashirgonbadi et al. (2022). Moreover, more sorting by colors (e.g., PET bottle blue, PET bottle green, PET bottle opaque, etc.) will be realized. A fraction of plastic packaging waste that is actually collected in the mixed waste streams would also be sorted, i.e., known as post-sorting municipal solid waste (MSW), and sent for recycling, as suggested by Brouwer et al. (2018) in the Dutch plastic waste management infrastructure.

Automotive sector: It is forecasted that post-shredder technology (PST) will be used to sort more polymer types (from the already currently sorted in the ATFs such as polypropylene (PP) and polyethylene (PE)). Looking at the relatively high labor cost in Europe (Europur, 2016), improvement of sensor- and density-based separation techniques (e.g., near infrared sorting) are more likely than manual dismantling of ELVs. Hence, it is assumed that polymers such as PUR and polyamide (PA) from ELVs will be sorted using PST and will be sent for recycling in the future, while manual dismantling will only reach 17% maximum efficiency. Moreover, it is assumed that PUR recycling (i.e., rebonded technique) will not be improved by 2030.

Electronic sector: WEEE pre-processing is expected to improve, with more manual dismantling and depollution and mechanical sorting capabilities. This is also improved by a better design of electronic products to be manually dismantled, as also investigated in the study of Lase et al. (2021).

Building and construction and agriculture sectors: Waste management infrastructure of plastic waste in CDW and APW is expected to be established with a better on-site sorting by the workers, farmers, and growers in the future (Bauer, 2019; Agriculture Plastic Environment, 2021). Obviously, the speed of improvements in these sectors will be accelerated if new regulations and targets are to be put into force in the future (e.g., mandatory separate collection or plastic or minimum recycling targets as shown in the other sectors).

SECTION 3: LIST OF TRANSFER COEFFICIENTS (TC) USED FOR MATERIAL FLOW ANALYSIS (MFA) MODEL IN THE 2018 SCENARIO (S0 IN THE MAIN TEXT)

Table A.1 List of transfer coefficients (TCs) used in MFA model for the packaging sector for the 2018 scenario (S0 in the main text). The Triangular distribution of plastic packaging waste sorting to mixed plastic bales is 6% (min.) , 8% (mode), and 10% (max), estimated from Brouwer et al. (2018), Kleinhans et al. (2020), and Picuno et al. (2021). PTT stands for pots, trays, and tubes packaging products.

Polymer Type	**Products**	**Transfer Coefficients**			**Source**
		Min	*Mode*	*Max*	
$\eta_{collection}$					
LDPE	Monolayer flexible packaging	27%	38%	56%	Eriksen et al., 2020, Kawecki et al., 2018; Watkins et al., 2020; Hestin et al., 2017, Roosen et al., 2022
	Multilayer flexible packaging	27%	38%	56%	
HDPE	Bottles, containers	21%	44%	76%	
	PTTs	21%	44%	76%	
PP	Monolayer flexible packaging	15%	17%	38%	
	Multilayer flexible packaging	15%	17%	38%	
	Bottles, containers	29%	38%	45%	
	PTTs	29%	38%	45%	
PET	Beverage bottles	55%	58%	79%	
	Trays	25%	39%	55%	
PS	PTTs and dairy products	2%	17%	56%	
EPS	Food packaging	2%	15%	56%	
$^{1}\eta_{primary\ bales}$					
LDPE	Monolayer flexible packaging	21%	79%	89%	Antonopoulos et al., 2021; Watkins et al., 2020; Kleinhans et al., 2021
	Multilayer flexible packaging	0%	0%	0%	
HDPE	Bottles, containers	53%	76%	91%	
	PTTs	53%	76%	91%	
PP	Monolayer flexible packaging	0%	0%	0%	
	Multilayer flexible packaging	0%	0%	0%	
	Bottles, containers	31%	50%	89%	
	PTTs	31%	50%	89%	
PET	Beverage bottles	45%	89%	97%	
	Trays	45%	58%	91%	

PS	PTTs and dairy products	31%	48%	79%	
EPS	Food packaging	31%	48%	79%	
[2] $\eta_{secondary\ bales}$					
LDPE	Monolayer flexible packaging	0%	0%	0%	Antonopoulos et al., 2021; Watkins et al., 2020; Kleinhans et al., 2021
	Multilayer flexible packaging	21%	79%	89%	
HDPE	Bottles, containers	3%	4%	5%	
	PTTs	3%	4%	5%	
PP	Monolayer flexible packaging	21%	50%	89%	
	Multilayer flexible packaging	21%	50%	89%	
	Bottles, containers	3%	4%	5%	
	PTTs	3%	4%	5%	
PET	Beverage bottles	0%	0%	0%	
	Trays	0%	0%	0%	
PS	PTTs and dairy products	0%	0%	0%	
EPS	Food packaging	0%	0%	0%	
[3] $\eta_{recycling}$ of primary bales (mechanical)					
LDPE	Monolayer flexible packaging	50%	71%	94%	Antonopoulos et al., 2021, Kawecki et al., 2018, Brouwer et al., 2018
	Multilayer flexible packaging	0%	0%	0%	
HDPE	Bottles, containers	70%	84%	95%	
	PTTs	70%	84%	95%	
PP	Monolayer flexible packaging	0%	0%	0%	
	Multilayer flexible packaging	0%	0%	0%	
	Bottles, containers	53%	71%	95%	
	PTTs	53%	71%	95%	
PET	Beverage bottles	63%	80%	95%	
	Trays	63%	80%	95%	
PS	PTTs and dairy products	57%	66%	90%	
EPS	Food packaging	57%	66%	90%	
[4] $\eta_{recycling}$ of secondary bales (mechanical)					
LDPE	Monolayer flexible packaging	0%	0%	0%	Lase et al., 2022, Civancik-Uslu et al., 2021, Watkins et al., 2020
	Multilayer flexible packaging	50%	59%	71%	
HDPE	Bottles, containers	50%	59%	71%	
	PTTs	50%	59%	71%	
PP	Monolayer flexible packaging	50%	59%	71%	
	Multilayer flexible packaging	50%	59%	71%	

	Bottles, containers	50%	59%	71%
	PTTs	0%	0%	0%
PET	Beverage bottles	0%	0%	0%
	Trays	0%	0%	0%
PS	PTTs and dairy products			
EPS	Food packaging			

[1] *LDPE – Monolayer flexible, HDPE – Bottles, containers, and PTTs, and PP – Bottles, containers, and PTTs are forwarded into PE Film bales, PE Rigid bales, and PP rigid bales respectively*

[2] *LDPE – Multilayer flexible packaging and PP – Monolayer and multilayer flexible packaging are forwarded into MPO Flex bales. HDPE – Bottles, container, and PTTs as well as PP – Bottles, containers, and PTTs are forwarded to MPO Rigid bales (as secondary bales)*

[3] *Mechanical recycling of primary bales refer to the recycling of plastic packaging waste in PE Film, HDPE Rigid, PP Rigid, PET, and PS Bales*

[4] *Mechanical recycling of secondary bales refer to the recycling of plastic packaging in MPO Flex and MPO Rigid bales*

Table A.2 List of transfer coefficients (TCs) used in MFA model for the electronic sector in the 2018 scenario (S0 in the main text).

Polymer Type	Products	Transfer Coefficients			Source
		Min	*Mode*	*Max*	
[1]$\eta_{collection}$					
PP	Dishwashers, laundry machines, dryers, etc.	37%	41%	54%	Watkins et al., 2020, Accili et al., 2019
	Food processors, kettles, vacuum cleaners, etc.	28%	32%	54%	
PS	Fridges, etc.	37%	41%	54%	
ABS	Vacuum cleaners, etc.	28%	32%	54%	
$\eta_{sorting}$					
PP	Dishwashers, laundry machines, dryers, etc.	44%	66%	75%	Lase et al., 2021, Watkins et al., 2020, assumption for the max values
	Food processors, kettles, vacuum cleaners, etc.	44%	66%	75%	
PS	Fridges, etc.	44%	66%	75%	
ABS	Vacuum cleaners, etc.	44%	66%	75%	
$\eta_{recycling}$ (mechanical)					
PP	Dishwashers, laundry machines, dryers, etc.	51%	60%	83%	Eriksen et al., 2020, Kawecki et al., 2018, Watkins et al., 2020
	Food processors, kettles, vacuum cleaners, etc.	51%	60%	83%	
PS	Fridges, etc.	51%	60%	83%	
ABS	Vacuum cleaners, etc.	51%	60%	83%	

[1] *The division between collection large equipment (fridges, dishwashers, etc.) and small equipment (vacuum cleaner, food processors, etc.)*

Table A.3 List of transfer coefficients (TCs) used in MFA model for the automotive sector in the 2018 scenario (S0 in the main text).

Polymer Type	Products	Transfer Coefficients			Source
		Min	*Mode*	*Max*	
$\eta_{collection}$					
PP	Bumper, body sides, dashboards, etc.	60%	80%	87%	Maury et al., 2022, Watkins et al., 2020
PUR	Seat paddings, cushions, etc.	60%	80%	87%	
PA	Battery casing, break hoses, etc.	60%	80%	87%	
$\eta_{dismantling}$					
PP	Bumper, body sides, dashboards, etc.	7%	10%	13%	Maury et al., 2022, Watkins et al., 2020; Aigner, 2020
PUR	Seat paddings, cushions, etc.	7%	10%	13%	
PA	Battery casing, break hoses, etc.	7%	10%	13%	
$\eta_{shredding}$					
PP	Bumper, body sides, dashboards, etc.	88%	90%	95%	Aigner, 2020 ; Assumption for the max. values
PUR	Seat paddings, cushions, etc.	88%	90%	95%	
PA	Battery casing, break hoses, etc.	88%	90%	95%	
[1] η_{PST}					
PP	Bumper, body sides, dashboards, etc.	40%	45%	50%	Maury et al., 2022; Aigner, 2020
PUR	Seat paddings, cushions, etc.	0%	0%	0%	
PA	Battery casing, break hoses, etc.	0%	0%	0%	
[2] $\eta_{recycling}$ (mechanical)					
PP	Bumper, body sides, dashboards, etc.	45%	56%	70%	Maury et al., 2022; Watkins et al., 2020; ISOPA, n.d; ISOPA, 2021
PUR	Seat paddings, cushions, etc.	45%	56%	70%	
PA	Battery casing, break hoses, etc.	0%	0%	0%	

[1] *Post shredding technology (PST) does not sort PUR and PA, mainly only sorts PO and styrenic polymers*

[2] *PUR is effectively recycled when manually dismantled through rebond flexible technique recycling, estimated from ISOPA (ISOPA, n.d; ISOPA, 2021)*

Table A.4 List of transfer coefficients (TCs) used in MFA model for the building and construction sector in the 2018 scenario (S0 in the main text).

Polymer Type	Products	Transfer Coefficients			Source
		Min	*Mode*	*Max*	
$\eta_{collection}$					
PVC	Window profiles, doors, flooring, etc.	60%	75%	100%	Kawecki et al., 2018, Watkins et al., 2020. Assumption for the min. values
HDPE	Pipes, etc.	60%	73%	100%	
EPS	Insulation, etc.	60%	68%	100%	
$\eta_{on-site\ sorting}$					
PVC	Window profiles, doors, flooring, etc.	75%	85%	90%	Kawecki et al., 2018, Watkins et al., 2020
HDPE	Pipes, etc.	6%	23%	33%	
EPS	Insulation, etc.	4%	9%	14%	
$\eta_{recycling}$ (mechanical)					
PVC	Window profiles, doors, flooring, etc.	50%	55%	80%	Eriksen et al., 2020; Kawecki et al., 2018, Watkins et al., 2020
HDPE	Pipes, etc.	50%	70%	80%	
EPS	Insulation, etc.	50%	70%	80%	

Table A.5 List of transfer coefficients (TCs) used in MFA model for the agriculture sector in the 2018 scenario (S0 in the main text).

Polymer Type	Products	Transfer Coefficients			Source
		Min	*Mode*	*Max*	
$\eta_{collection}$					
LDPE	Mulching, silage films, etc.	60%	66%	98%	Watkins et al. (2020) ; Eriksen et al. (2020)
HDPE	Nets, bale wraps, etc.	50%	58%	60%	
PP	Twines, etc.	50%	53%	60%	
$\eta_{on-site\ sorting}$					
LDPE	Mulching, silage films, etc.	75%	83%	100%	Kawecki et al., 2018, Watkins et al., 2020; Assumption for the min. values
HDPE	Nets, bale wraps, etc.	2%	5%	28%	
PP	Twines, etc.	90%	99%	100%	
$\eta_{recycling}$					
LDPE	Mulching, silage films, etc.	60%	78%	90%	Kawecki et al., 2018, Watkins et al., 2020 ; Assumption for the min values of HDPE and PP
HDPE	Nets, bale wraps, etc.	60%	79%	90%	
PP	Twines, etc.	60%	63%	90%	

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SECTION 4: PROJECTIONS OF IMPROVED COLLECTION RATES, SORTING YIELD, AND MECHANICAL RECYCLING YIELD OF PLASTIC WASTE IN 2030

Figure A.10–A.12 show the projections of collection rates in 2030 based on historical data from Hestin et al. (2017), Eurostat (2021; 2022b).

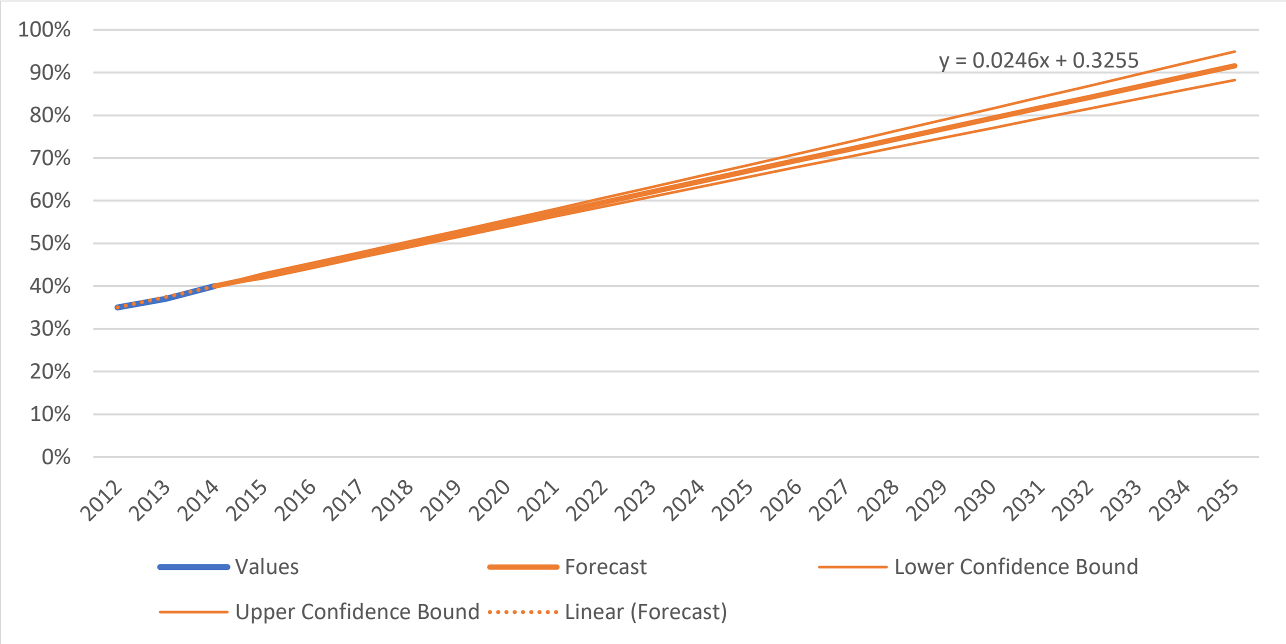

Figure A.10 Projection of separate collection rates (either as single stream or commingled) of plastic packaging waste based on Hestin et al. (2017) for the period 2012 – 2014.

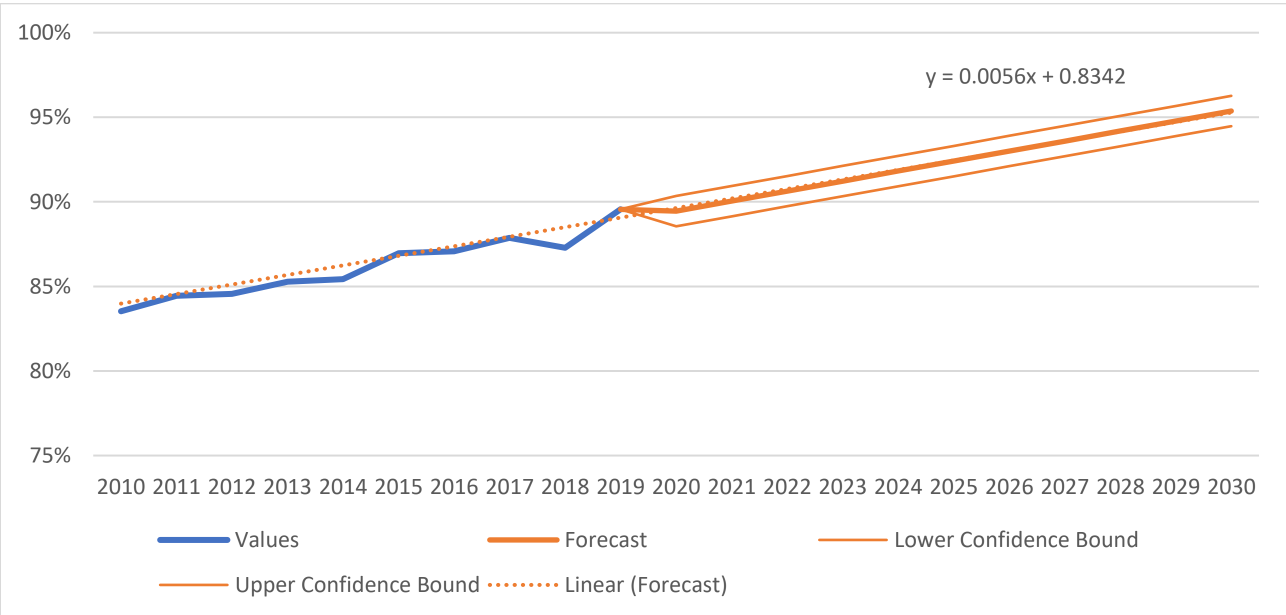

Figure A.11 Projection of future ELVs collection rates in compliance with ELV Directive calculated as the ratio between the reported mass of recycled ELV and the ELV waste generation for the period 2010 – 2019 Eurostat (2021).

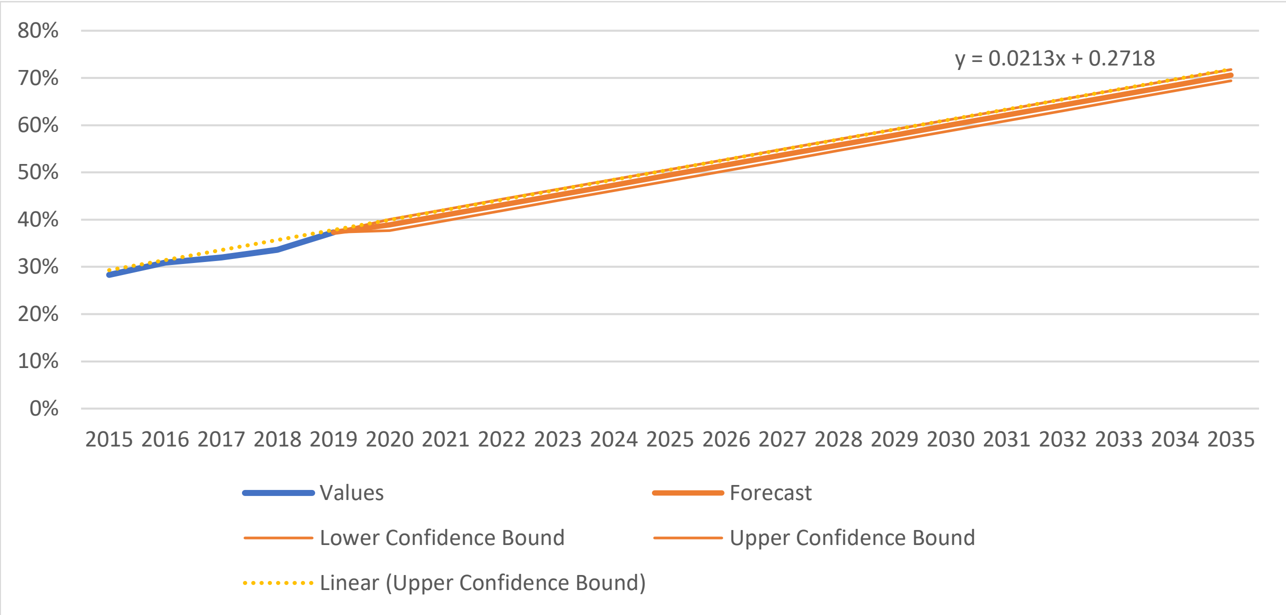

Figure A.12 Projection of future WEEE collection rates in compliance with WEEE Directive. The collection rates are calculated as the ratio between the reported mass of recycled WEEE and the WEEE waste generation for the period 2015 – 2019 Eurostat (2022b).

Figure A.13 – A.22 show the projections of sorting and recycling yields in 2030 based on the assumption that maximum (i.e., based on the best known sorting and recycling techniques) efficiencies will be reached in 2035, retrieved from Antonopoulos et al. (2021).

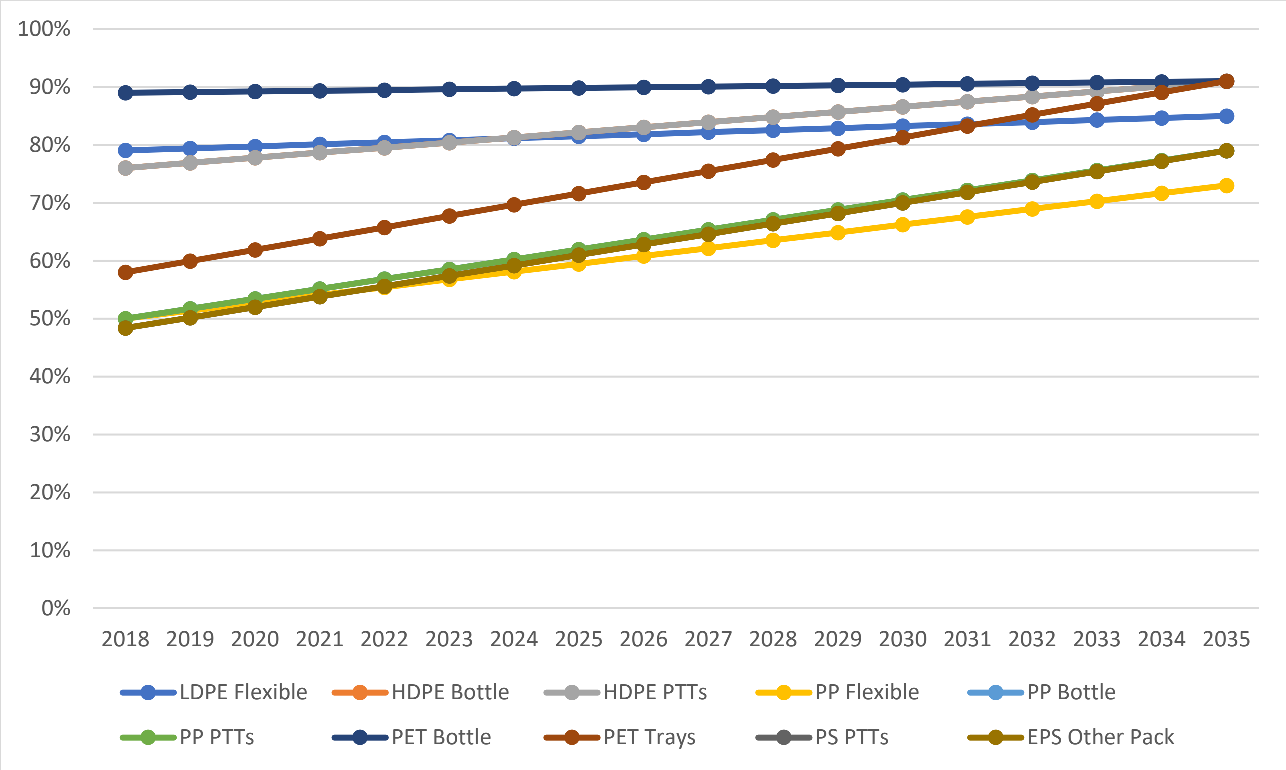

Figure A.13 Projections of improved sorting yields of separately collected at source plastic packaging waste from 2018 to 2035.

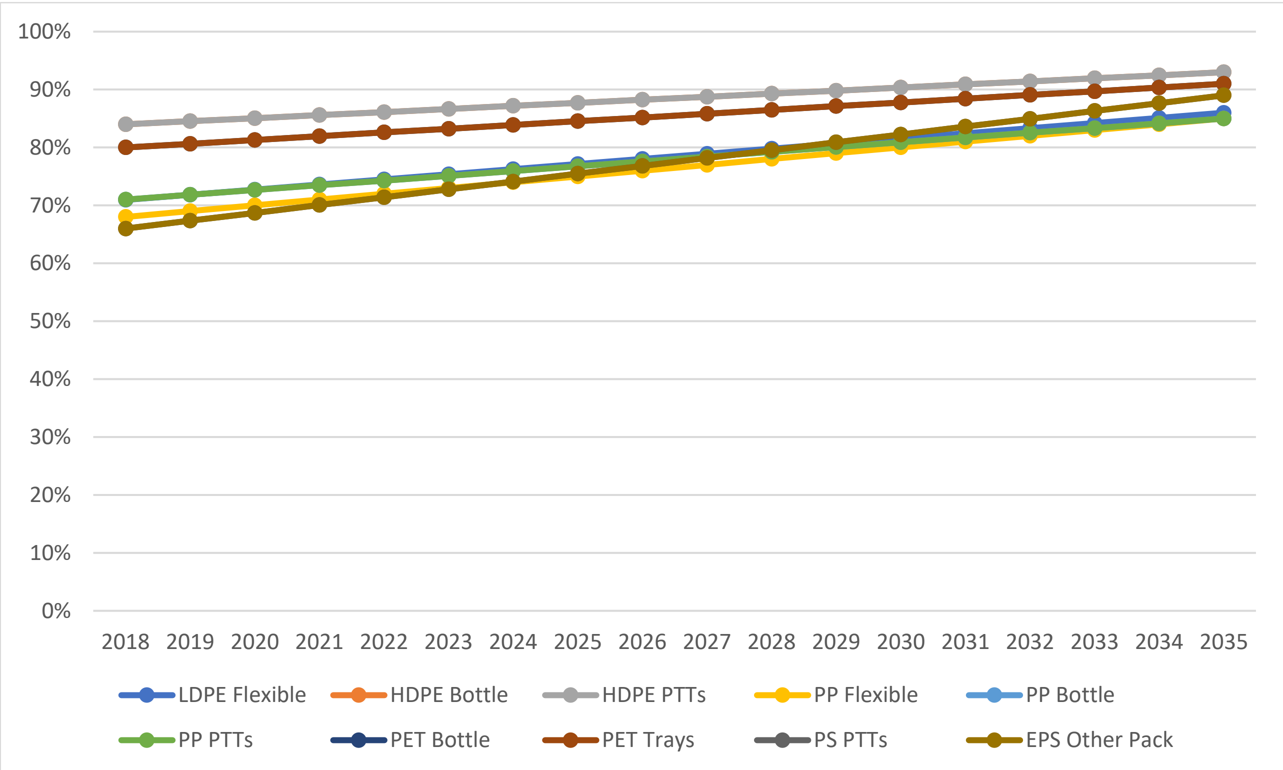

Figure A.14 Projections of improved recycling yields of sorted plastic packaging waste from 2018 to 2035.

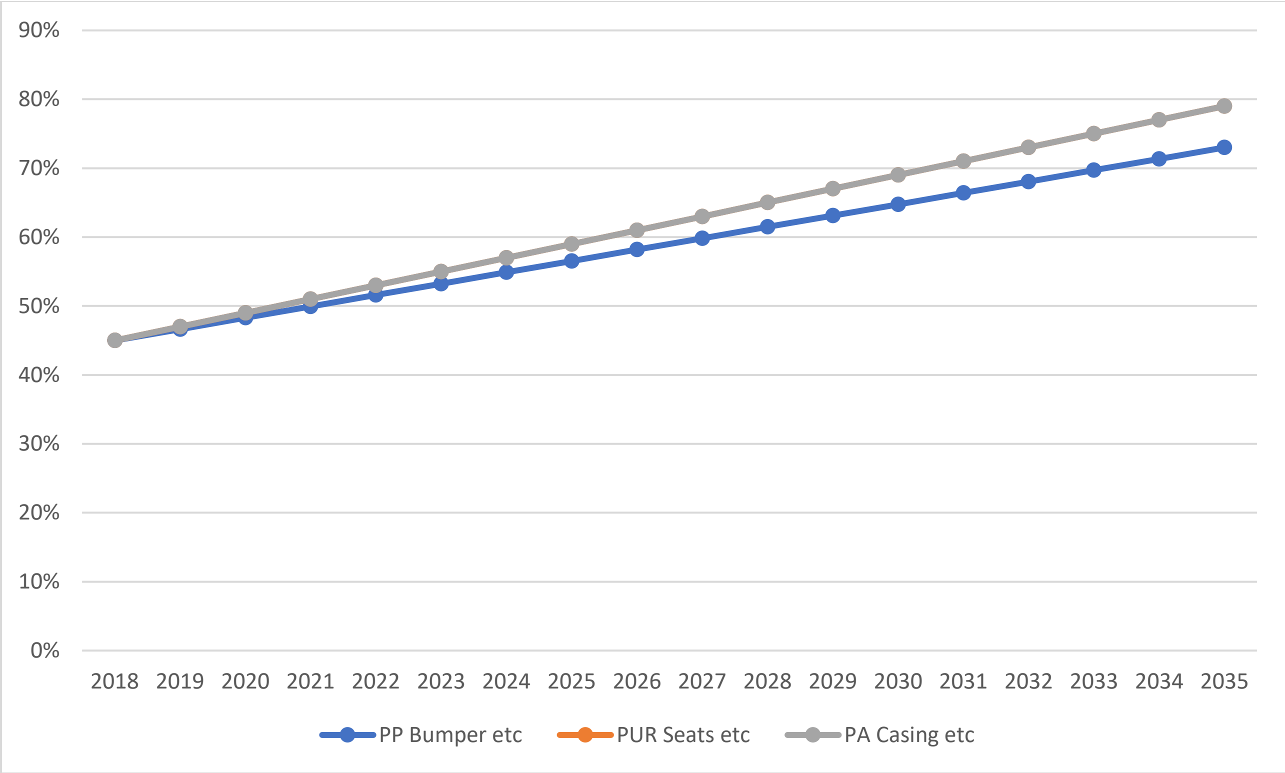

Figure A.15 Projections of improved post-sorting technologies' yields of automotive shredder residue ('light fluffs' fraction) from 2018 to 2035. The yield of post-sorting PUR (orange curve) is identical to post-sorting PA (grey curve).

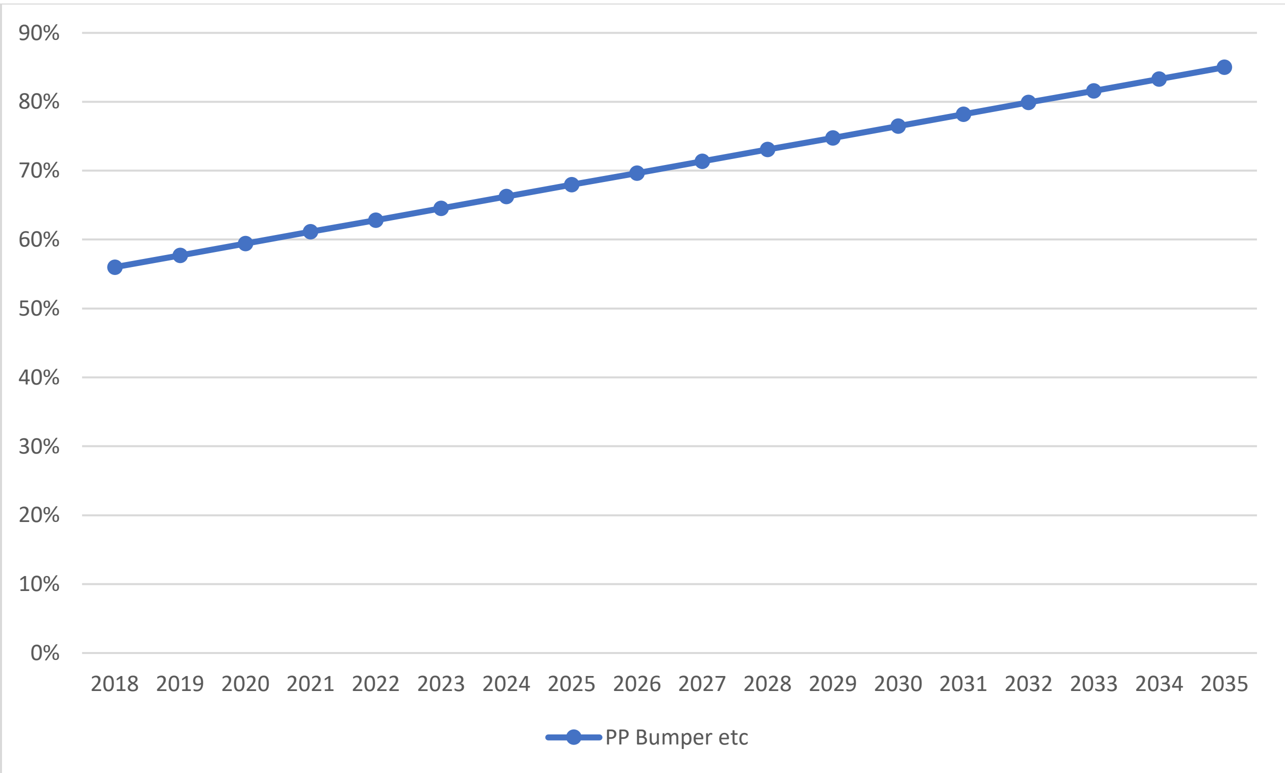

Figure A.16 Projections of improved recycling yields of sorted PP from ELVs in from 2018 to 2035.

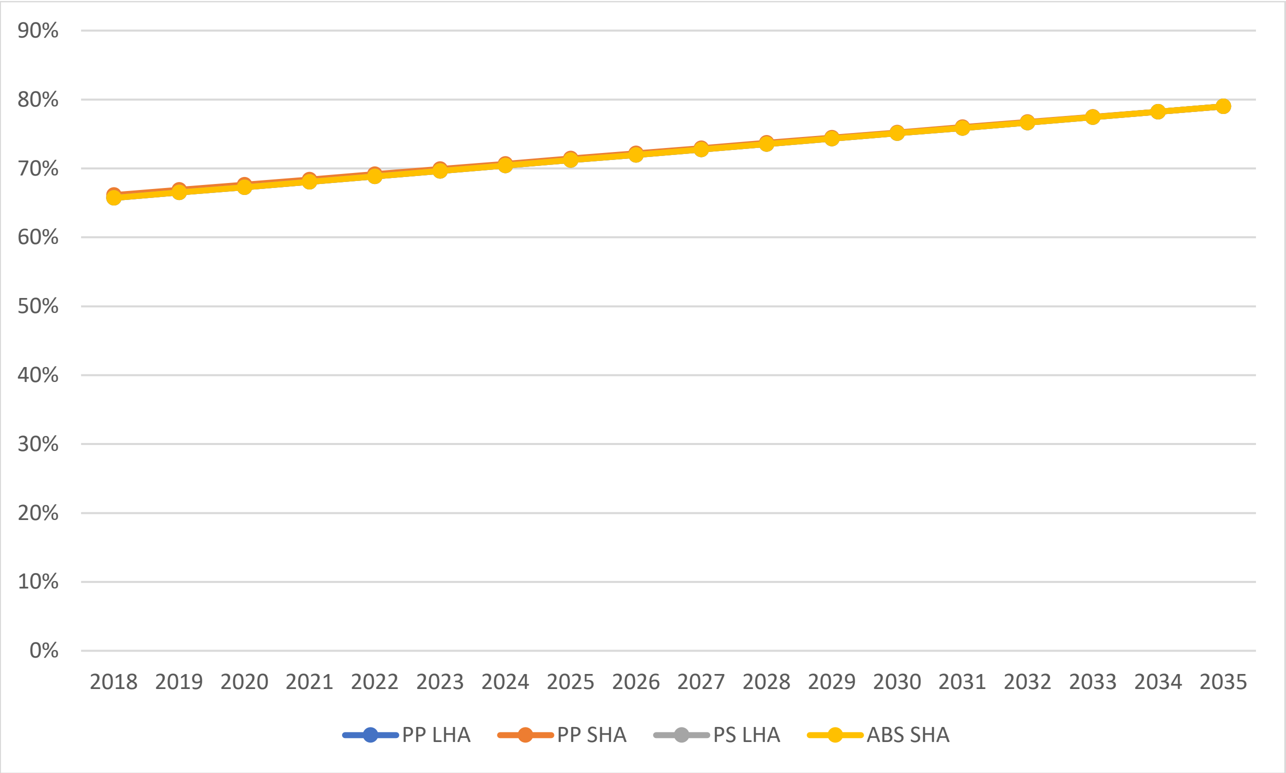

Figure A.17 Projections of improved WEEE preprocessing yields of collected WEEE from 2018 to 2035.

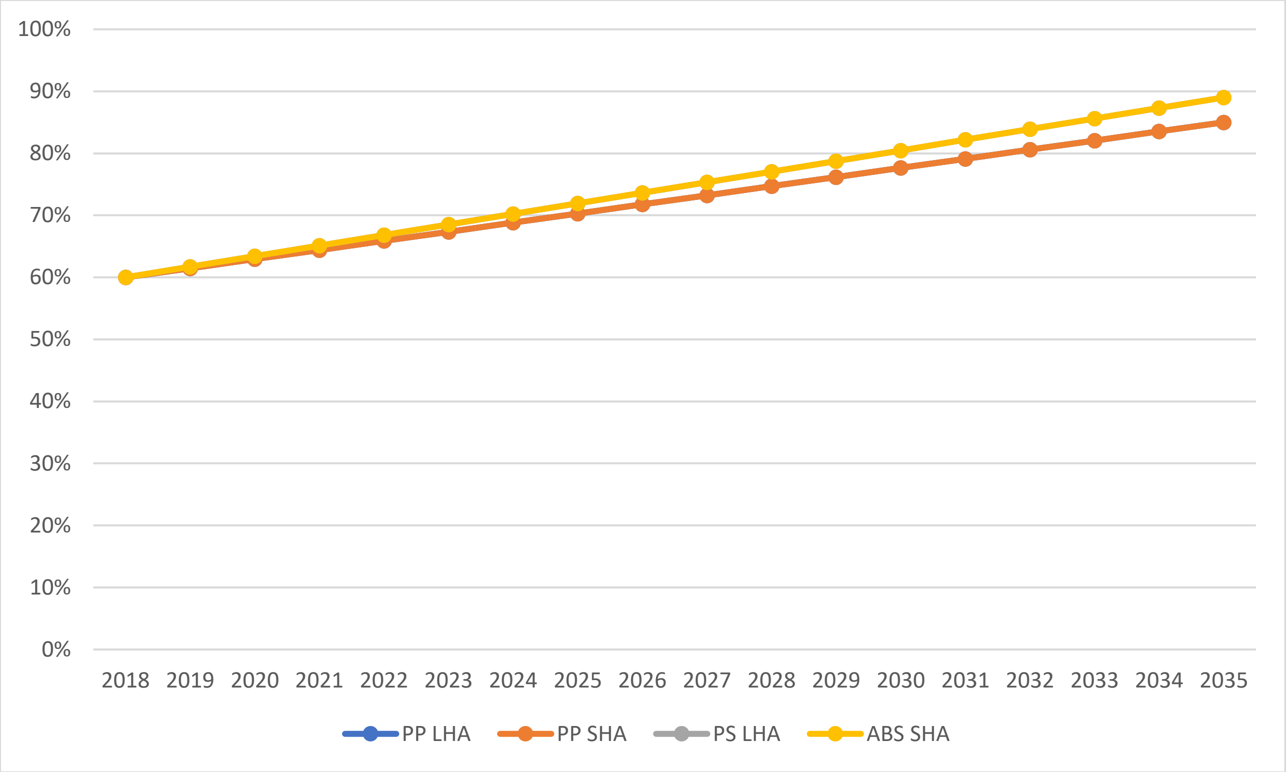

Figure A.18 Projections of improved recycling yields of sorted plastic from WEEE from 2018 to 2035.

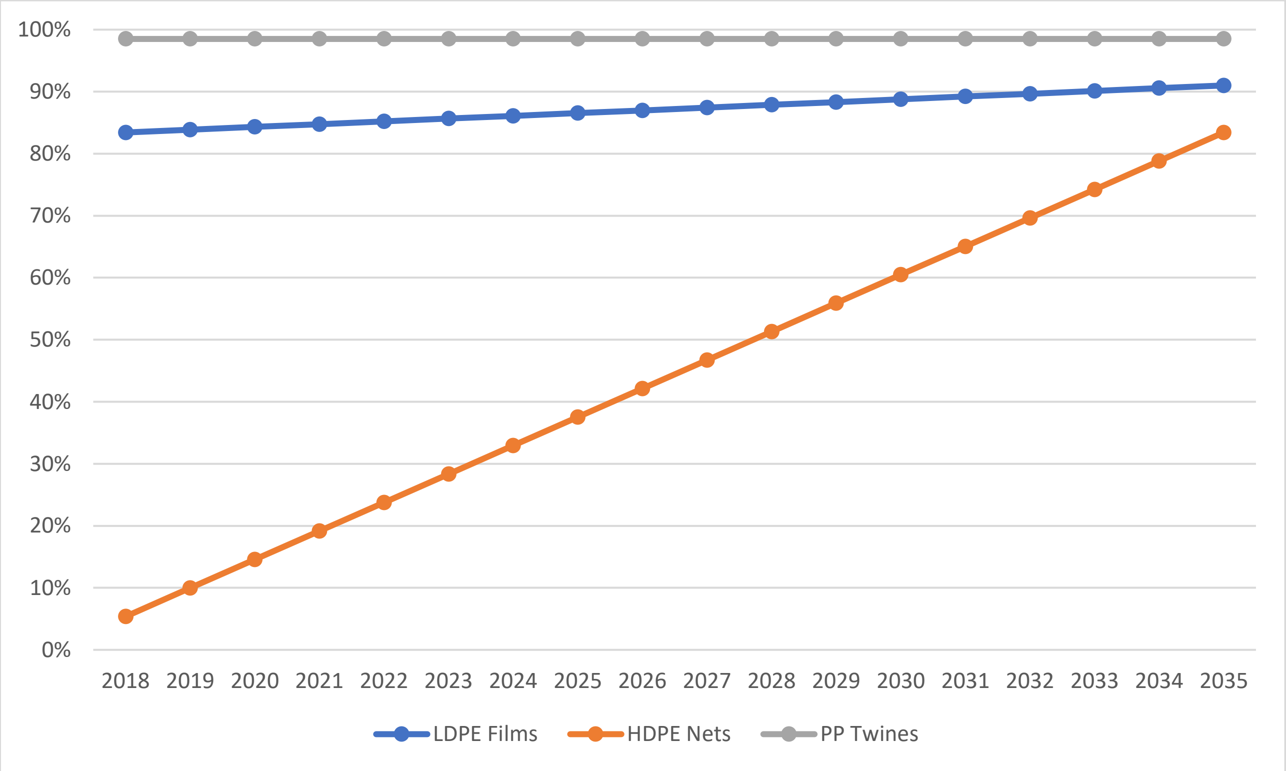

Figure A.19 Projections of improved on-site sorting yields of APW from 2018 to 2035.

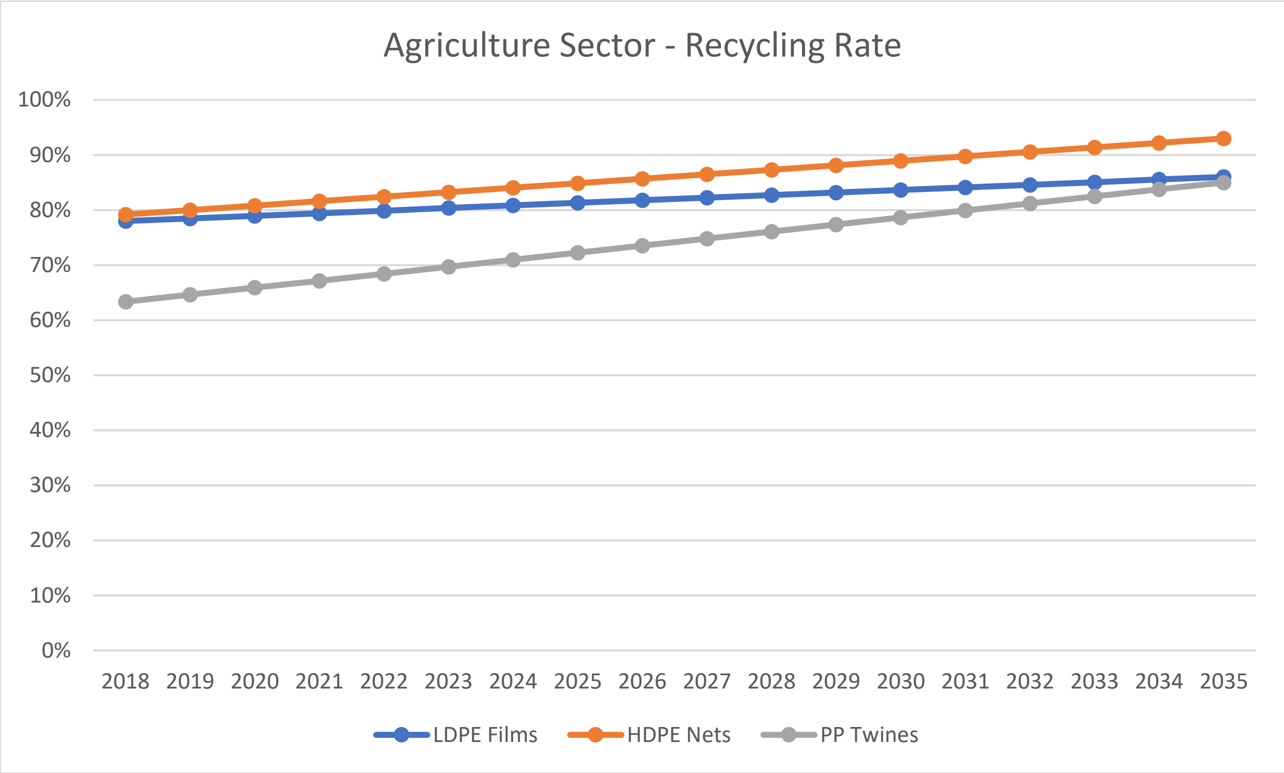

Figure A.20 Projections of improved recycling yields of sorted APW from 2018 to 2035.

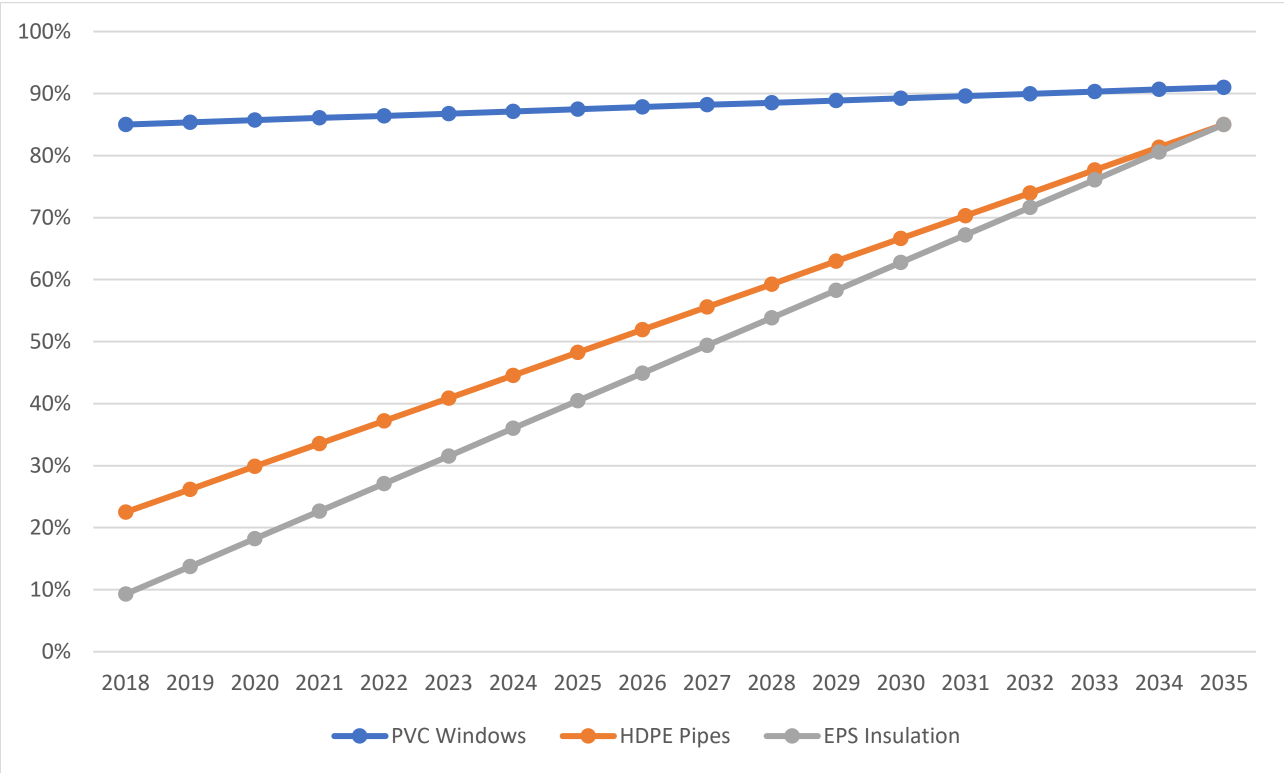

Figure A.21 Projections of improved on-site sorting yields of plastic waste in CDW from 2018 to 2035.

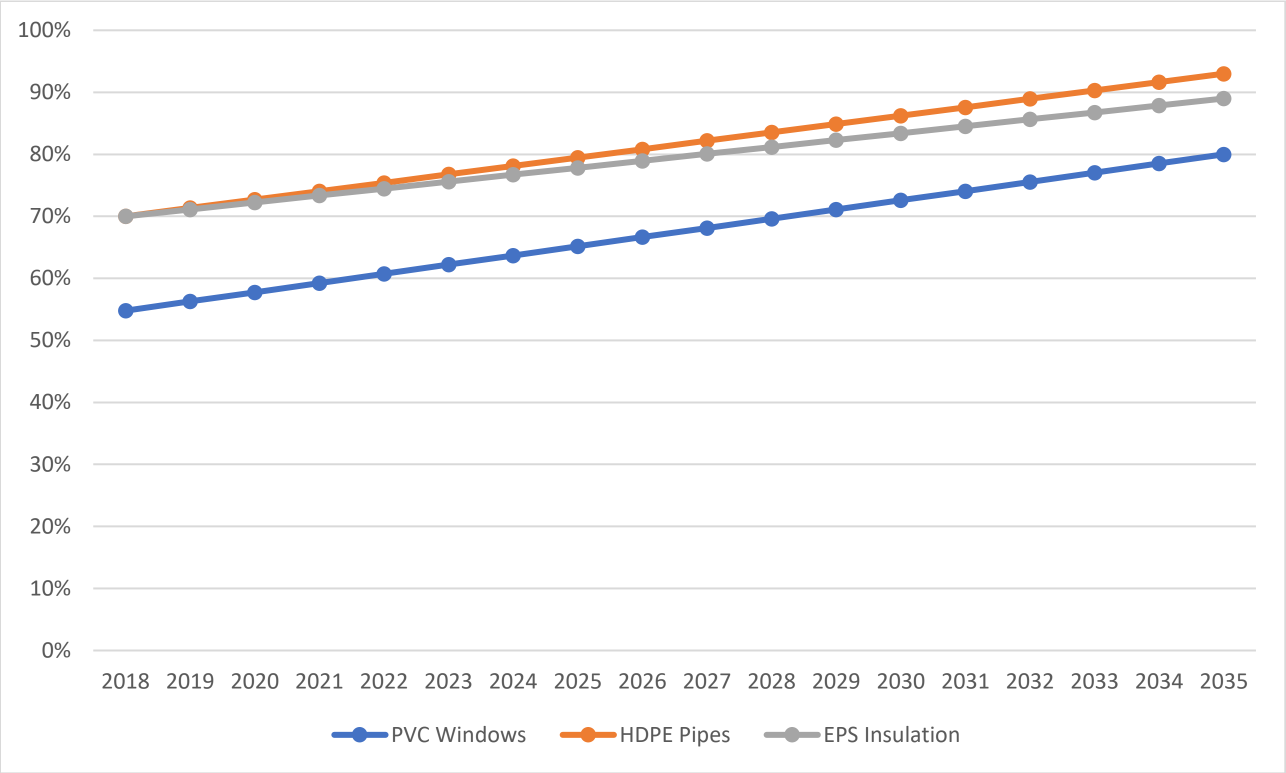

Figure A.22 Projections of improved recycling yields of sorted plastic waste in CDW from 2018 to 2035.

SECTION 5: LIST OF TRANSFER COEFFICIENTS (TC) USED FOR MATERIAL FLOW ANALYSIS (MFA) MODEL IN 2030 SCENARIOS

Table A.6 – A.10 show the TCs used in the MFA for the different CR options. Note that the TCs for polymerization are assumed to be 95% (min.), 98% (mode), and 99% (max.), as suggested by Jeswani et al. (2021).

Table A.6 List of transfer coefficients (TCs) used in MFA model for pyrolysis mixed polyolefin (MPO) in 2030, estimated from (Civanvik-Uslu, 2021; Kusenberg et al., 2022a; Kusenberg et al., 2022b; Genuino et al., 2022; Kusenberg et al., 2022d; Zhao et al., 2021; Ghalomi et al., 2021).

Cracking and condensation: Yield to Pyrolysis Oil			**Distillation: Yield to naphtha**			**Steam Crackers: Yield to monomer**		
Min	*Mode*	*Max*	*Min*	*Mode*	*Max*	*Min*	*Mode*	*Max*
70%	80%	89%	50%	56%	65%	45%	53%	55%

Table A. 7 List of transfer coefficients (TCs) used in MFA model for pyrolysis of styrenic polymers in 2030, estimated from Civancik-Uslu et al. (2021) and Zayoud et al. (2022)

Cracking and condensation: Yield to Pyrolysis Oil			**Distillation: Yield to styrene**		
Min	*Mode*	*Max*	*Min*	*Mode*	*Max*
89%	92%	95%	36%	45%	56%

Table A.8 List of transfer coefficients (TCs) used in MFA model for gasification in 2030, estimated from Mastellone (2019), Lopez et al. (2018), Brems et al. (2015), Mastellone and Zaccariello (2013), Arena (2012), Zhao et al. (2021), and Lee et al. (2008).

Gasification process: Yield to Syngas			**Fischer-Tropsch Synthesis: Yield to Monomer**		
Min	*Mode*	*Max*	*Min*	*Mode*	*Max*

70%	88%	90%	35%	48%	55%

Table A. 9 List of transfer coefficients (TCs) used in MFA model for chemical depolymerization of PET, PUR and PA, estimated from Kol et al. (2021), Vollmer et al. (2020), Sinha and Patel (2010), Shen et al. (2010), and Nikje et al. (2011).

Chemical Depolymerization: Yield to monomer

	Min	*Mode*	*Max*
PET	80%	98%	100%
PUR	60%	75%	90%
PA	60%	75%	90%

Table A. 10List of transfer coefficients (TCs) used in MFA model for solvent-based purification of PVC and EPS, estimated from Kol et al. (2021), Ügdüler et al. (2020), Walker et al. (2020) and Naviroj et al. (2019).

Solvent-based purification: Yield to polymer

	Min	*Mode*	*Max*
PVC	91%	95%	98%
EPS	88%	95%	98%

SECTION 6: WASTE QUANTITIES IN 2018 AND 2030

Table A11 shows the waste quantity as input to the MFA model in the 2018 scenario (S0 in the main text), and the 2030 scenarios (S1 – S5 in the main text).

Table A.11 Waste categories and quantities (in kt) considered in this study, adapted from Watkins et al. (2020), Plastic Europe (2019b), Hestin et al. (2017), European Commission (2020c), European Committee of Domestic Equipment Manufacturers (2017), and Maury et al. (2022). PTTs: Pots, trays, and tubes products. The projection of waste quantities in 2030 can be found in Appendix A–Section 7.

Sectors	**Waste categories**		**Waste quantities (in kt)**		
	Polymer type	*Relevant products*	*in 2018*	*in 2030*	*in 2030**
Packaging	LDPE	Monolayer flexible packaging	3,956	4,826	4,993
		Multilayer flexible packaging	989	1,207	1,373
	HDPE	Bottles and container	1,758	2,145	2,312
		PTTs	448	547	714
	PP (Flexible)	Monolayer flexible packaging	401	490	656
		Multilayer flexible packaging	100	122	289
	PP (Rigid)	Bottles and containers	296	362	528
		PTTs	756	922	1,088
	PET	Beverage bottles	2,984	3,641	3,807
		Trays	1,316	1,605	1,772
	PS	PTTs and diary packaging	842	1,027	1,194
	EPS	Food packaging	392	478	645
Total			*14,241*	*17,371*	*19,371*
Building & Construction	PVC	Window profile, flooring, doors, etc.	700	764	1,430
	HDPE	Pipes etc.	169	184	851
	EPS	Insulation etc.	140	152	819
Total			*1,009*	*1,101*	*3,100*
Automotive	PP	Bumper, body side, dashboards etc.	290	346	1,012
	PUR	Seats padding, cushions etc.	154	184	850
	PA	Battery casing, break hoses etc.	180	214	881
Total			*625*	*743*	*2,743*
Electronic	PP	Dishwasher, laundry machine, dryer etc.	351	405	905
		Food processing, hot water, vacuum cleaner etc.	49	57	557
	PS	Fridges etc.	230	266	766
	ABS	Vacuum cleaner etc.	150	173	673
Total			*780*	*901*	*2,901*

Agriculture	LDPE	Mulching, silage films etc.	576	661	1,327
	HDPE	Nets, bale wraps etc.	55	63	729
	PP	Twines etc.	80	92	758
Total			*711*	*815*	*2,815*
Grand total			**17,363**	**20,931**	**30,931**

**Waste quantities (in kt) in 2030, including the addition of 'missing plastic' after the estimated quantities of 'missing plastic' (i.e., 20% of total plastic demand) are normalized to the quantities of waste generation*

SECTION 7: PROJECTIONS OF WASTE QUANTITIES IN 2030

The quantities (in kt) of the selected polymer in 2030 are projected using linear regression based on the historical waste generation (for the period of around 2010 and 2018) found in statistical Eurostat databases (Eurostat, 2021; 2022a; 2022b; 2022c). Later, the information on the annual growth per sector is extracted and applied to estimate the quantity of waste generated (Table A.11). Figures A.23–A.27 show the raw data points of waste projections in different sector.

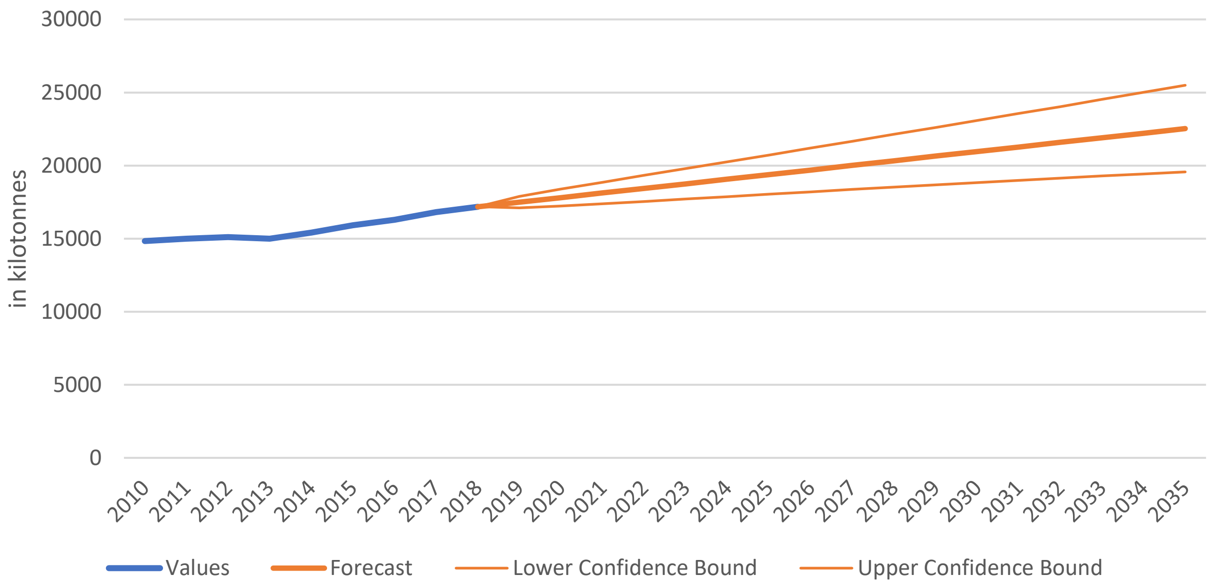

Figure A.23 The projection of packaging waste generation from 2018 to 2035 is estimated from the historical waste generation based on Eurostat (Eurostat, 2022a). The annual growth of packaging waste generation is estimated to be 1.4 – 1.8%.

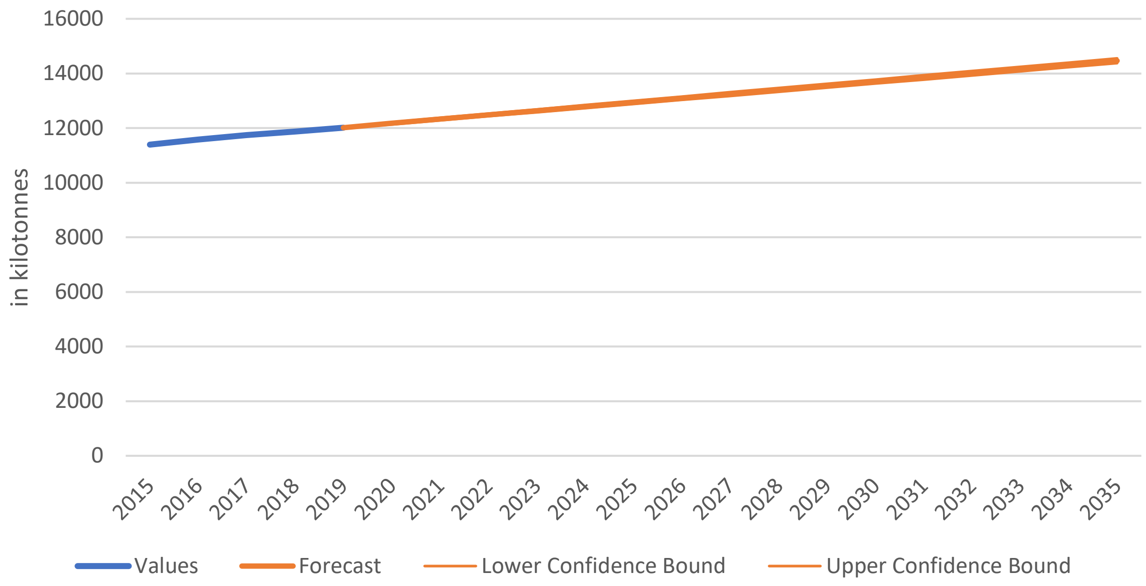

Figure A.24 The projection of WEEE generation from 2019 to 2035 is estimated from the historical WEEE generation based on Eurostat (Eurostat, 2022b). The annual growth of WEEE generation is estimated to be 1.1 – 1.2%.

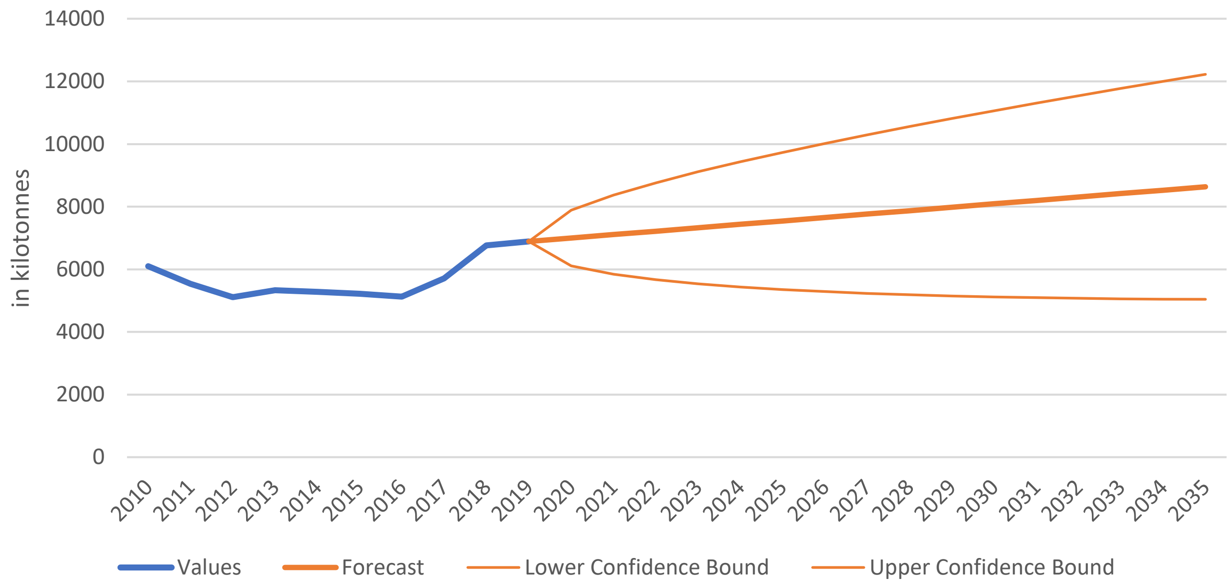

Figure A.25 The projection of ELV generation from 2019 to 2035 is estimated from the historical ELV generation based on Eurostat (Eurostat, 2021). The annual growth of ELV generation is estimated to be 1.3 – 1.6%.

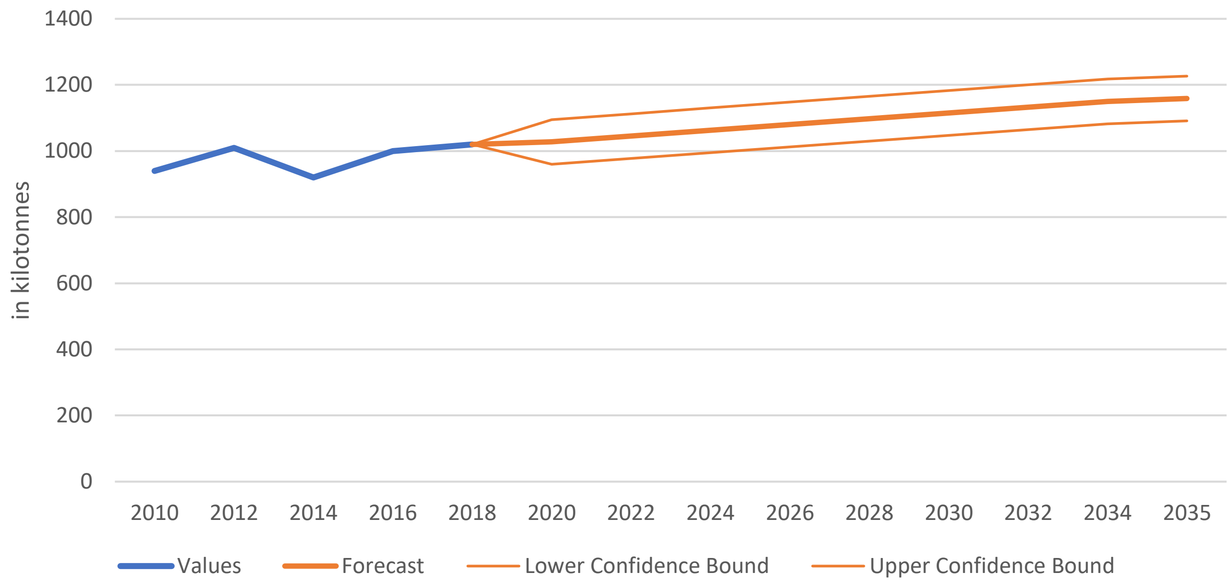

Figure A.26 The projection of building and construction waste generation from 2019 to 2035 is estimated from the historical data of NACE sector F (related to the construction activities) based on Eurostat (Eurostat, 2022c). The annual growth of CDW generation is estimated to be 0.8 – 0.9%.

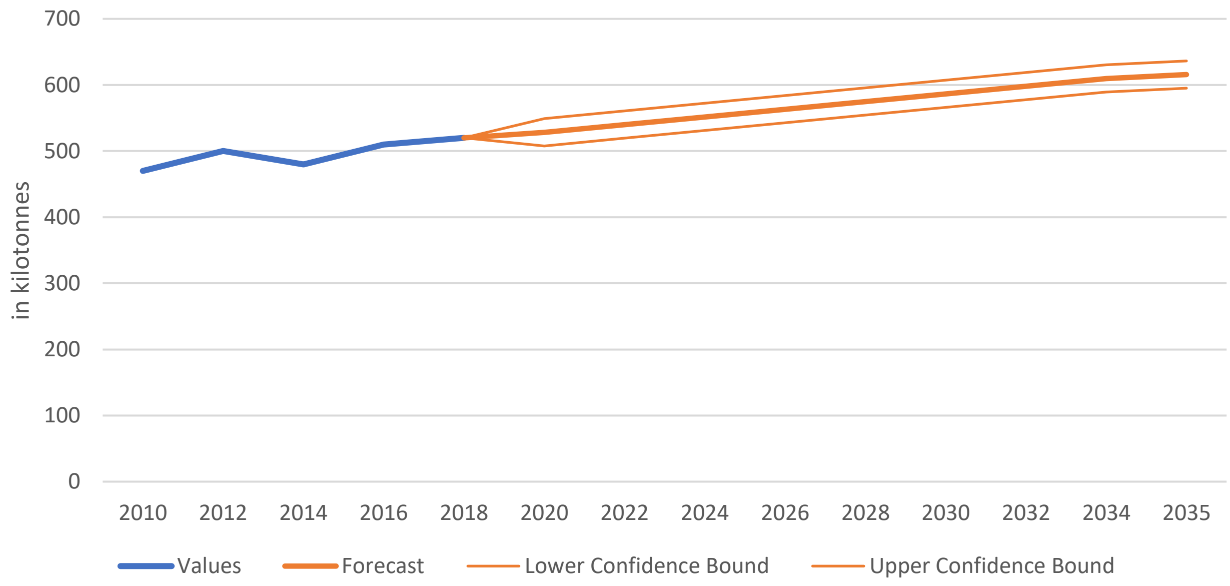

Figure A.27 The projection of building and construction waste generation from 2019 to 2035 is estimated from the historical data of NACE sector A (related to agriculture, forestry, and fishing activities) based on Eurostat (Eurostat, 2022c). The annual growth of APW generation is estimated to be 1.0 – 1.1%

SECTION 8: MARKET UPTAKE OF THE RECYCLED PLASTIC BASED ON WATKINS ET AL. (2020) AND EUROPEAN COMMISSION (2020)

Table A.12 summarizes the share of market uptake of recycled plastic in 2018, as reported by Watkins et al. (2020) and European Commission (2020c) that are used to estimate the recycled content availability in 2030. The study by Watkins et al. (2020) is used to estimate the market uptake of recycled plastic from the packaging, agriculture, and construction sector. The study by European Commission (2020c) is used to estimate the market uptake of recycled plastic from the electronic and automotive sector.

The market uptake of recycled plastics from CR is estimated by averaging the share of the other recycled plastic types in their respective sector (e.g., the share of market uptake of recycled plastics from CR in the agriculture sector is averaged from the market uptake of recycled LDPE, HDPE, and PP in the same sector). The share for polyolefin (PO) rigid regranulate is averaged from the market uptake of PE rigid and PP rigid regranulates.

Table A.12 The share of market uptake of recycled plastic as reported by Watkins et al. (2020) and European Commission (2020c) to estimate recycled content availability in 2030. MSW: municipal solid waste. CR: chemical recycling.

Sectors and recycled plastic types	Market uptake of recycled plastic from its respective sector (and recycled plastic types)					
	Agriculture	Packaging	Electronic	Construction	Automotive	Others
Packaging						
PE Films	0%	14%	0%	10%	0%	76%
PP Films	0%	14%	0%	10%	0%	76%
PO Films	0%	14%	0%	10%	0%	76%
PE Rigid	0%	20%	0%	80%	0%	0%
PP Rigid	0%	10%	10%	20%	50%	10%
PO Rigid	0%	15%	5%	50%	25%	5%
PET	0%	58%	0%	0%	0%	42%
PS	0%	0%	0%	0%	0%	100%
Post-sorted MSW	0%	18%	2%	23%	9%	48%
Recycled plastic from CR	0%	18%	2%	23%	9%	48%
Automotive						
PP	4%	11%	10%	36%	8%	31%
PUR	4%	11%	10%	36%	8%	31%
PA	4%	11%	10%	36%	8%	31%
Recycled plastic from CR	4%	11%	10%	36%	8%	31%

Electronic						
PP	4%	11%	10%	36%	8%	31%
PS	4%	11%	10%	36%	8%	31%
ABS	4%	11%	10%	36%	8%	31%
Recycled plastic from CR	4%	11%	10%	36%	8%	31%
Building and construction						
PVC	0%	0%	0%	100%	0%	0%
HDPE	0%	25%	0%	25%	0%	50%
PS	0%	0%	0%	0%	0%	100%
Recycled plastic from CR	0%	8%	0%	42%	0%	50%
Agriculture						
LDPE	35%	25%	0%	20%	0%	20%
HDPE	0%	0%	0%	80%	0%	20%
PP	100%	0%	0%	0%	0%	0%
Recycled plastic from CR	45%	8%	0%	33%	0%	13%

To quantify the potential recycled content (in %) in 2030, the potential plastic demand in 2030 (per sector) is forecasted using linear regression based on the historical data for the period 2014 – 2020 from Plastics Europe reports (Plastics Europe, 2015; 2016; 2017; 2018; 2019b; 2020). The results of linear regression from historical data for the period 2014 – 2020 can be found in Figure A.28 and projected quantities in Table A.13 below. From Figure A.28, the 'slope' and 'intercept' are calculated and used to quantify the projected quantities in 2030.

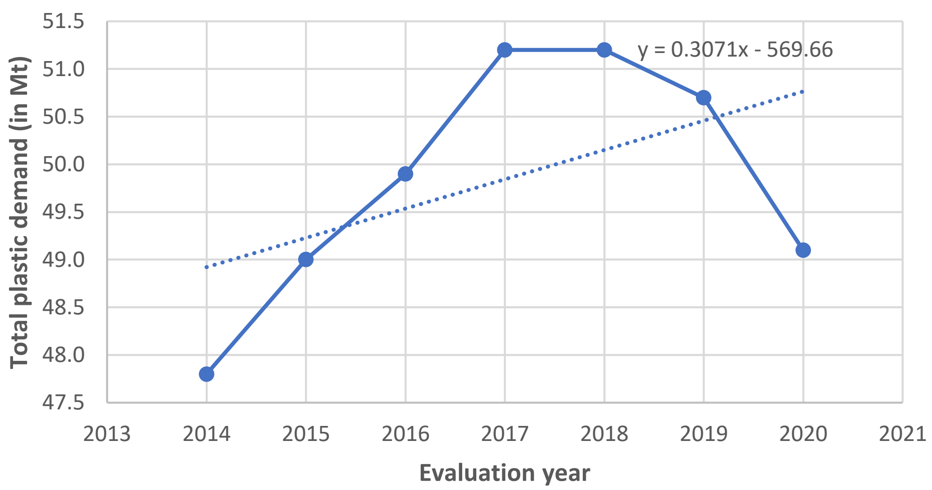

Figure A.28 Linear regression of plastic waste demand in 2014 – 2020 to quantify the projected plastic demand in 2030.

Table A.13 Projected quantities of plastic demand (in Mt) in 2030.

Quantities by Sector (in Mt)						Total Demand (in Mt)
Packaging	*Construction*	*Automotive*	*Electronic*	*Agriculture*	*Others*	
22.1	11.4	5.5	3.8	1.7	9.3	53.8

SECTION 9: MFA RESULTS OF DIFFERENT POLYMERS AT PROCESS LEVEL ACROSS DIFFERENT SECTORS IN 2018 AND 2030

Table A.14 shows the mass quantity (in kt, per sector) of each modeling output (i.e., recycled plastic as polymer, base chemicals, and fuels) for all considered scenarios in 2018 and 2030 (S0 – S5 in the main text). The modeling results are broken down into different outputs from different sectors (Table A.14 and Table A.15).

Table A.14 The summary of mass outputs per sector (in kt) for all scenarios (S0 – S5), including the standard deviation (in %). Acronyms: CR (chemical recycling), EoL-RR (end-of-life recycling rate), MR (mechanical recycling), P2C (plastic-to-chemicals), P2F (plastic-to-fuel), P2P (plastic-to-plastic), and solvent-based recycling (SBR).

	S0	*S1*	*S2*	*S3*	*S4*	*S5*
Packaging						
Waste Input	14,240	17,371±7%	17,371±7%	17,371±7%	17,371±7%	19,371±7%
P2P	2,456±10%	8,528±6%	6,602±7%	10,182±7%	10,568±7%	11,649±7%
P2P from MR	2,456±10%	8,528±6%	-	7,167±7%	7,951±7%	8,693±7%
P2P from CR and SBR	-	-	6,602±7%	3,015±6%	2,617±6%	2,956±6%
P2C	-	-	6,133±6%	3,708±6%	3,447±6%	3,890±6%
P2F	-	-	1,009±4%	594±4%	550±4%	620±4%
Automotive						
Waste Input	625	743±23%	743±23%	743±23%	743±23%	2,743±23%
P2P	65±12%	285±7%	267±7%	384±7%	399±7%	1,444±7%
P2P from MR	65±12%	285±7%	-	193±7%	215±7%	732±7%
P2P from CR and SBR	-	-	267±7%	191±6%	184±6%	712±6%
P2C	-	-	223±6%	140±6%	131±6%	487±6%
P2F	-	-	36±4%	22±4%	20±4%	76±4%
Electronic						
Waste Input	780	901±1%	901±1%	901±1%	901±1%	2,901±23%
P2P	132±10%	377±8%	194±7%	427±7%	455±7%	1,462±7%
P2P from MR	132±10%	377±8%	-	338±7%	378±7%	1,214±7%
P2P from CR and SBR	-	-	194±7%	89±6%	77±6%	248±6%

P2C	-	-	285±6%	112±6%	93±6%	299±6%
P2F	-	-	43±4%	18±4%	15±4%	47±4%
Building and Construction						
Waste Input	1,009	1,101±7%	1,101±7%	1,101±7%	1,101±7%	3,100±23%
P2P	306±14%	619±9%	647±7%	746±7%	756±7%	2,107±7%
P2P from MR	306±14%	619±9%	-	556±7%	619±7%	1,721±7%
P2P from CR and SBR	-	-	647±7%	190±6%	137±6%	386±6%
P2C	-	-	226±6%	172±6%	166±6%	465±6%
P2F	-	-	36±4%	27±4%	26±4%	73±6%
[1]*EoL-RR*						
Agriculture						
Waste Input	711	815±4%	815±4%	815±4%	815±4%	2,815±23%
P2P	314±13%	468±9%	196±7%	523±7%	561±7%	1,875±7%
P2P from MR	314±13%	468±9%	-	418±7%	468±7%	1,530±7%
P2P from CR and SBR	-	-	196±7%	105±6%	94±6%	345±6%
P2C	-	-	382±6%	142±6%	114±6%	465±6%
P2F	-	-	65±4%	23±4%	18±4%	73±6%
[1]**Overall**						
Waste Input	17,367	20,931±10%	20,931±10%	20,931±10%	20,931±10%	30,931±10%
P2P	3,273±9%	10,277±5%	7,905±6%	12,262±7%	12,740±7%	18,536±7%
P2P from MR	3,273±9%	10,277±5%	-	8,672±7%	9,630±7%	13,890±7%
P2P from CR and SBR	-	-	7,905±6%	3,590±6%	3,110±6%	4,646±6%
P2C	-	-	7,247±6%	4,272±6%	3,951±6%	5,556±6%
P2F	-	-	1,189±4%	683±4%	628±4%	881±4%

[1] *The 'overall' data points quantify the sum of mass quantities (in kt) from all sectors and aggregated calculations of the circularity indicators*

Table A. 15 Summary of the circularity indicators for all scenarios in 2018 (S0) and 2030 (S1–S5), per sector (e.g., packaging, automotive, etc.). Values are rounded in %, including the standard deviation (in %). Acronyms: CR (chemical recycling), EoL-RR (end-of-life recycling rate), MR (mechanical recycling), P2C (plastic-to-chemical), P2F (plastic-to-fuel), P2P (plastic-to-plastic), and solvent-based recycling (SBR).

	S0	*S1*	*S2*	*S3*	*S4*	*S5*
Packaging						
P2P	17%±2%	49%±3%	38%±3%	59%±3%	61%±3%	61%±3%
P2P from MR	*17%±2%*	*49%±3%*	-	*41%±3%*	*46%±3%*	*46%±3%*
P2P from CR and SBR	-	-	*38%±3%*	*17%±1%*	*15%±1%*	*15%±1%*
P2C	-	-	35%±2%	21%±1%	20%±1%	20%±1%
P2F	-	-	6%±0%	3%±0%	3%±0%	3%±0%
[1]EoL-RR	17%±2%	49%±3%	73%±4%	80%±3%	81%±3%	81%±3%
Automotive						
P2P	10%±1%	38%±3%	36%±3%	52%±3%	54%±3%	54%±3%
P2P from MR	*10%±1%*	*38%±3%*	-	*26%±1%*	*29%±2%*	*29%±2%*
P2P from CR and SBR	-	-	*36%±3%*	*26%±1%*	*25%±1%*	*25%±1%*
P2C	-	-	30%±2%	19%±1%	18%±1%	18%±1%
P2F	-	-	5%±0%	3%±0%	3%±0%	3%±0%
[1]EoL-RR	10%±1%	38%±3%	66%±4%	71%±4%	72%±4%	72%±4%
Electronic						
P2P	17%±2%	42%±3%	22%±2%	48%±3%	50%±4%	50%±4%
P2P from MR	*17%±2%*	*42%±3%*	-	*38%±1%*	*42%±3%*	*42%±3%*
P2P from CR and SBR	-	-	*22%±2%*	*10%±1%*	*8%±0%*	*8%±0%*
P2C	-	-	32%±2%	12%±1%	10%±1%	10%±1%
P2F	-	-	5%±0%	2%±0%	2%±0%	2%±0%
[1]EoL-RR	17%±2%	42%±3%	54%±4%	59%±4%	60%±3%	60%±3%
Building and construction						
P2P	30%±2%	56%±5%	59%±4%	68%±4%	69%±4%	69%±4%
P2P from MR	*30%±2%*	*56%±5%*	-	*51%±3%*	*56%±3%*	*56%±3%*
P2P from CR and SBR	-	-	*59%±4%*	*17%±1%*	*12%±1%*	*12%±1%*
P2C	-	-	21%±1%	16%±1%	15%±1%	15%±1%
P2F	-	-	3%±0%	2%±0%	2%±0%	2%±0%

[1]EoL-RR	30%±2%	56%±5%	80%±5%	84%±5%	84%±5%	84%±5%

Agriculture						
P2P	44%±5%	57%±5%	24%±2%	64%±4%	69%±4%	69%±4%
P2P from MR	*44%±5%*	*57%±5%*	-	*51%±3%*	*57%±3%*	*57%±3%*
P2P from CR and SBR	-	-	*24%±2%*	*13%±1%*	*12%±1%*	*12%±1%*
P2C	-	-	47%±3%	17%±1%	14%±1%	14%±1%
P2F	-	-	8%±0%	3%±0%	2%±0%	2%±0%
[1]EoL-RR	44%±5%	56%±5%	71%±3%	81%±5%	83%±5%	83%±5%

[1]EoL-RR considers only P2P and P2C because P2F recycling does not conform to the definition of ‘recycling’ in WFD (European Commission, 2018a; 2008)

Figure A.29 – A.33 show the Sankey diagrams (MFA results) of plastic waste treatment in five different sectors in **S0**

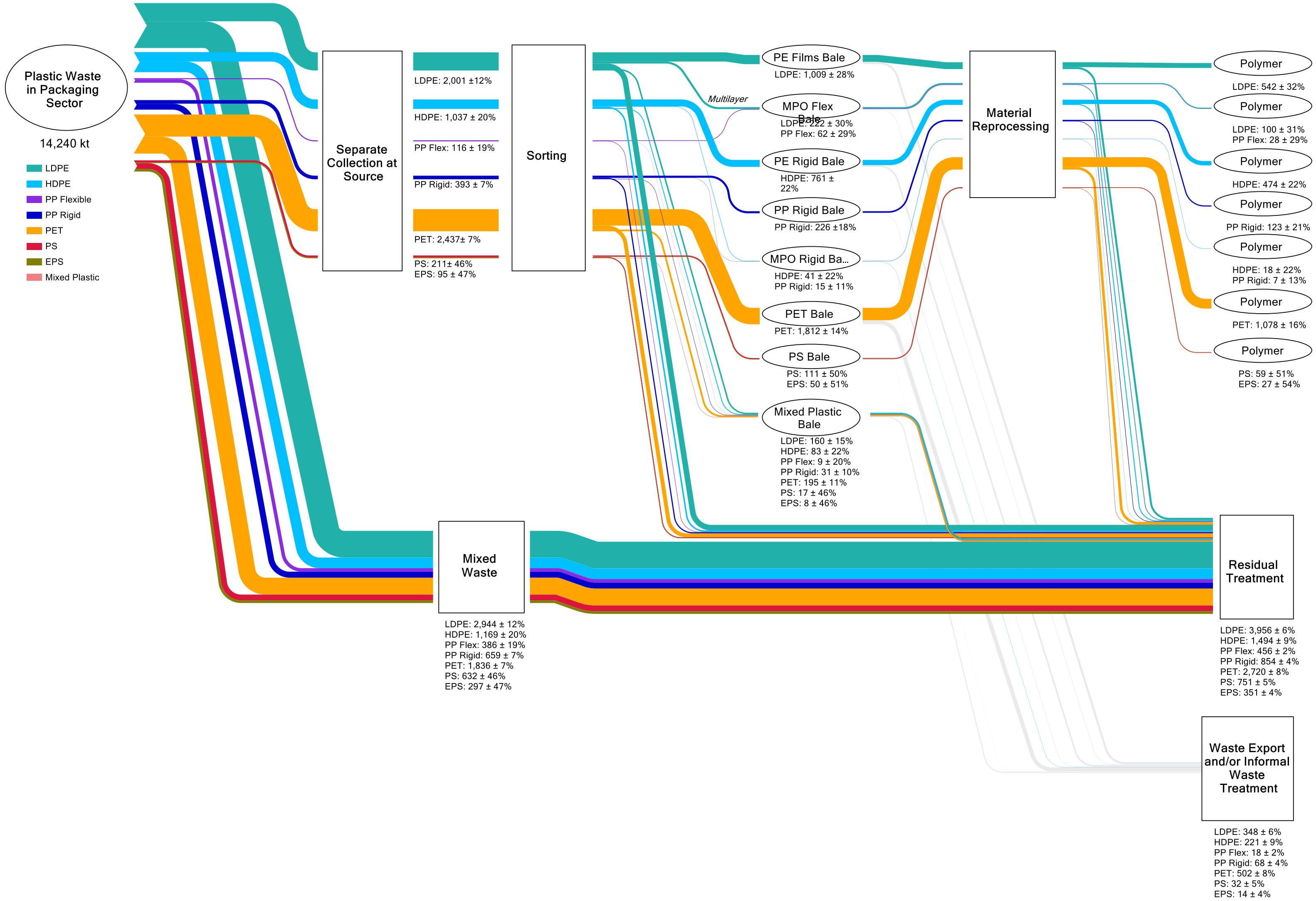

Figure A.29 MFA results of plastic waste treatment in the packaging sector in 2018 (S0).

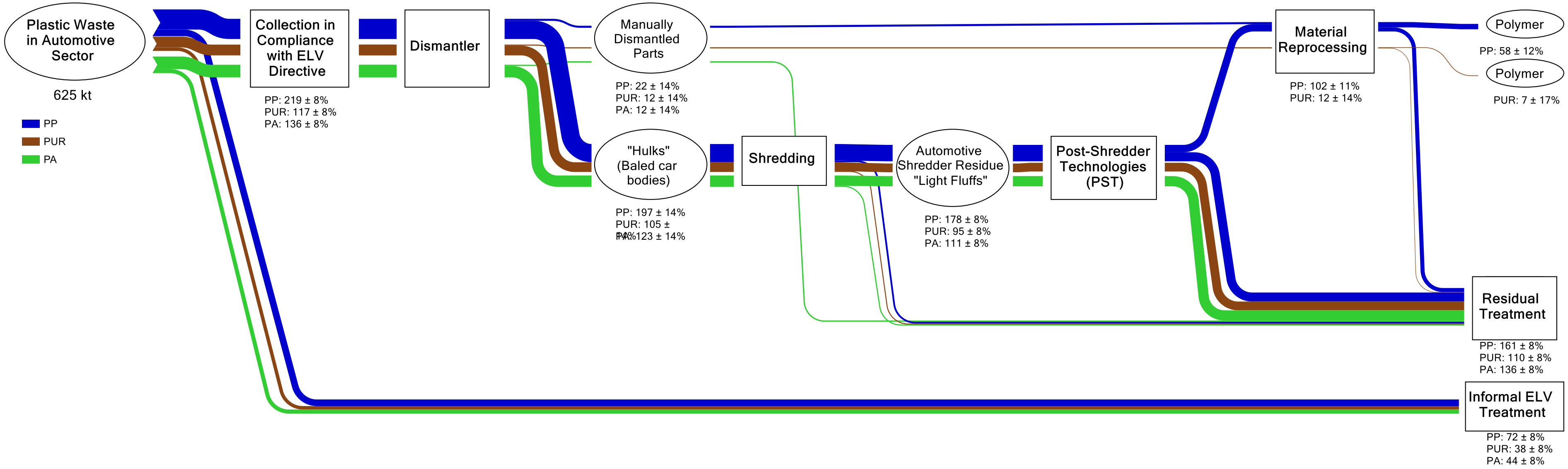

Figure A.30 MFA results of plastic waste treatment in the automotive sector in 2018 (S0).

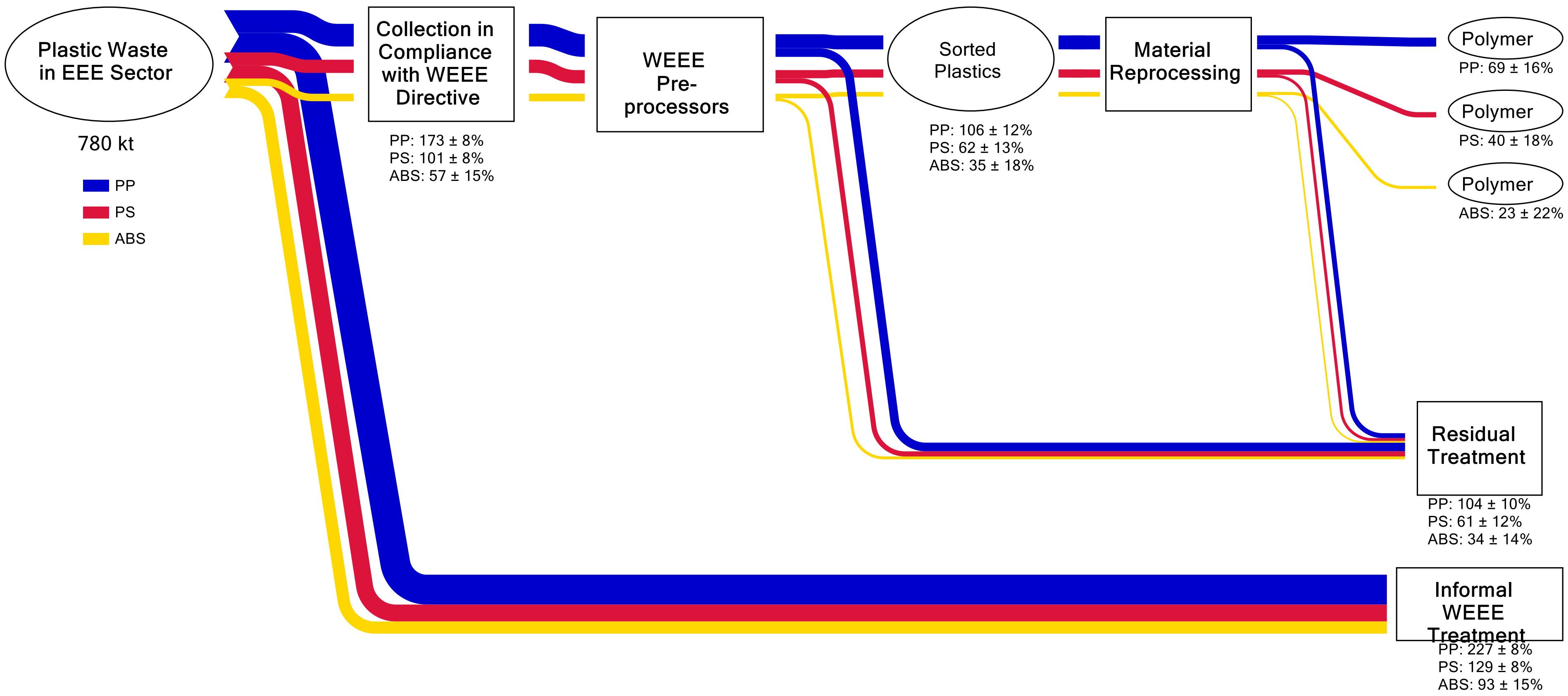

Figure A.31 MFA results of plastic waste treatment in the electronic sector in 2018 (S0).

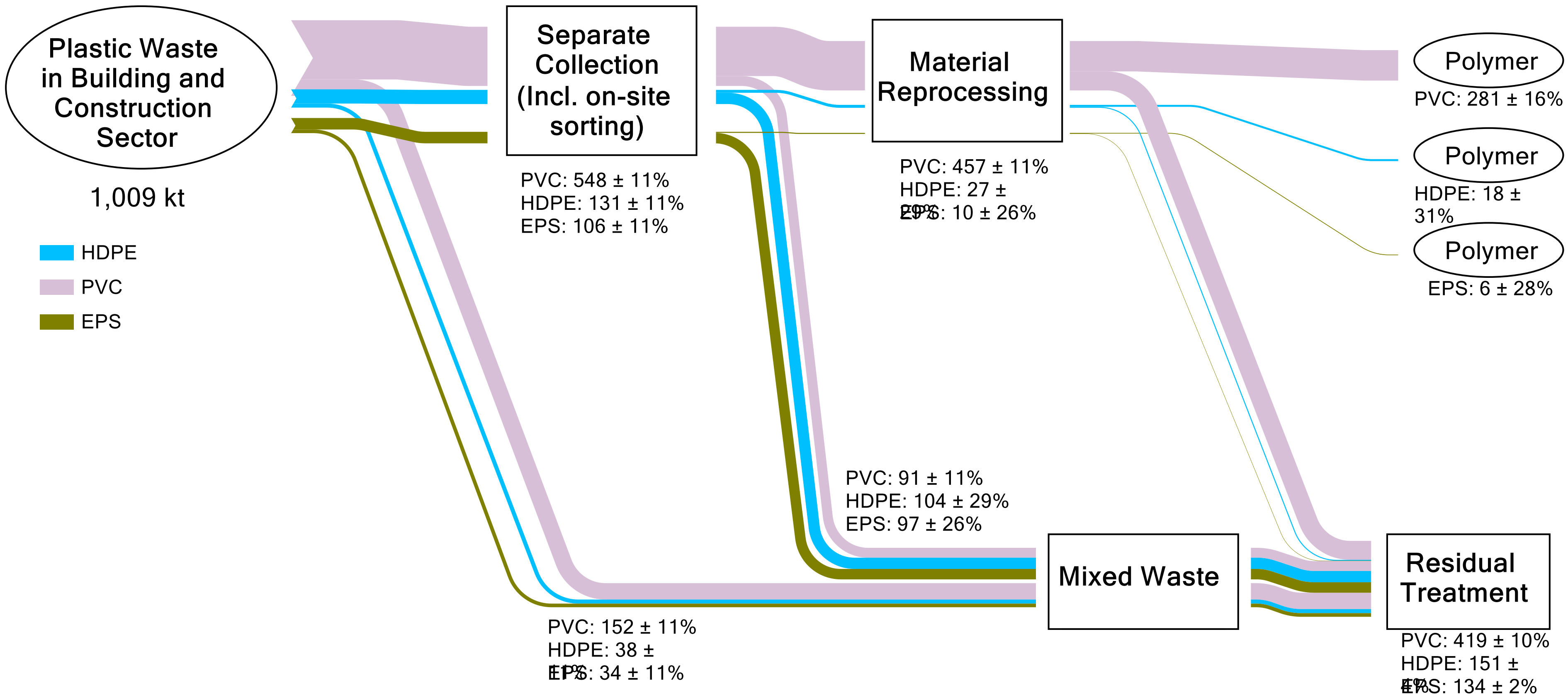

Figure A. 32 MFA results of plastic waste treatment in the building and construction sector in 2018 (S0).

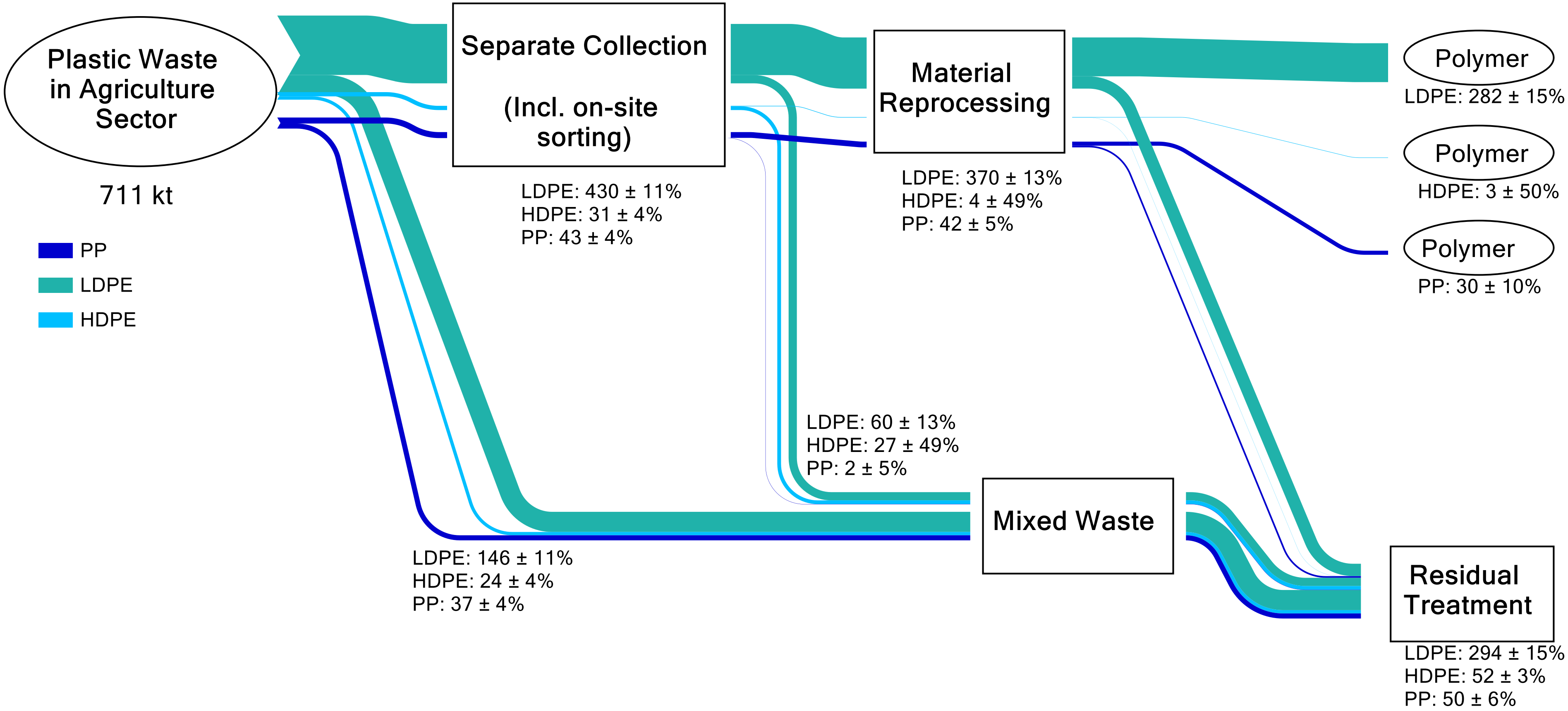

Figure A. 33 MFA results of plastic waste treatment in the agriculture sector in 2018 (S0).

Figure A.34 – A.38 show the Sankey diagrams (MFA results) of plastic waste treatment in five different sectors in **S1**

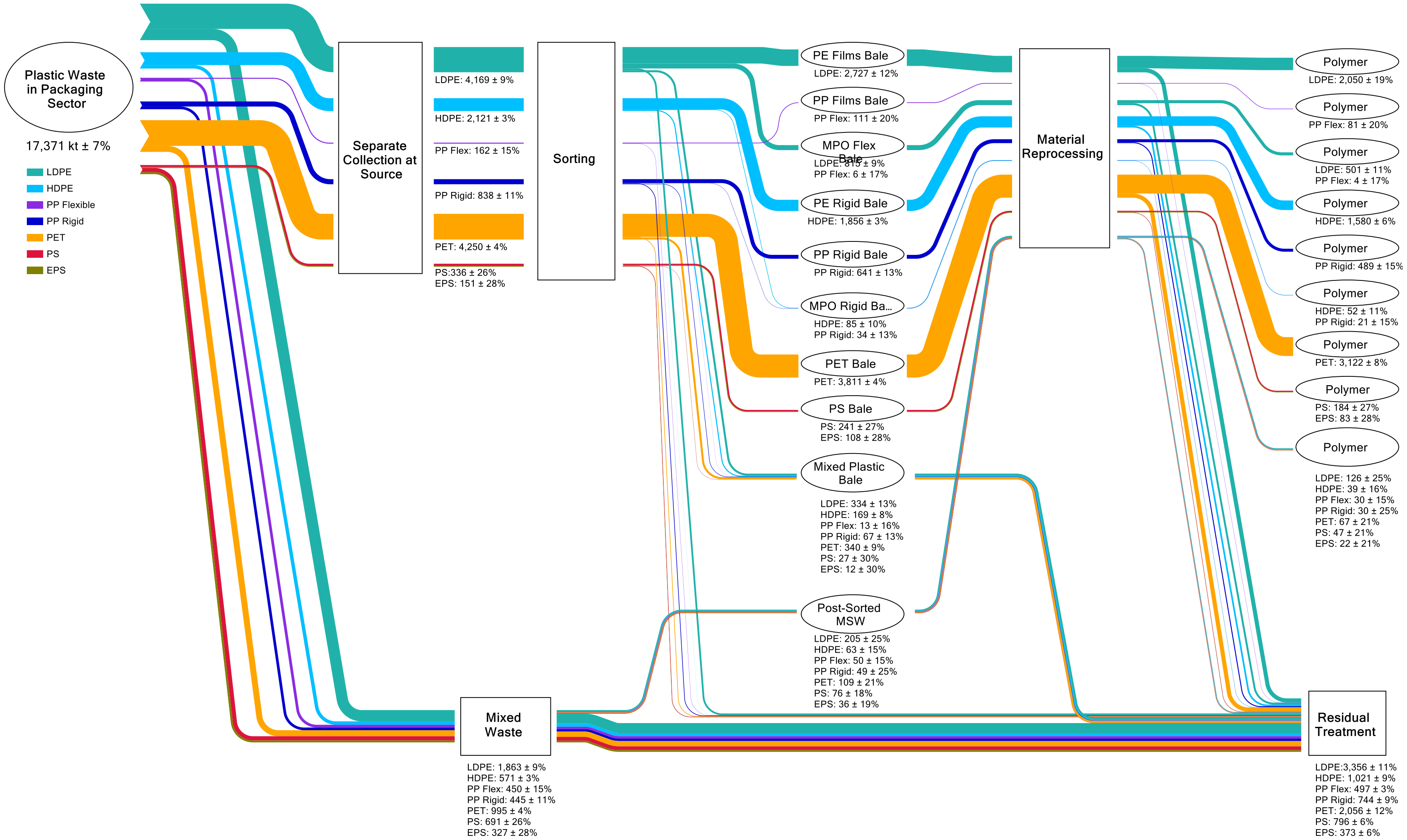

Figure A.34 MFA results of plastic waste treatment in the packaging sector in 2030, S1.

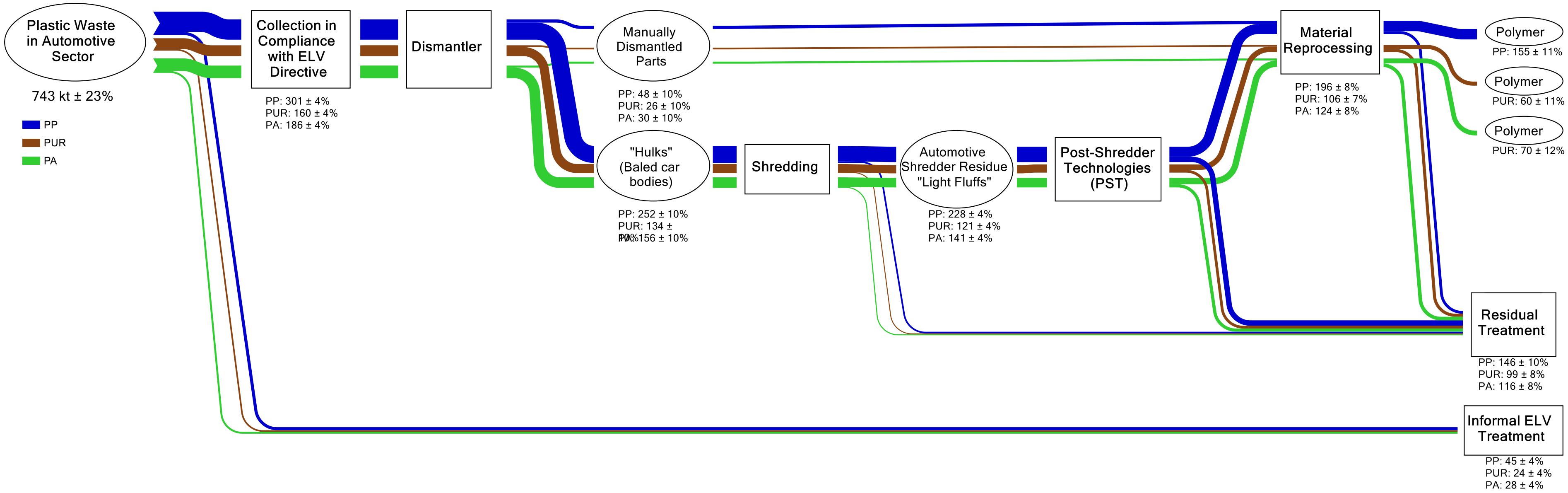

Figure A. 35 MFA results of plastic waste treatment in the automotive sector in 2030, S1.

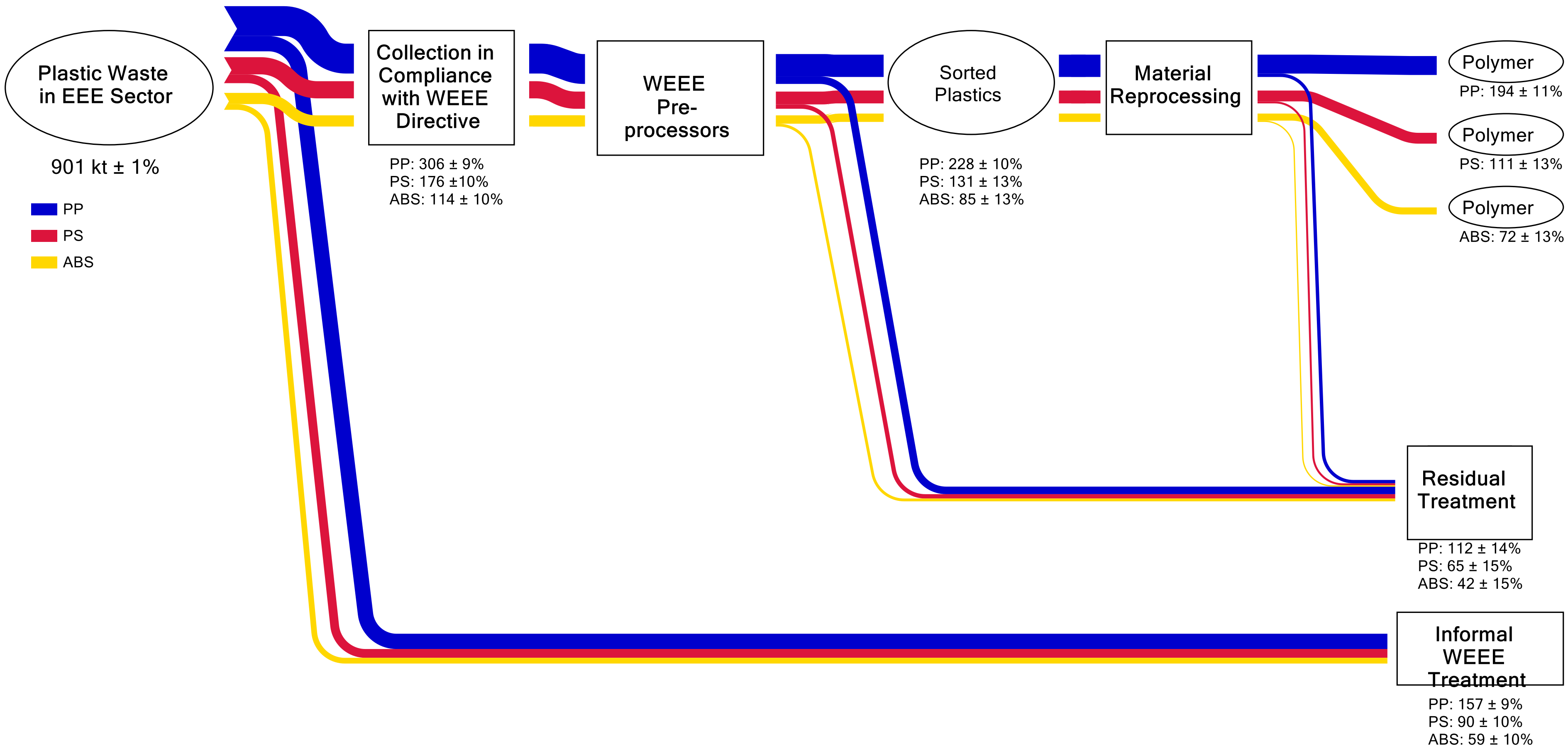

Figure A.36 MFA results of plastic waste treatment in the electronic sector in 2030, S1.

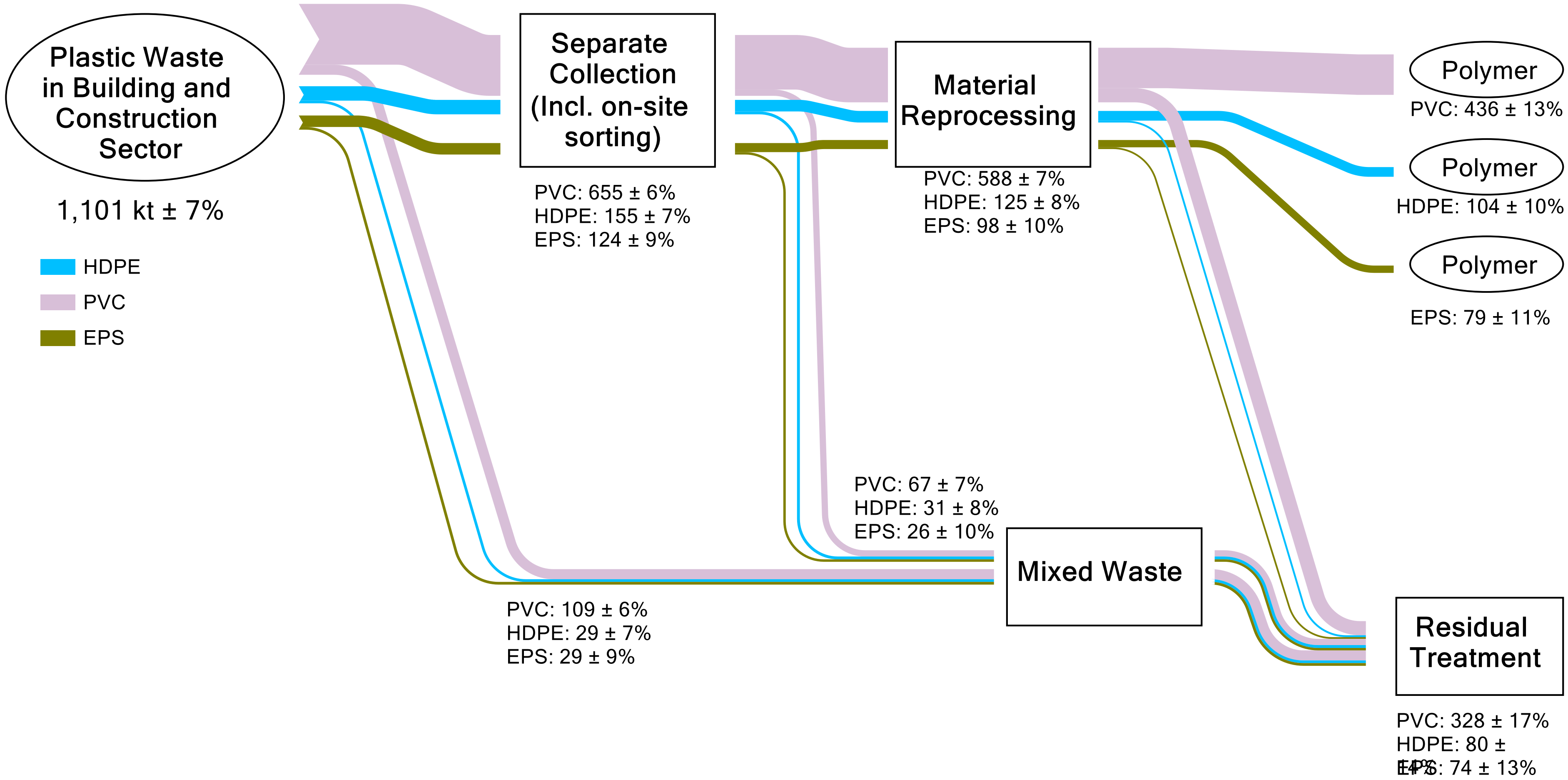

Figure A.37 MFA results of plastic waste treatment in the building and construction sector in 2030, S1.

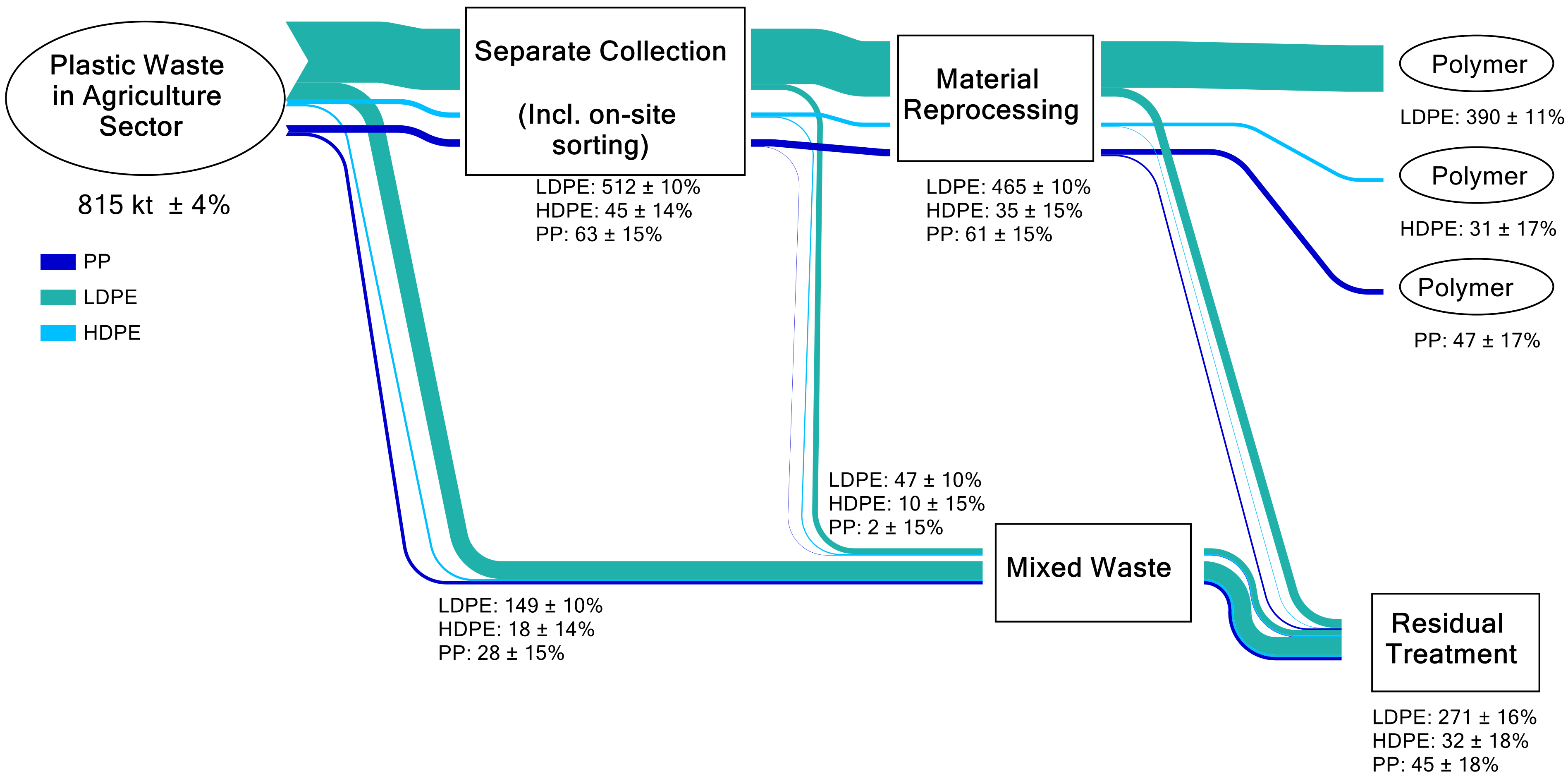

Figure A.38 MFA results of plastic waste treatment in the agriculture sector in 2030, S1.

Figure A.39 – A.43 show the Sankey diagrams (MFA results) of plastic waste treatment in five different sectors in **S2.**

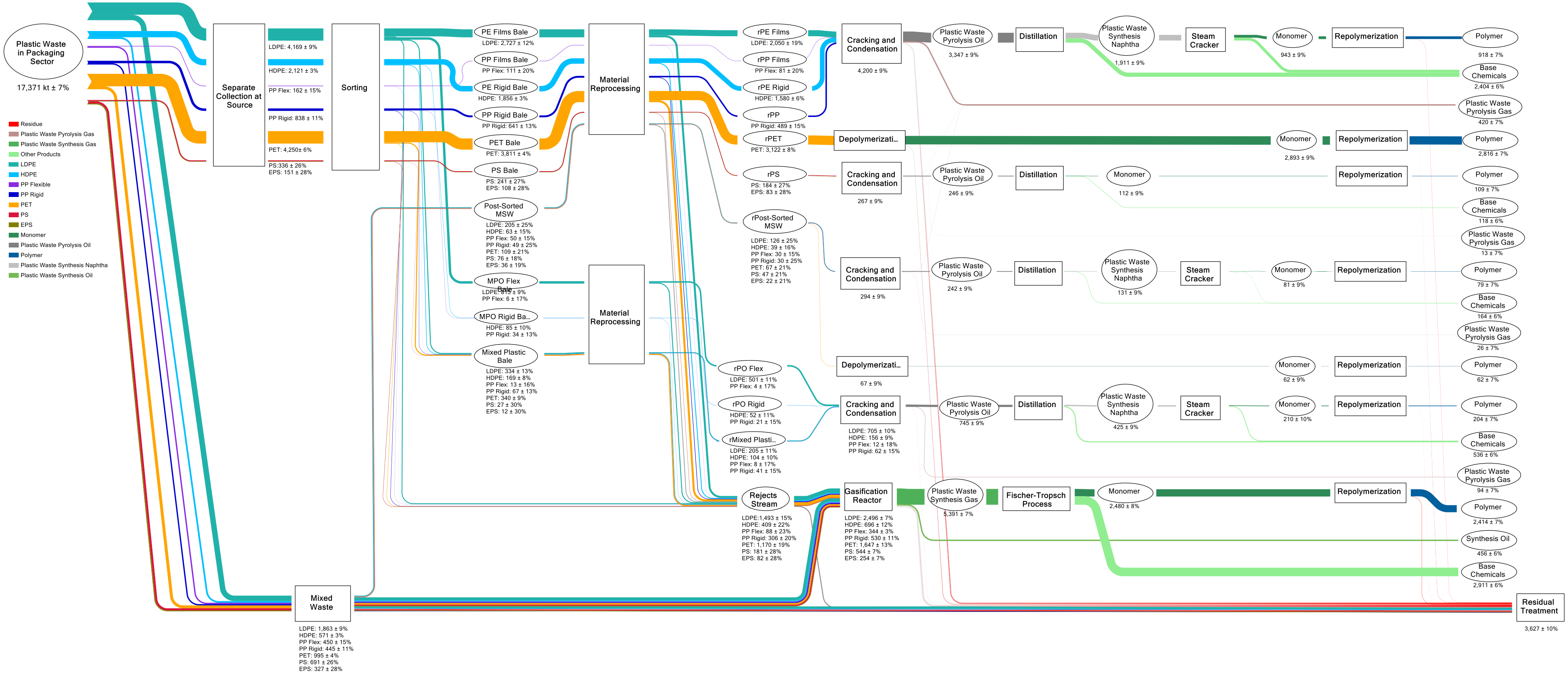

Figure A.39 MFA results of plastic waste treatment in the packaging sector in 2030, S2.

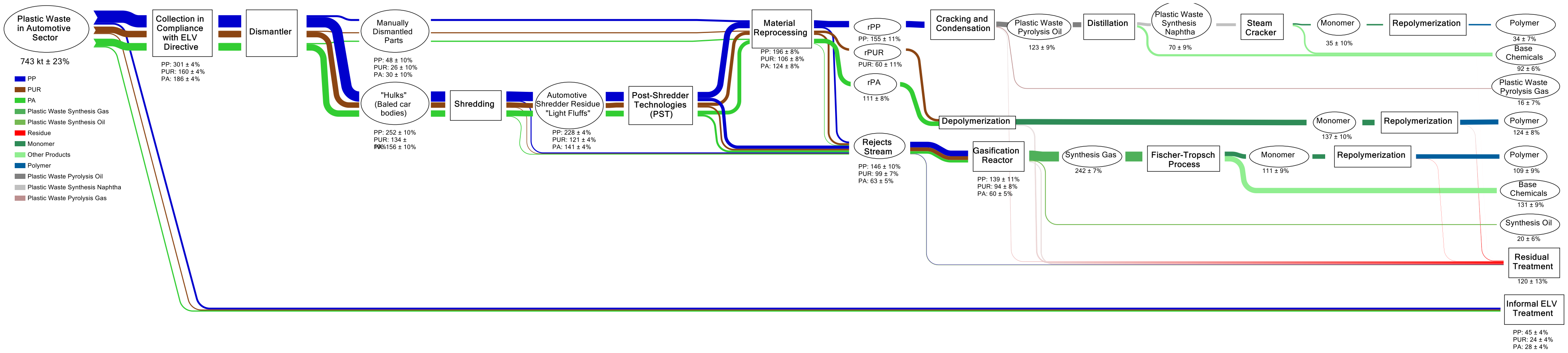

Figure A.40 MFA results of plastic waste treatment in the automotive sector in 2030, S2.

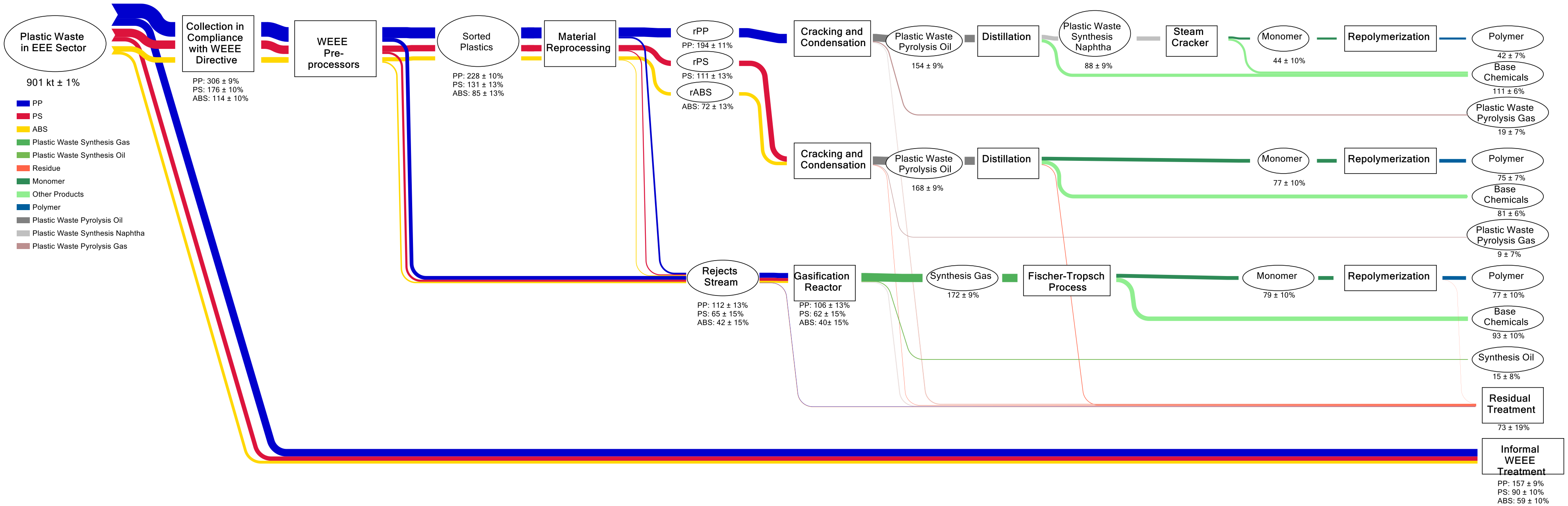

Figure A.41 MFA results of plastic waste treatment in the electronic sector in 2030, S2.

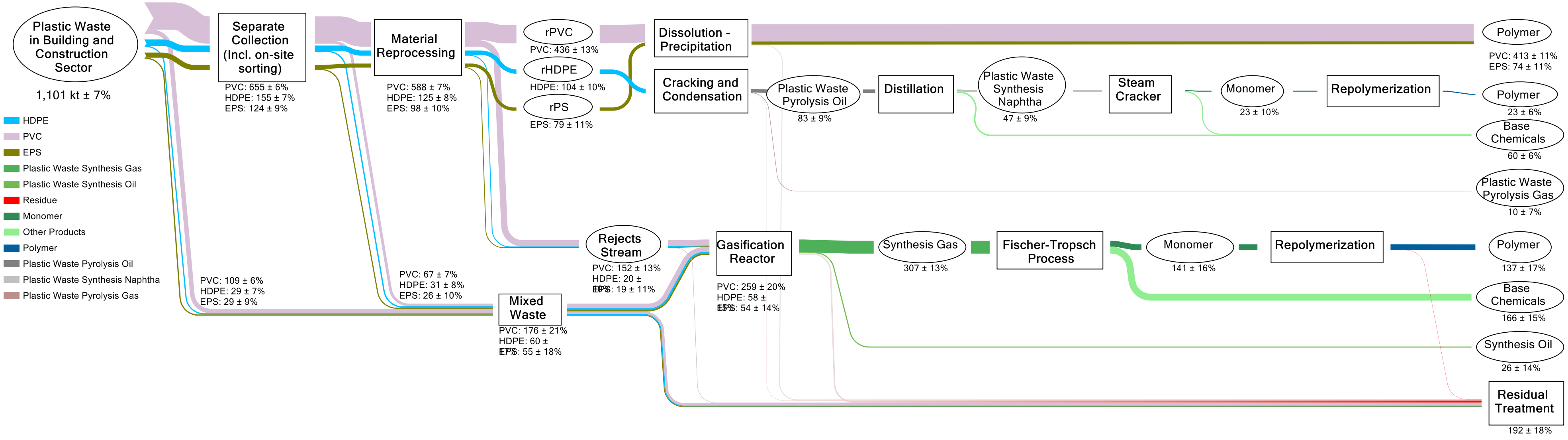

Figure A.42 MFA results of plastic waste treatment in the building and construction sector in 2030, S2.

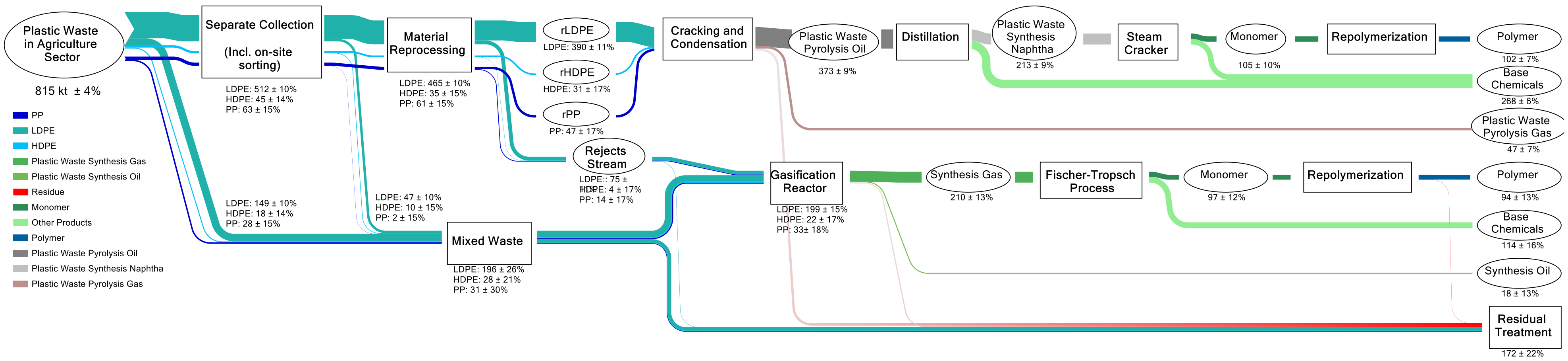

Figure A.43 MFA results of plastic waste treatment in the agriculture sector in 2030, S2.

Figure A.44 – A.48 show the Sankey diagrams (MFA results) of plastic waste treatment in five different sectors in **S3.**

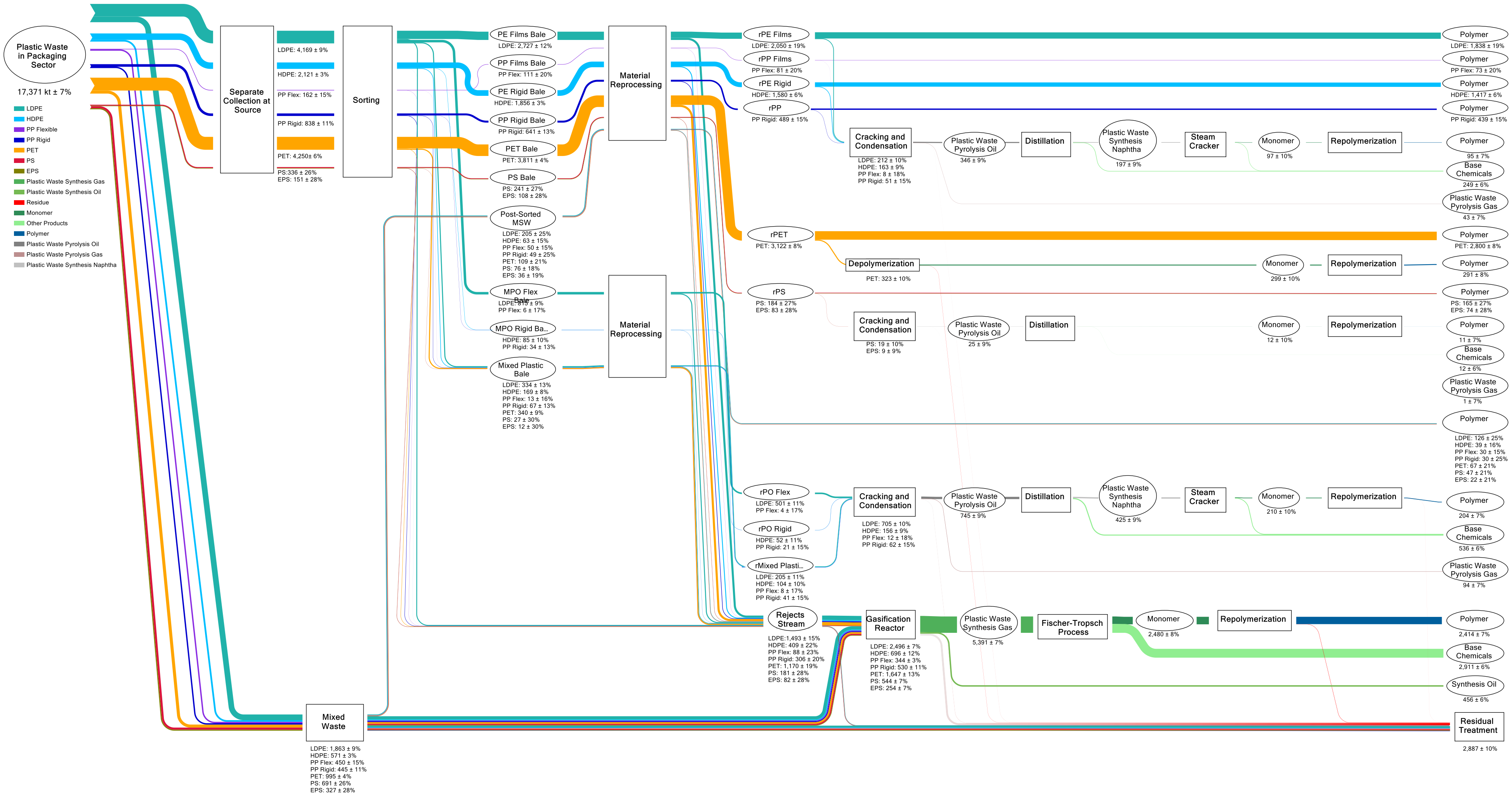

Figure A.44 MFA results of plastic waste treatment in the packaging sector in 2030, S3.

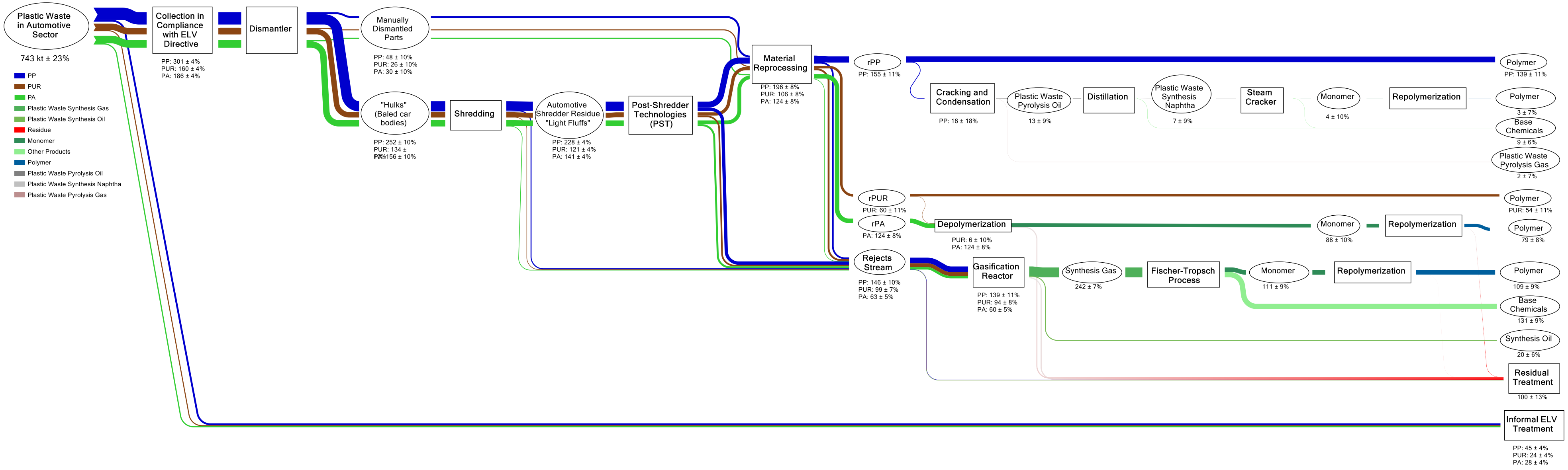

Figure A.45 MFA results of plastic waste treatment in the automotive sector in 2030, S3.

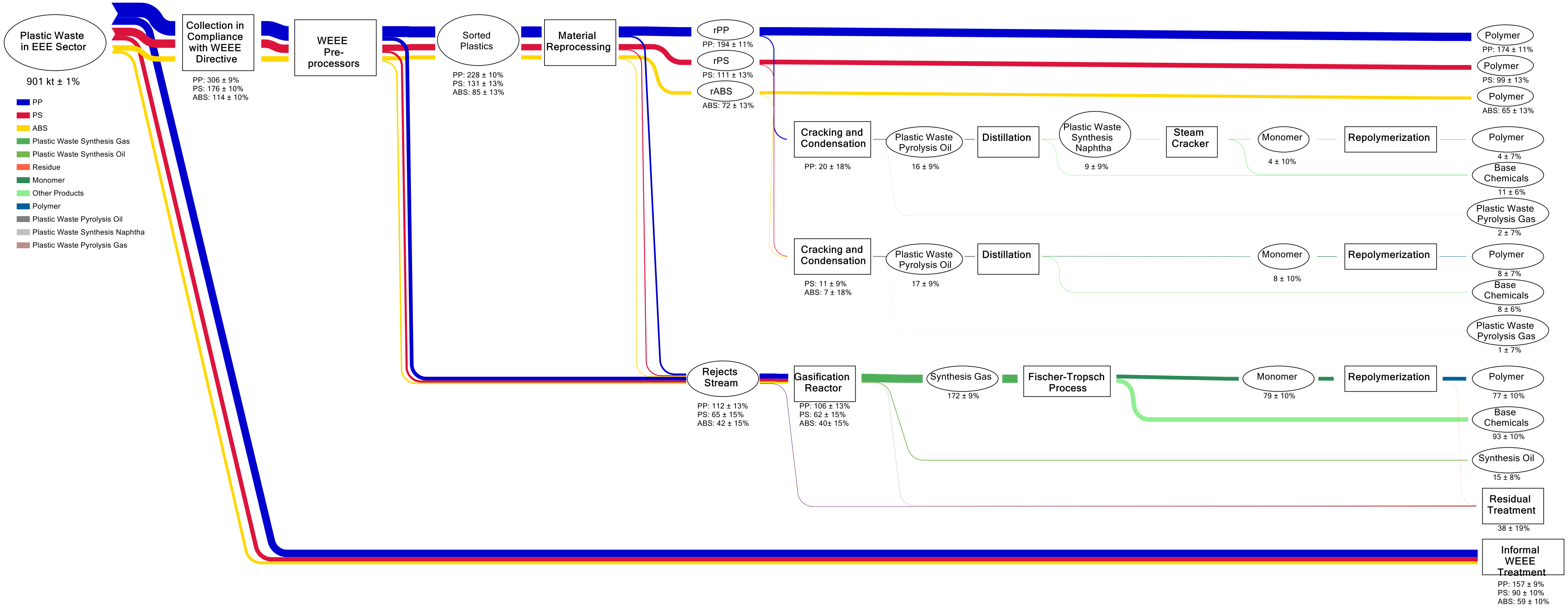

Figure A.46 MFA results of plastic waste treatment in the electronic sector in 2030, S3.

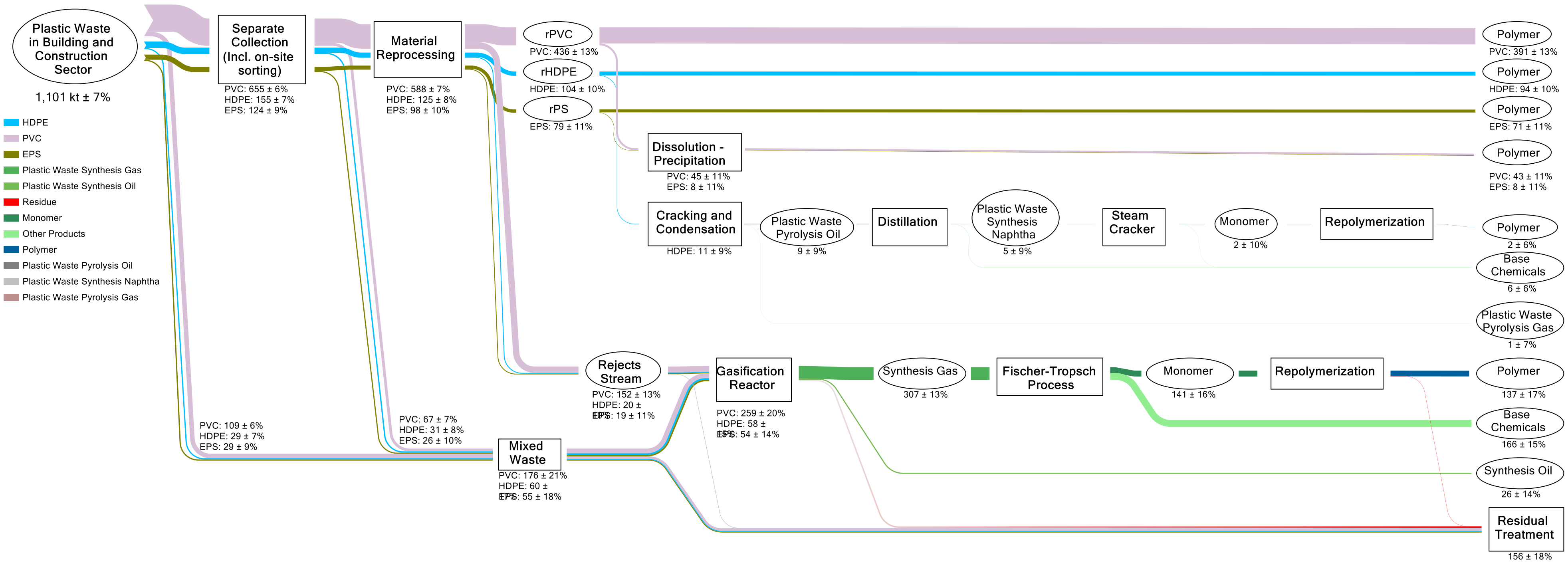

Figure A.47 MFA results of plastic waste treatment in the building and construction sector in 2030, S3.

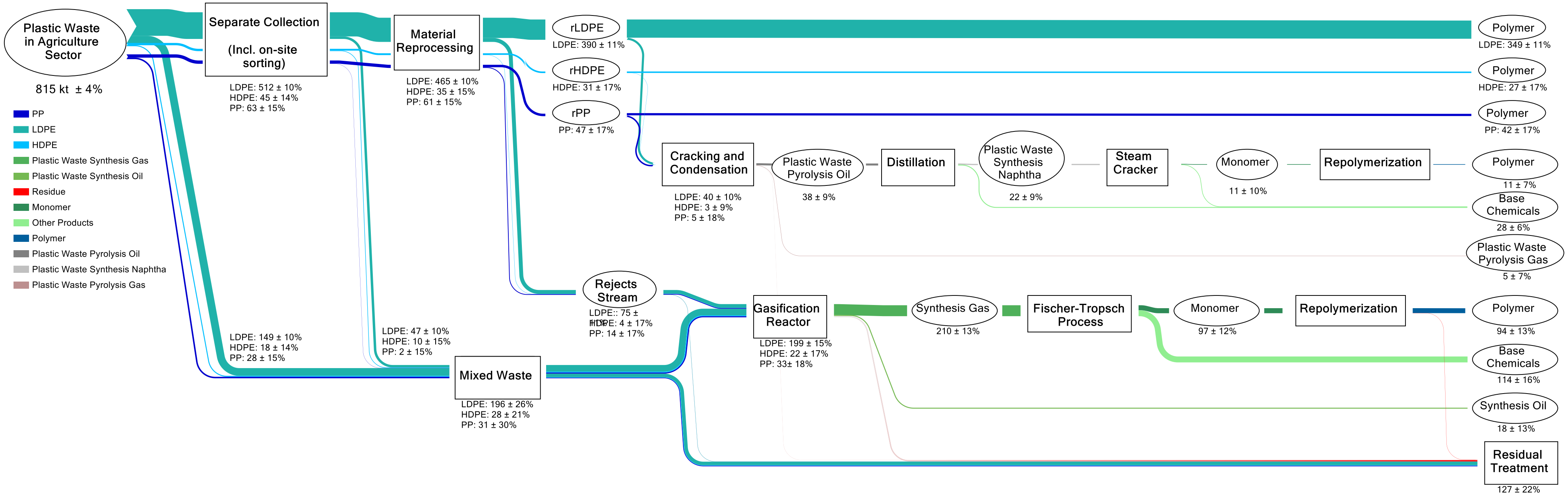

Figure A.48 MFA results of plastic waste treatment in the agriculture sector in 2030, S3.

Figure A.49 – A.53 show the Sankey diagrams (MFA results) of plastic waste treatment in five different sectors in **S4**

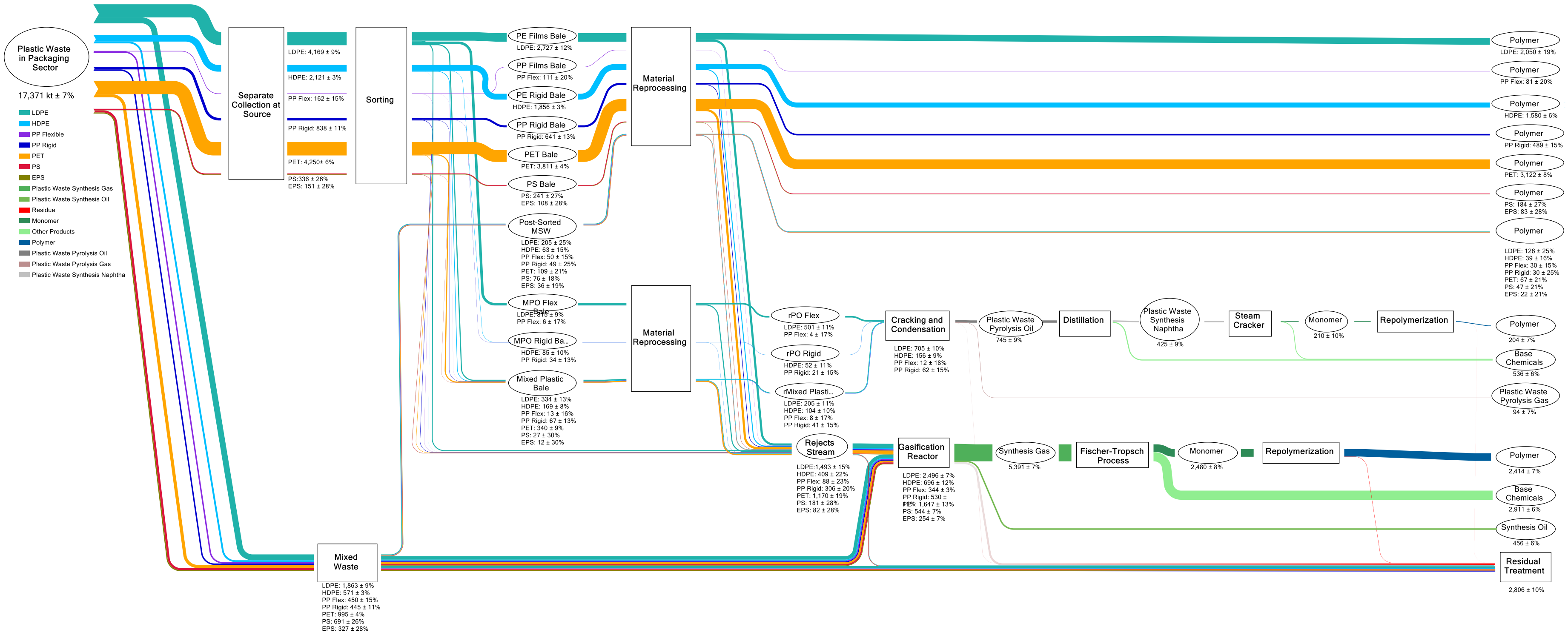

Figure A.49 MFA results of plastic waste treatment in the packaging sector in 2030, S4.

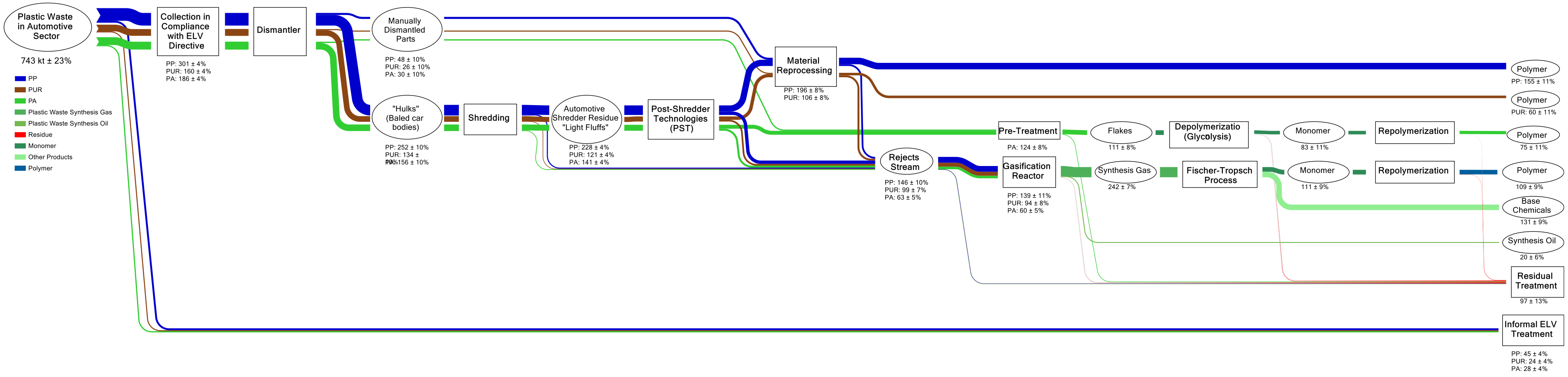

Figure A. 50 MFA results of plastic waste treatment in the automotive sector in 2030, S4.

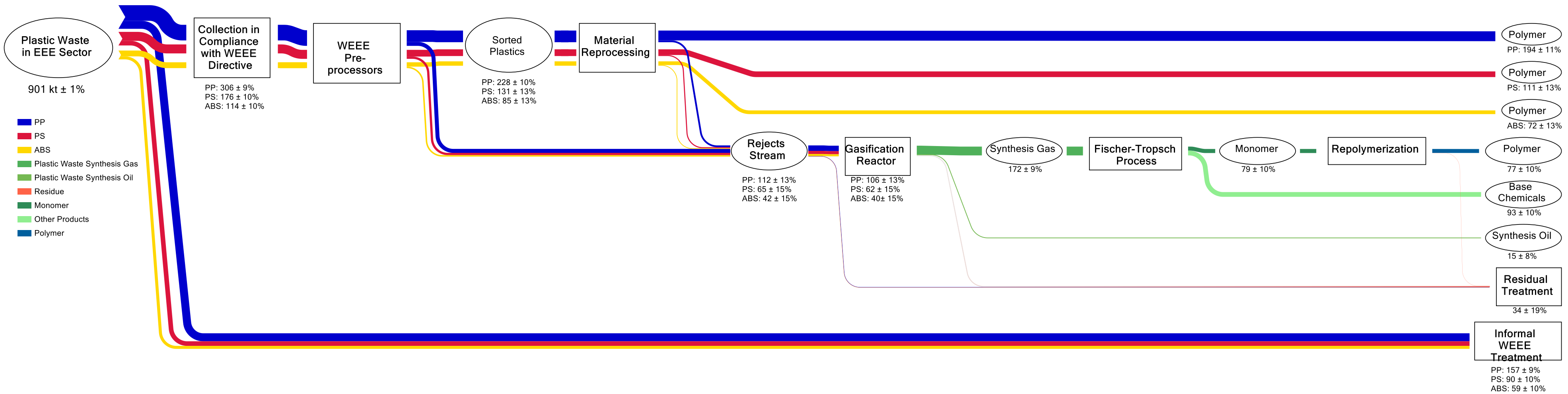

Figure A.51 MFA results of plastic waste treatment in the electronic sector in 2030, S4.

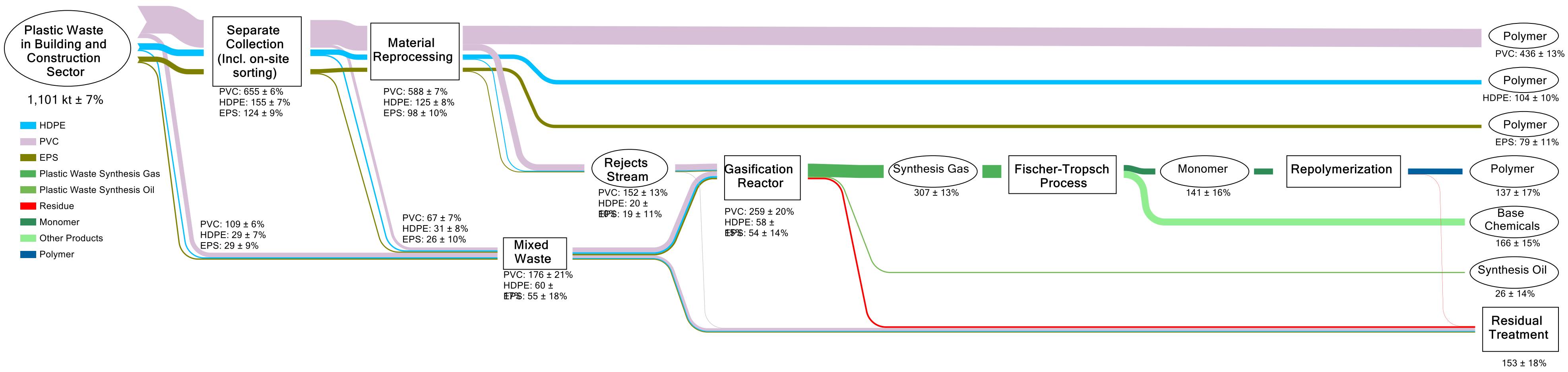

Figure A.52 MFA results of plastic waste treatment in the building and construction sector in 2030, S4.

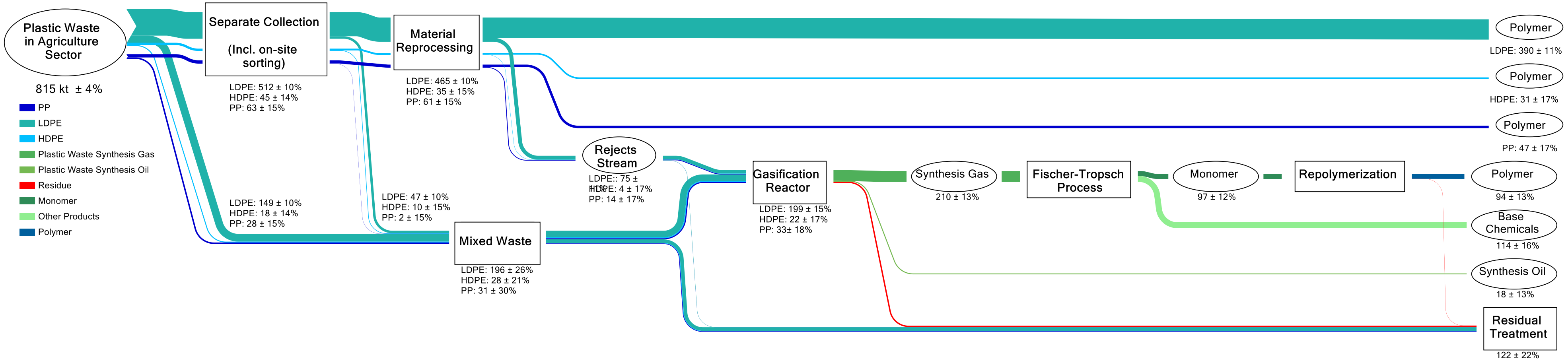

Figure A.53 MFA results of plastic waste treatment in the agriculture sector in 2030, S4.

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B

APPENDIX B. MODELLING CURRENT AND FUTURE FLOWS OF PLASTIC FROM WEEE RECYCLING

SECTION 1: SALES OF VACUUM CLEANERS, COFFEE MACHINES, AND ELECTRIC SHAVERS FROM 1980 – 2018, AND PROJECTED SALES FROM 2019 – 2030

Product sales' data points from 1980 – 2018 were extracted from European Commission Report, Eurostat website, and national WEEE report in Belgium and The Netherlands (Eurostat, 2022b; European Commission, 2019b; National WEEE Register, 2019; Recupel, 2018; Recupel, 2013; Huisman et al., 2012). The sales of selected electronic products are expected to grow 1.6% annually from 2019 – 2030 as suggested by Baldé et al. (2017) for countries with the highest purchasing power parity (PPP) like Belgium and The Netherlands. On average, vacuum cleaners, coffee machines, and electric shavers account for 14%, 4%, and 3% of the overall small household appliances (SHA) sales in Belgium respectively. In The Netherlands, vacuum cleaners, coffee machines, and electric shaver account for 16%, 4%, and 4% of the small household appliances sales respectively. The data points are presented in Figure B.1 and Table B.1.

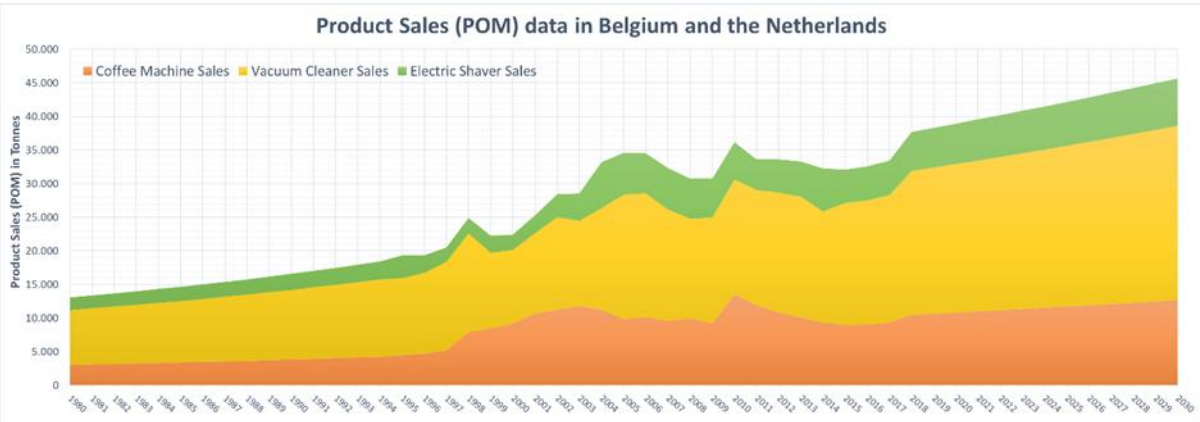


Figure B.1 Annual sales of the selected products from 1980 – 2030 in Belgium and the Netherlands.

Table B.1 Sales of the vacuum cleaner, coffee machines, and electric shavers from 1980 – 2018 and projected sales from 2019 – 2030 (in tonnes).

Year	*Belgium*			*The Netherlands*			*Total*
	Coffee Machine	*Vacuum Cleaner*	*Electric Shaver*	*Coffee Machine*	*Vacuum Cleaner*	*Electric Shaver*	
1980	1.921	2.883	646	1.138	5.254	1.235	13.077
1981	1.962	2.944	660	1.169	5.398	1.268	13.402
1982	2.001	3.002	673	1.199	5.534	1.300	13.708
1983	2.042	3.064	687	1.228	5.668	1.332	14.021
1984	2.083	3.125	701	1.258	5.807	1.364	14.337
1985	2.126	3.190	715	1.289	5.953	1.399	14.673
1986	2.170	3.256	730	1.323	6.108	1.435	15.022
1987	2.215	3.324	745	1.359	6.273	1.474	15.390
1988	2.263	3.396	762	1.395	6.442	1.514	15.772
1989	2.322	3.483	781	1.432	6.613	1.554	16.185
1990	2.374	3.562	799	1.472	6.795	1.597	16.597
1991	2.432	3.648	818	1.514	6.989	1.642	17.042
1992	2.490	3.736	838	1.556	7.185	1.688	17.493
1993	2.552	3.830	859	1.599	7.383	1.735	17.958
1994	2.613	3.921	879	1.641	7.580	1.781	18.415
1995	2.648	3.717	1.071	1.859	7.695	2.358	19.346
1996	2.677	4.150	795	2.077	7.808	1.834	19.342
1997	2.892	4.192	838	2.311	8.985	1.307	20.525
1998	3.444	5.345	878	4.457	9.328	1.448	24.901
1999	3.779	4.670	918	4.704	6.548	1.649	22.268
2000	4.117	4.276	885	5.034	6.721	1.392	22.425
2001	3.836	4.649	852	6.872	7.320	1.829	25.375
2002	4.345	5.304	1.162	6.909	8.487	2.224	28.430
2003	4.858	4.893	1.474	6.935	7.795	2.619	28.573
2004	4.278	5.826	1.290	6.953	9.306	5.538	33.190
2005	4.567	7.203	1.105	5.269	11.327	5.105	34.577
2006	4.479	8.605	1.305	5.667	9.791	4.667	34.513
2007	3.466	6.464	1.962	6.117	10.134	4.231	32.373
2008	4.116	4.294	1.753	5.871	10.496	4.231	30.761
2009	3.703	4.669	1.577	5.522	11.046	4.237	30.754
2010	3.258	5.237	1.991	10.314	11.819	3.557	36.176
2011	3.050	6.292	1.737	8.866	10.825	2.868	33.683
2012	3.233	7.359	1.718	7.641	10.447	3.217	33.614
2013	3.100	6.079	1.696	6.963	11.927	3.569	33.335
2014	3.068	4.306	2.202	6.271	12.220	4.203	32.269
2015	3.083	4.548	1.111	5.930	13.611	3.819	32.102
2016	2.798	4.129	1.008	6.255	14.356	4.028	32.574
2017	3.191	4.709	1.150	6.198	14.226	3.992	33.466

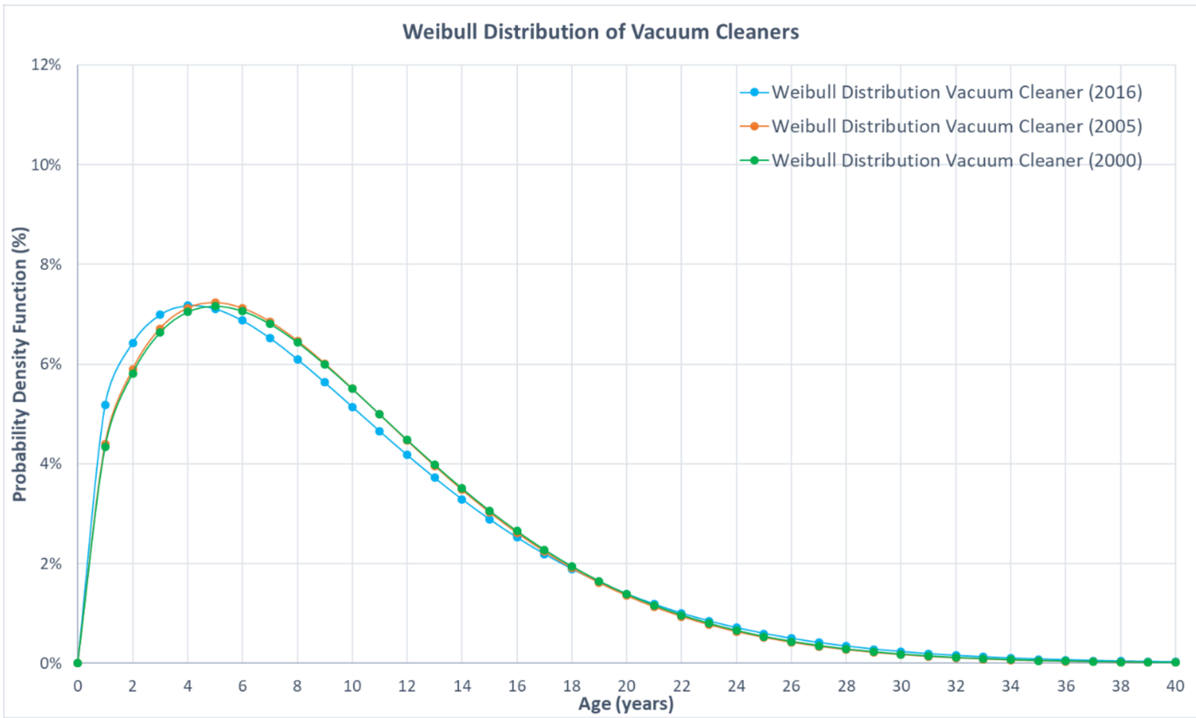
2018	3.242	4.784	1.168	7.247	16.633	4.667	37.741
2019	3.294	4.861	1.187	7.363	16.899	4.741	38.345
2020	3.347	4.938	1.206	7.480	17.170	4.817	38.958
2021	3.400	5.017	1.225	7.600	17.444	4.894	39.582
2022	3.455	5.098	1.245	7.722	17.723	4.973	40.215
2023	3.510	5.179	1.265	7.845	18.007	5.052	40.859
2024	3.566	5.262	1.285	7.971	18.295	5.133	41.512
2025	3.623	5.346	1.306	8.098	18.588	5.215	42.177
2026	3.681	5.432	1.326	8.228	18.885	5.299	42.851
2027	3.740	5.519	1.348	8.360	19.187	5.383	43.537
2028	3.800	5.607	1.369	8.493	19.494	5.470	44.234
2029	3.861	5.697	1.391	8.629	19.806	5.557	44.941
2030	3.923	5.788	1.413	8.767	20.123	5.646	45.660

SECTION 2: WEIBULL PARAMETER VALUES FROM LITERATURE

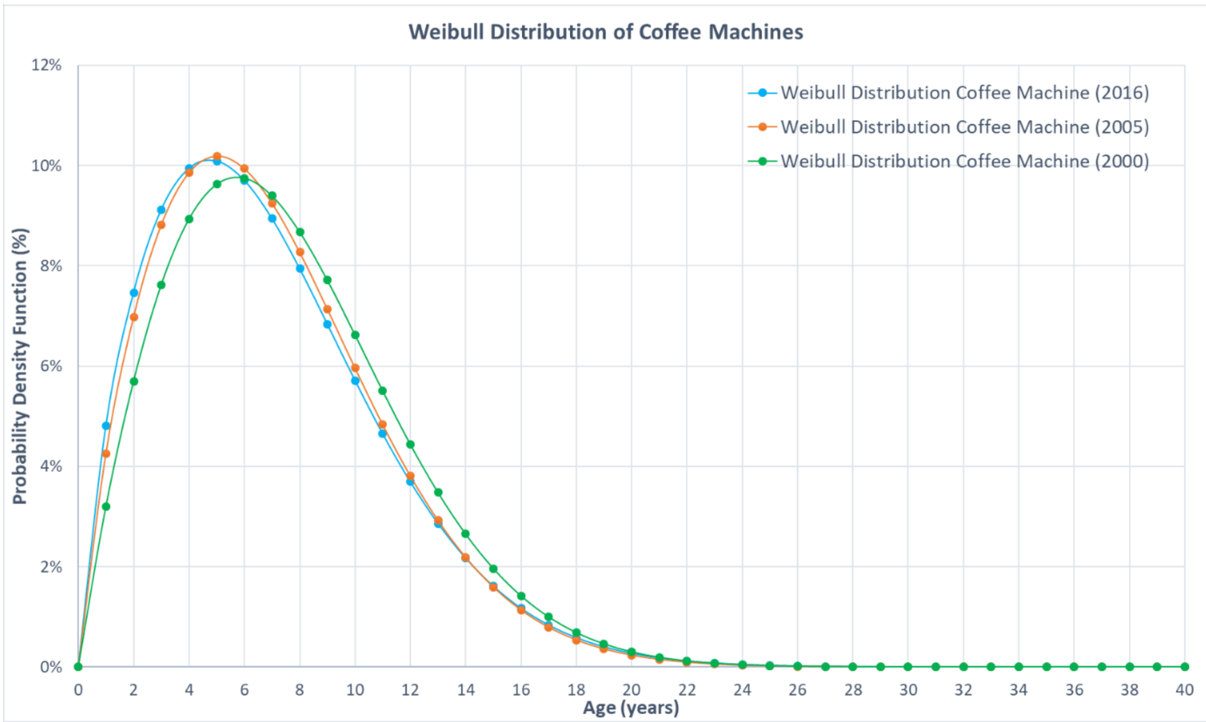
Weibull parameters and distribution profile, reproduced from Wang (2014) and Forti et al. (2018) research, are crucial information in constructing lifespan distribution profiles modeling. The values are shown in Table B.2 and the profiles are shown in Figure B.2. In the Weibull distribution profiles (Figure B.2), the x-axis describes the disposal time (in years) of each electronic products while the y-axis describes the probability density values (in %). The disposal rate of different electronic products is determined by examining the slope of the curves. The steeper the slope means the product stays shorter in the consumers' possession before eventually being thrown away, and vice versa. The horizontal shift of the profile demonstrates the lifespan of the electronic products, including their service and storage ("hibernation") time.

Table B.2 Weibull Parameter used for WEEE modeling.

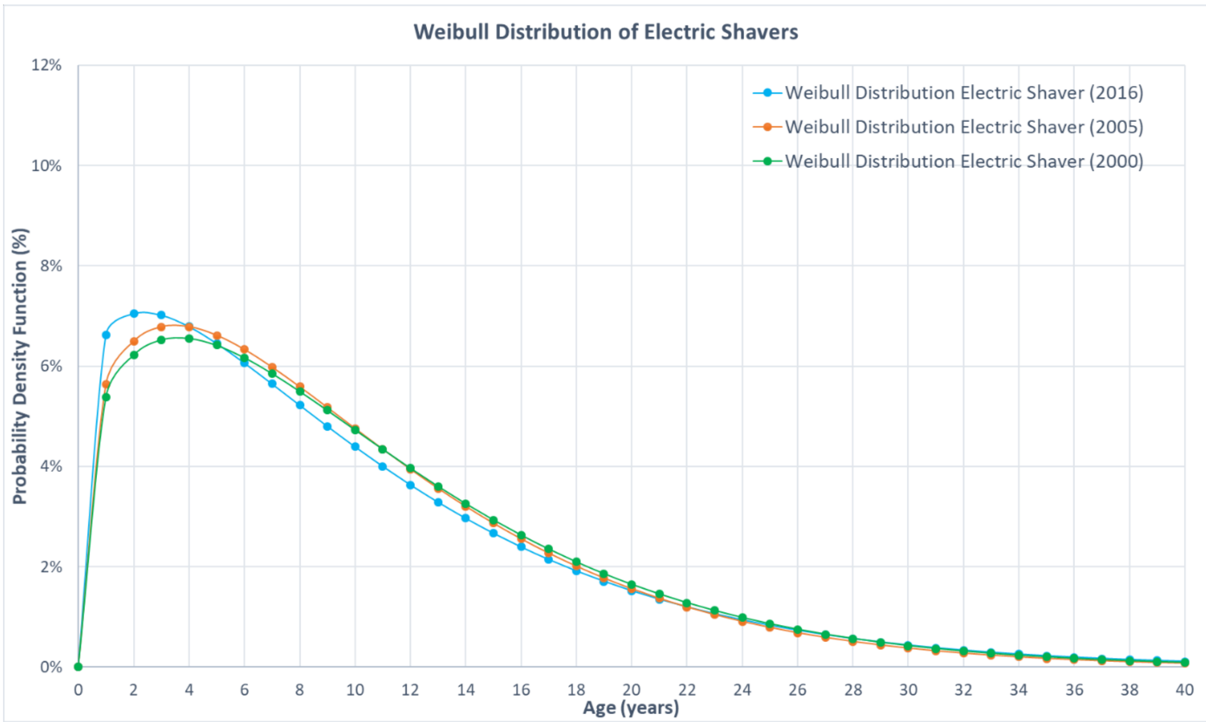
Data Source	***Vacuum Cleaner***			***Coffee Machine***			***Electric Shaver***			***Year***
	α(t)	***β(t)***	***Avg. (years)***	***α(t)***	***β(t)***	***Avg. (years)***	***α(t)***	***β(t)***	***Avg. (years)***	
Wang (2014)	*1,5*	*10,4*	*9*	*1,9*	*8,5*	*8*	*1,3*	*11,2*	*10*	*2000*
Wang (2014)	*1,5*	*10,3*	*9*	*1,8*	*7,9*	*8*	*1,3*	*10,8*	*10*	*2005*
Forti et al. (2018)	*1,4*	*10,2*	*9*	*1,7*	*7,8*	*7*	*1,2*	*10,7*	*10*	*2016*
Eurostat (2022b)	*1,5*	*10,3*	*9*	*1,7*	*7,8*	*7*	*1,3*	*10,7*	*10*	*1980 - 2018*


Weibull Distribution of Vacuum Cleaners
Weibull Distribution Vacuum Cleaner (2016)
Weibull Distribution Vacuum Cleaner (2005)
Weibull Distribution Vacuum Cleaner (2000)
Probability Density Function (%)
Age (years)
12%
10%
8%
6%
4%
2%
0%
0 2 4 6 8 10 12 14 16 18 20 22 24 26 28 30 32 34 36 38 40

(A)


Weibull Distribution of Coffee Machines
Weibull Distribution Coffee Machine (2016)
Weibull Distribution Coffee Machine (2005)
Weibull Distribution Coffee Machine (2000)
Probability Density Function (%)
Age (years)
12%
10%
8%
6%
4%
2%
0%
0 2 4 6 8 10 12 14 16 18 20 22 24 26 28 30 32 34 36 38 40

(B)



(C)

Figure B.2 Weibull Distribution Profiles for Vacuum Cleaners (A), Coffee Machines (B), and Electric Shavers (C). Reproduced from Wang (2014) and Forti et al. (2018).

SECTION 3: COLLECTION RATE, PRE-PROCESSING EFFICIENCY, AND PLASTIC CONTENT IN BASE, INTERMEDIATE, AND POSITIVE SCENARIOS

Table B.3 The evolution of the market share of robotic vacuum cleaners (RVCs) in different scenarios.

	Market Share of Robotic Vacuum Cleaners (RVCs)		
	Base Scenario	*Intermediate Scenario*	*Positive Scenario*
2020	9%	9%	9%
2021	11%	12%	14%
2022	12%	15%	19%
2023	14%	18%	24%
2024	15%	21%	29%
2025	17%	24%	34%
2026	18%	27%	39%
2027	20%	30%	44%

2028	21%	33%	49%
2029	23%	36%	54%
2030	24%	39%	59%

Table B.4 The evolution of collection rate, pre-processing efficiency, and plastics content in different scenarios.

	Base Scenario			**Intermediate Scenario**			**Positive Scenario**		
	Collection	*Pre-Processing*	*Plastics Content*	*Collection*	*Pre-Processing*	*Plastic Content*	*Collection*	*Pre-Processing*	*Plastics Content*
2020	49%	48%	73%	49%	48%	73%	49%	58%	73%
2021	50%	49%	71%	52%	51%	71%	54%	53%	70%
2022	52%	51%	71%	55%	54%	70%	58%	57%	70%
2023	53%	52%	71%	58%	57%	71%	63%	62%	69%
2024	54%	54%	70%	61%	60%	69%	67%	66%	68%
2025	56%	55%	70%	64%	63%	69%	72%	71%	67%
2026	57%	56%	70%	67%	66%	68%	76%	75%	66%
2027	59%	58%	69%	70%	69%	68%	81%	80%	65%
2028	60%	59%	69%	73%	72%	67%	85%	84%	64%
2029	61%	61%	69%	76%	75%	67%	90%	89%	63%
2030	63%	62%	69%	79%	78%	66%	94%	93%	62%

C

APPENDIX C. MATERIAL FLOW ANALYSIS, RECYCLING PERFORMANCE, AND ECONOMIC BALANCE OF AN IMPROVED MECHANICAL RECYCLING PROCESS FOR POST-CONSUMER HOUSEHOLD FLEXIBLE PLASTICS

SECTION 1: PILOT SORTING TRIALS PROTOCOL OF THE QRP ADDITIONAL SORTING OF DSD 310-1 AND DSD 323-2 BALES

Section 1.1 Process flow diagram of DSD 310-1 and DSD 323-2 trials

Figure C.1 shows the flow diagram of the performed trials at Nationaal Testcentrum Circulaire Plastics (NTCP) facility, following sorting protocols developed by NTCP. There are three setups that are tested during the trial, namely:

1. Processing DSD 310-1 to retrieve PE Film Natural and all colors PE films (**Trial 1**)
2. Processing DSD 323-2 to retrieve PP and PO (PP + PE) (**Trial 2**)
3. Mixed of DSD 310-1 and DSD 323-2 to retrieve PE Natural, PP, and PO (**Trial 3**)

From these trials, three information are obtained:

1. Waste composition of DSD 310-1 & DSD 323-2 bales (Table 4.1 in the main text)
2. Separation efficiencies of five Optical Sorters:
    a. NIR-VIS LDPE Natural, targeting PE film transparent clear
    b. NIR PE Cleaner, targeting non-PE materials
    c. NIR PP Film, targeting PP materials
    d. NIR PP Cleaner, targeting non-PP materials
    e. NIR PO Cleaner, targeting remaining residual and contaminants (non-PO materials)
3. Material composition of the output streams

The information mentioned above are determined by identifying material composition at every sampling point and quantifying the mass balance of the trials.

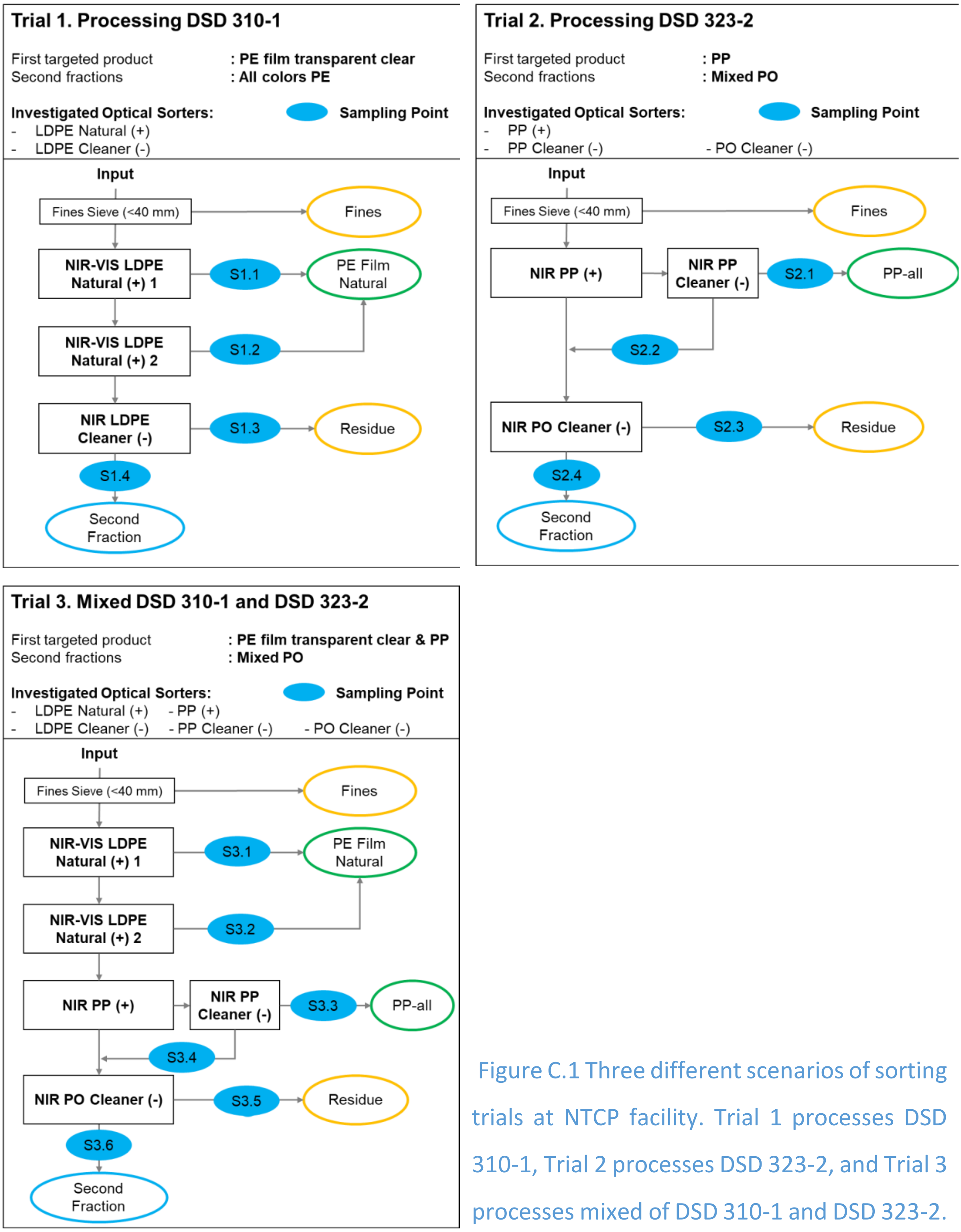


Figure C.1 Three different scenarios of sorting trials at NTCP facility. Trial 1 processes DSD 310-1, Trial 2 processes DSD 323-2, and Trial 3 processes mixed of DSD 310-1 and DSD 323-2.

Section 1.2 Material handling and preparation for the trials

Seven bales DSD 310-1 (net weight 3600 kg) and six bales DSD 323-2 (3540 kg) are prepared for the trials. On the arrival and preparation for the trials, manual debaling is performed at the facility to remove clogged materials, which is partly attributed to the baling and compression at the sorting facilities. The debaling process is also supported by a crane to further loosen the material upon the trials. Before the materials being fed into the testing line, all the metals are carefully removed because the overbelt magnet and windshifters are switched off during the trial. Ballistic separator is switched on only to loosening the materials (i.e. not meant for material separation). After the weight scale had been calibrated, the materials for the test are weighed and fed into the NTCP sorting line.

Section 1.3 Material sampling protocol

The output materials at each sampling point are collected into big bags, weighed, and labelled. Thereafter, the following sampling procedure is followed:

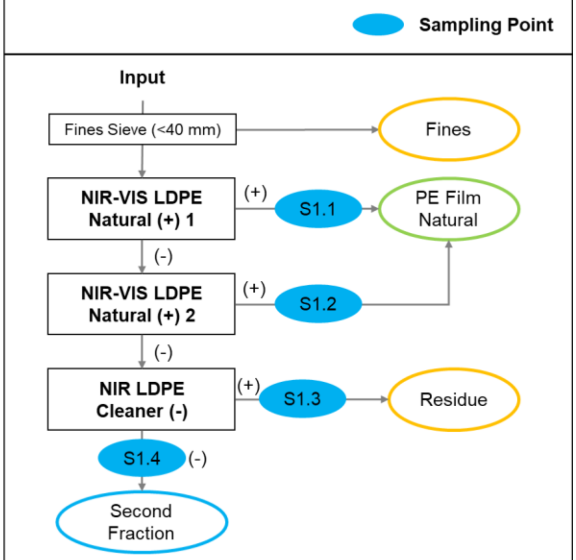
1. Three big bags are randomly selected for material composition characterization.
2. Two out of the selected three bags (approximately 160 kg in weight) are further chosen (labelled as bag sample A and B) while one bag (labelled bag sample C) is stored as a spare.
3. The materials from each bag are mixed on the floor and equally divided in four equal segments (see Figure C.2)
4. From this division, two out of four segments (approximately 60 kg) are randomly chosen and mixed again
5. The same procedure as in step three and four is repeated until around 7.5 kg of sample are collected for material composition characterization
6. The randomly collected materials are then visually inspected, weighed and manually classified according to the waste category (see Table 4.1 in the main text)

Figure C.2 Division of the output streams at each sampling point from the two randomly selected big bags that are collected at each sampling point.

Section 1.4 Results: mass balance of the trials

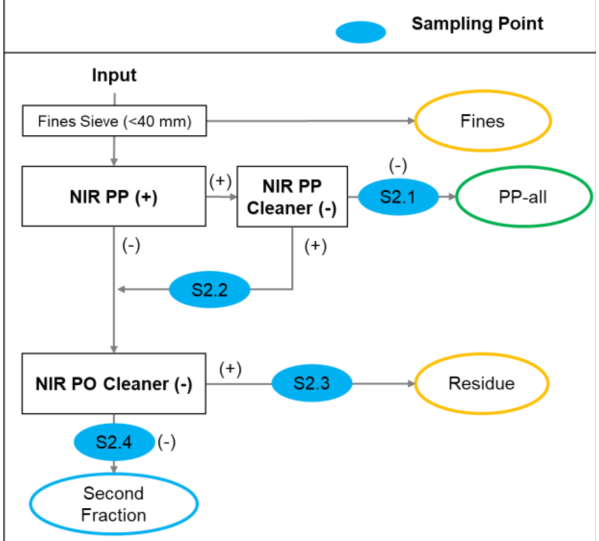
Figure C.3 shows the mass balances of the three different performed trials. The data on mass balances and the material composition characterization (see section C.1.5) are then combined in order to determine the waste input composition (using reverse mass balance calculation) and separation efficiencies (see section C.1.6).

Trial 1. Processing DSD 310-1



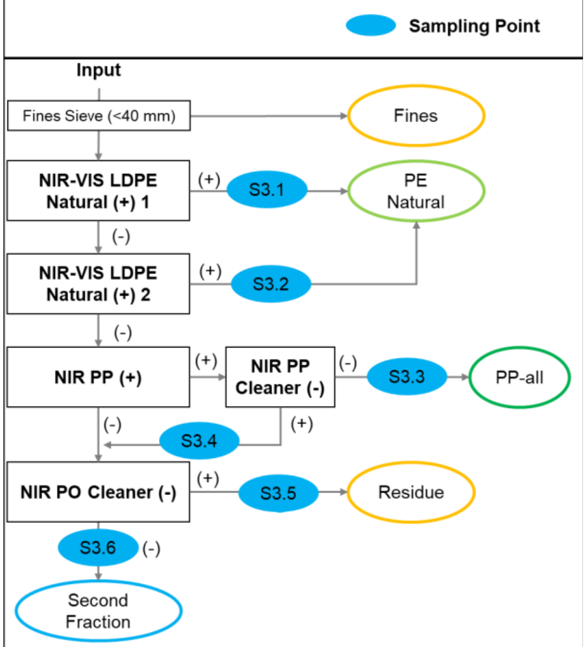
Mass input: **1857.2 kg**			
Process	(+) [kg]	(-) [kg]	Fines [kg]
NIR-VIS LDPE Natural (+) 1	327.8	1497.2	32.2
NIR-VIS LDPE Natural (+) 2	151.8	1316.8	
NIR LDPE Cleaner (-)	117.6	1170.4	

Trial 2. Processing DSD 323-2



Mass input: **1983.2 kg**			
Process	(+) [kg]	(-) [kg]	Fines [kg]
NIR PP (+)	532.2	1497.2	102.6
NIR PP Cleaner (-)	138.8	383.2	
NIR PO Cleaner (-)	264.6	1170.4	

Trial 3. Mixed DSD 310- and DSD 323-2



Mass input: **1917.8 kg**			
Process	(+) [kg]	(-) [kg]	Fines [kg]
NIR-VIS LDPE Natural (+) 1	224.4	1620.6	72.8
NIR-VIS LDPE Natural (+) 2	118.6	1474.6	
NIR PP (+)	262	1187	
NIR PP Cleaner (-)	65.4	192.2	
NIR PO Cleaner (-)	192.8	1009	

Figure C.3 The mass balances of three trials. Please note that during the trial some spills occur.

Section 1.5 Results: material composition and characterization

The following Table C.1 shows the average material composition (from bag samples A and B) at each sampling point.

Table C.1 Average material composition of bag sample at each sampling point from two randomly selected big bags.

Trial 1	Sampling Point S1.1	Sampling Point S1.2	Sampling Point S1.3	Sampling Point S1.4
PE Film Transparent Clear	81%	77%	4%	36%
PE Film Others	13%	17%	4%	30%
PP Film Transparent	0%	0%	22%	1%
PP Film Others	0%	0%	4%	1%
Other Plastic Film	1%	1%	12%	4%
Multi-material Films	1%	1%	8%	6%
PE Rigid	0%	0%	0%	2%
PP Rigid	1%	0%	10%	1%
Paper and Residue	3%	4%	36%	20%
Total	100%	100%	100%	100%

Trial 2	Sampling Point S2.1	Sampling Point S2.2	Sampling Point S2.3	Sampling Point S2.4
PE Film Transparent Clear	2%	21%	4%	32%
PE Film Others	1%	8%	3%	15%
PP Film Transparent	34%	10%	1%	5%
PP Film Others	10%	12%	2%	4%
Other Plastic Film	4%	3%	8%	3%
Multi-material Films	1%	6%	9%	8%
PE Rigid	0%	0%	0%	1%
PP Rigid	32%	7%	1%	5%
Paper and Residue	16%	34%	72%	28%
Total	100%	100%	100%	100%

Trial 3	Sampling Point S3.1	Sampling Point S3.2	Sampling Point S3.3	Sampling Point S3.4	Sampling Point S3.5	Sampling Point S3.6
PE Film Transparent Clear	77%	75%	2%	28%	4%	30%
PE Film Others	16%	15%	1%	12%	2%	23%
PP Film Transparent	1%	1%	30%	12%	2%	4%
PP Film Others	0%	0%	14%	6%	0%	1%
Other Plastic Film	1%	1%	2%	2%	11%	4%
Multi-material Films	2%	1%	2%	5%	9%	10%
PE Rigid	0%	0%	0%	0%	0%	2%
PP Rigid	1%	1%	31%	9%	0%	3%
Paper and Residue	3%	6%	18%	24%	72%	23%
Total	100%	100%	100%	100%	100%	100%

Section 1.6 Quantification of the separation efficiencies for each NIR used in the trials

Figure C.4 shows an example of the quantification of separation efficiencies based on the mass balances (Figure C.3) and mass composition data (Table 4.1 in the main text). In this example (Figure C.4), 898.9 kg LDPE Transparent Clear is fed into the sorting line [see Trial 1, NIR-VIS LDPE Natural (+) 1], in which 265 kg are sorted to the output stream (PE Natural) and 633.9 kg are forwarded to the next NIR machine. In this case, the separation efficiency of LDPE Transparent Clear at NIR-VIS LDPE Natural (+) is 29% to PE Stream and 71% to the subsequent process. Thereafter, the average value of the separation efficiency from four different test runs (i.e., two are performed in trial 1 and trial 3, respectively) is used in the MFA model.

The same calculation is applied to the rest of the waste category at every NIR in the trials. The same principle is applied to quantify the total of 297 separation efficiencies, i.e., 27 waste categories *x* 11 NIRs from three different trials.

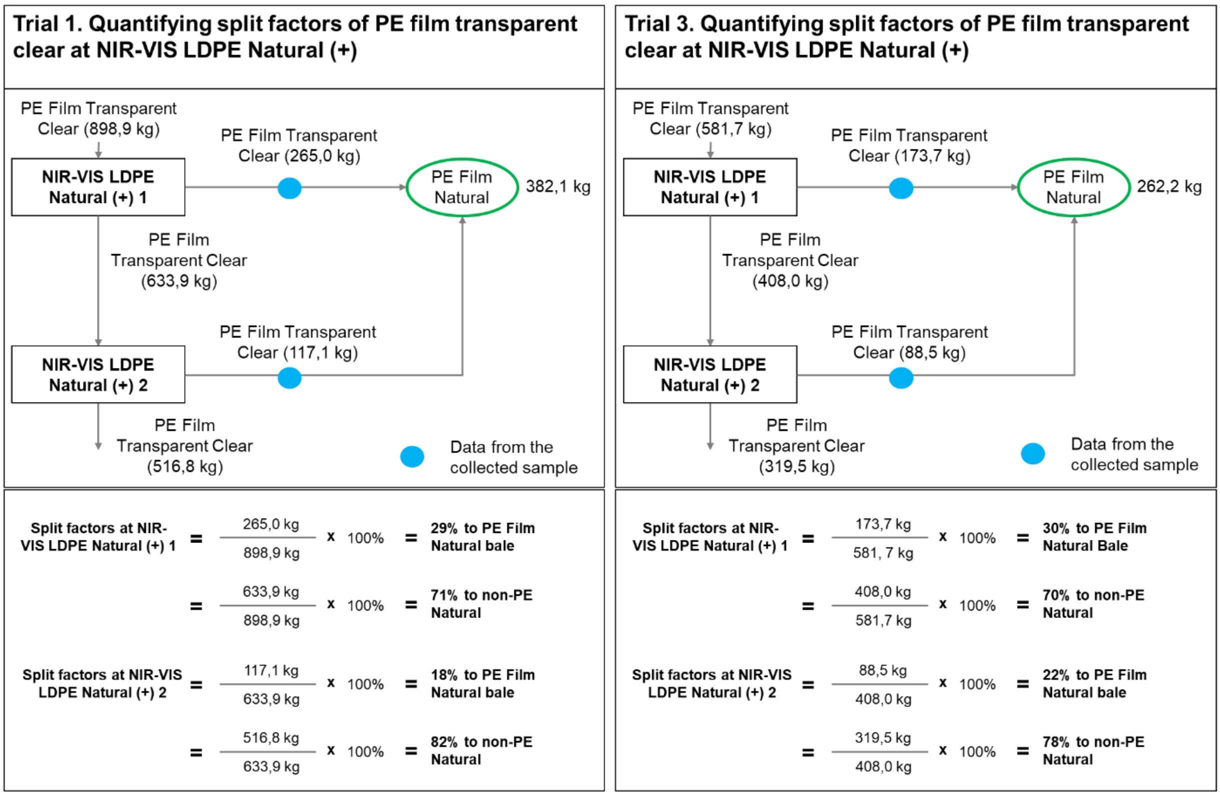


Figure C.4 An illustration of sorting efficiency quantification based on the material flow analysis. In this example, the mass flow of LDPE Transparent Clear [in kg] is quantified based on the reverse mass balance calculation from four different test runs in two different trials.

SECTION 2: SEPARATION EFFICIENCIES BASED ON THE PILOT TRIALS

Table C.2 Average separation efficiency values for 27 waste categories in 11 NIRs. The values are used in MFA.

NIRs		NIR-VIS LDPE Natural (+)		NIR LDPE Cleaner (-)		NIR PP (+)		NIR PP Cleaner (-)		NIR PO Cleaner (-)	
Output Stream		*PE film natural bale*	*Non-PE Natural*	*PE Flex bale*	*Non-PE*	*PP Cleaner (-)*	*Non-PP*	*PP Film bale*	*Non-PP*	*PO New bale*	*Non-PO*
PE Film	Transparent Clear	25%	75%	99%	1%	7%	93%	20%	80%	97%	3%
	Transparent clear printed	9%	91%	100%	0%	5%	95%	4%	96%	99%	1%
	Transparent coloured	7%	93%	98%	2%	10%	90%	19%	81%	95%	5%
	Opaque coloured	11%	89%	99%	1%	3%	97%	11%	89%	97%	3%
	Black	4%	96%	97%	3%	5%	95%	64%	36%	95%	5%
	Metalized	0%	100%	0%	100%	1%	99%	0%	100%	100%	0%
PP Film	Transparent Clear	0%	100%	31%	69%	66%	34%	91%	9%	93%	7%
	Transparent clear printed	1%	99%	41%	59%	67%	33%	87%	13%	92%	8%
	Transparent coloured	0%	100%	0%	100%	2%	98%	100%	0%	100%	0%
	Opaque coloured	1%	99%	63%	37%	65%	35%	90%	10%	95%	5%
	Black	1%	99%	98%	2%	4%	96%	58%	42%	0%	100%
	Metalized	0%	100%	40%	60%	43%	57%	52%	48%	91%	9%
Other Film	Plastic	1%	99%	87%	13%	23%	77%	55%	45%	54%	46%
Multi-material Flexibles	Aluminium laminate	1%	99%	10%	90%	13%	87%	66%	34%	36%	64%
	Paper laminate	1%	99%	70%	30%	11%	89%	44%	56%	38%	62%
	Other laminate	2%	98%	100%	0%	5%	95%	54%	46%	96%	4%
PE Rigid	All	0%	100%	100%	0%	3%	97%	5%	95%	100%	0%
PP Rigid	All	4%	96%	39%	61%	65%	35%	98%	2%	97%	3%
Other plastics flexibles	Textile, fabric	2%	98%	36%	64%	37%	63%	92%	8%	32%	68%
	Nets	2%	98%	97%	3%	6%	94%	62%	38%	98%	2%
	Foamed	3%	97%	75%	25%	5%	95%	52%	48%	52%	48%
Paper	Print, cardboard	1%	99%	51%	49%	7%	93%	43%	57%	33%	67%
	Hygiene, tissue	0%	100%	15%	85%	14%	86%	34%	66%	22%	78%
Residue	Compound	0%	100%	32%	68%	53%	47%	52%	48%	46%	54%
	Clogged	3%	97%	96%	4%	24%	76%	75%	25%	88%	12%
	Others	1%	99%	85%	15%	13%	87%	59%	41%	33%	67%
	Fine fraction	4%	96%	82%	18%	11%	89%	58%	42%	84%	16%

**The highlighted separation efficiency are the targeted materials detected by the NIR sensors. The colors follow the color coding reported at the trials setup and mass balance calculation (section C.1.4).*

Table C.3 Average values of the aggregated separation efficiencies (shown in %) of recycling equipment for each waste category in sub-group level (see Table 4.1 in the main text for more details on the waste classification). *Except for metalized PE or metalized PP (see Table SI1): 30% floats and 70% sinks in the density separation ; 90% to regranulate and 10% to residue in the extrusion.

Recycling Unit	Cold Washing (in %)		Hot Washing (in %)		Density Separation (in %)		Extrusion (in %)	
Output	Next step	Residue	Next step	Residue	Float	Sink	Regranulate	Residue
PE Film Natural	99	1	99	1	98	2	97	3
PE Film Others	99	1	99	1	*98	2	*97	3
PP Film Transparent	99	1	99	1	99	1	97	3
PP Film Others	99	1	99	1	*99	1	*97	3
Other Plastic Film	99	1	99	1	30	70	5	95
Multi-material Film	99	1	99	1	30	70	5	95
PE Rigid	99	1	99	1	98	2	97	3
PP Rigid	99	1	99	1	99	1	97	3
Paper, Fines, and Residue	5	95	5	95	5	95	5	95

SECTION 3: TARGETED OUTPUT STREAMS FOR THE QUANTIFICATION OF NET RECOVERY

Table C.4 Targeted regranulates for every waste category, which is used as the reference for net recovery.

Targeted Regranulates	**Waste categories**		
	Main Group	*Sub-group*	*Sub-category*
rLDPE ; rPO Flex (conventional recycling) rPE Film Natural ; rPE Flex ; rPO New (QRP)	PE Films	PE Film Transparent Clear	
rLDPE ; rPO Flex (conventional recycling) rPE Flex ; rPO New (QRP)		PE Film Others	Transparent clear printed Transparent coloured Opaque coloured Black Metalized
	PE-Rigid		
rPO Flex (conventional recycling) rPP Film ; rPO New (QRP)	PP-Films	PP Film Transparent	Transparent clear Transparent clear printed Transparent coloured
		PP Film Others	Opaque coloured Black Metalized
	PP-Rigid		
Residual treatment	Other-film		Plastic
	Multi-material Film		Aluminium laminate Paper laminate Other laminate
	Other plastics film		Textile, fabric Nets Foamed
	Paper		Print, cardboard Hygiene tissue
	Residue		Compound Clogged Others
	Fines		

SECTION 4: MASS BALANCE OUTPUT AND DISTRIBUTION

Table C.5 Mass balance (shown in ton) of PE and PP materials (films and rigid) at input and different output streams via scenario 1: QRP with Tier 1 recycling.

QRP Scenario 1	Total input	rPE Film natural	rPE Flex	rPP Film	rPO New	Residue
PE film transparent clear	14460	3881	4966	65	4283	1266
PE film transparent clear printed	2097	254	1225	1	445	173
PE film transparent coloured	1808	164	1031	8	439	165
PE film opaque coloured	2365	300	1085	3	766	210
PE film black	910	46	517	8	255	85
PE film metalized	221	0	0	0	53	167
PP film transparent clear	1342	2	80	589	535	136
PP film transparent clear printed	1165	4	64	543	434	120
PP film transparent coloured	103	0	0	1	94	7
PP film opaque coloured	776	2	70	356	280	69
PP film black	194	2	97	2	0	93
PP film metalized	500	0	4	25	84	387
PE rigid	500	2	332	0	130	35
PP rigid	2420	19	86	1272	847	196

Table C.6 Output distribution (shown in % output mass) of PE and PP materials (films and rigid) in different output streams via scenario 1: QRP with Tier 1 recycling.

QRP Scenario 1	Total	rPE Film natural	rPE Flex	rPP Film	rPO New	Residue
PE film transparent clear	100	27	34	0	30	9
PE film transparent clear printed	100	12	58	0	21	8
PE film transparent coloured	100	9	57	0	24	9
PE film opaque coloured	100	13	46	0	32	9
PE film black	100	5	57	1	28	9
PE film metalized	100	0	0	0	24	76
PP film transparent clear	100	0	6	44	40	10
PP film transparent clear printed	100	0	5	47	37	10
PP film transparent coloured	100	0	0	1	92	7
PP film opaque coloured	100	0	9	46	36	9
PP film black	100	1	50	1	0	48
PP film metalized	100	0	1	5	17	77
PE rigid	100	0	66	0	26	7
PP rigid	100	1	4	53	35	8

Table C.7 Mass balance (shown in ton) of PE and PP materials (films and rigid) at input and different output streams via scenario 2: QRP with Tier 1 and Tier 2 recycling.

Scenario 2	Total input	rPE Film natural	rPE Flex	rPP Film	rPO New	Residue
PE film transparent clear	14460	3881	5119	65	4415	980
PE film transparent clear printed	2097	254	1263	1	459	121
PE film transparent coloured	1808	164	1063	8	452	120
PE film opaque coloured	2365	300	1119	3	789	153
PE film black	910	46	533	8	262	61
PE film metalized	221	0	0	0	59	161
PP film transparent clear	1342	2	83	589	552	117
PP film transparent clear printed	1165	4	66	543	447	105
PP film transparent coloured	103	0	0	1	97	4
PP film opaque coloured	776	2	72	356	289	58
PP film black	194	2	100	2	0	90
PP film metalized	500	0	4	25	94	377
PE rigid	500	2	342	0	134	21
PP rigid	2420	19	89	1272	873	167

Table C.8 Output distribution (shown in % output mass) of PE and PP materials (films and rigid) in different output streams via scenario 2: QRP with Tier 1 and Tier 2 recycling.

Scenario 2	Total	rPE Film natural	rPE Flex	rPP Film	rPO New	Residue
PE film transparent clear	100	27	35	0	31	7
PE film transparent clear printed	100	12	60	0	22	6
PE film transparent coloured	100	9	59	0	25	7
PE film opaque coloured	100	13	47	0	33	6
PE film black	100	5	59	1	29	7
PE film metalized	100	0	0	0	27	73
PP film transparent clear	100	0	6	44	41	9
PP film transparent clear printed	100	0	6	47	38	9
PP film transparent coloured	100	0	0	1	94	4
PP film opaque coloured	100	0	9	46	37	7
PP film black	100	1	52	1	0	46
PP film metalized	100	0	1	5	19	75
PE rigid	100	0	68	0	27	4
PP rigid	100	1	4	53	36	7

Table C.9 Mass balance (shown in ton) of PE and PP materials (films and rigid) at input and different output streams (regranulates and residue) via baseline scenario: conventional recycling.

Baseline scenario	Total input	Baseline DSD 310-1	Baseline DSD 323-2	Residue
PE film transparent clear	14460	9080	4544	864
PE film transparent clear printed	2097	1526	465	107
PE film transparent coloured	1808	1230	459	119
PE film opaque coloured	2365	1427	806	132
PE film black	910	579	269	62
PE film metalized	221	0	21	200
PP film transparent clear	1342	83	1020	239
PP film transparent clear printed	1165	68	955	142
PP film transparent coloured	103	0	79	24
PP film opaque coloured	776	73	629	74
PP film black	194	103	81	10
PP film metalized	500	4	124	371
PE rigid	500	345	134	21
PP rigid	2420	97	2074	249

Table C.10 Output distribution (shown in % output mass) of PE and PP materials (films and rigid) in different output streams (regranulates and residue) via conventional recycling.

Baseline scenario	Total input	Baseline DSD 310-1	Baseline DSD 323-2	Residue
PE film transparent clear	100	63	31	6
PE film transparent clear printed	100	73	22	5
PE film transparent coloured	100	68	25	7
PE film opaque coloured	100	60	34	6
PE film black	100	64	30	7
PE film metalized	100	0	9	91
PP film transparent clear	100	6	76	18
PP film transparent clear printed	100	6	82	12
PP film transparent coloured	100	0	77	23
PP film opaque coloured	100	9	81	9
PP film black	100	53	42	5
PP film metalized	100	1	25	74
PE rigid	100	69	27	4
PP rigid	100	4	86	10

SECTION 5: EXPERIMENTAL COMPOSITIONAL ANALYSES OF THE QRP FLAKE AND REGRANULATE

Table C.11 Modeled compositional data of the QRP flakes, and regranulates and experimental compositional analyses (shown in %). *The table only shows the composition of PE and PP that are characterized by DSC analysis (for regranulates) and FTIR analysis (for flakes). Detailed information of the experimental results can be found in Table C.12. In the Modeled composition data, PE and PP composition include all film (transparent, colored/printed, metalized) and rigid plastics.

	PE Film Natural				PE Flex			
	Flakes		Regranulates		Flakes		Regranulates	
Composition (in %)	***PE***	***PP***	***PE***	***PP***	***PE***	***PP***	***PE***	***PP***
Modeled Compositional Data: Scenario 1	99	1	99	< 1	91	4	96	4
Modeled Compositional Data: Scenario 2	99	1	99	< 1	91	4	96	4
Experimental Compositional Analysis: Scenario 1	90	7	92	6	90	7	87	8
Experimental Compositional Analysis: Scenario 2	90	7	92	6	-	-	78	13

Table continued below

	PP Film				PO New			
	Flakes		Regranulates		Flakes		Regranulates	
Composition (in %)	***PE***	***PP***	***PE***	***PP***	***PE***	***PP***	***PE***	***PP***
Modeled Compositional Data: Scenario 1	3	95	3	97	69	25	74	26
Modeled Compositional Data: Scenario 2	3	95	3	97	69	25	73	26
Experimental Compositional Analysis: Scenario 1	12	83	9	79	81	10	78	13
Experimental Compositional Analysis: Scenario 2	12	83	9	79	-	-	68	20

Table C.12 Results experimental compositional analysis from DSC and FTIR analysis (shown in %).

Flakes (via FTIR)	PE content	PP content	Rest
Tier 1 PE Film Natural	89.6	7.0	3.4
Tier 1 PE Flex	89.3	6.5	4.2
Tier 1 PP Film	11.5	82.7	5.8
Tier 1 PO New	81.2	10.4	8.4

Regranulate (via DSC)	PE content	PP content	Rest
Tier 1 PE Film Natural	92.0	6.0	2.0
Tier 1 PE Flex	87.0	8.0	5.0
Tier 1 PP Film	9.0	79.0	12.0
Tier 1 PO New	78.0	14.0	8.0
Tier 2 PE flex	78.0	13.0	9.0
Tier 2 PO new	68.0	20.0	12.0

SECTION 6: SENSITIVITY ANALYSES

Table C.13 The ± 25% values of DSD 310-1 and DSD 323-2 bale compositions for the sensitivity analysis.

DSD 310	Composition	(-) 25%	(+) 25%	**DSD 323** Composition	(-) 25%	(+) 25%
310_PE Film Transparent Clear	48,40%	36,30%	60,50%	23,90%	17,93%	29,88%
310_PE Film transparent clear printed	8,04%	6,03%	10,05%	2,44%	1,83%	3,05%
310_PE Film Transparent Coloured	6,62%	4,97%	8,28%	2,42%	1,81%	3,02%
310_PE Film Opaque Coloured	7,58%	5,69%	9,48%	4,24%	3,18%	5,30%
310_PE Film Black	3,14%	2,35%	3,92%	1,41%	1,06%	1,77%
310_PE Film Metalized	0,71%	0,54%	0,89%	0,39%	0,29%	0,48%
310_PP Film Transparent Clear	1,40%	2,25%	0,38%	5,31%	3,98%	6,64%
310_PP Film Transparent Clear Printed	0,85%	1,70%	0,23%	4,97%	3,73%	6,22%
310_PP Film Transparent Coloured	0,10%	0,95%	0,03%	0,41%	0,31%	0,52%
310_PP Film Opaque Coloured	0,60%	1,45%	0,16%	3,27%	2,46%	4,09%
310_PP Film Black	0,55%	1,40%	0,15%	0,42%	0,32%	0,53%
310_PP Film Metalized	0,19%	1,04%	0,05%	2,30%	1,73%	2,88%
310_Other Plastic Film	1,10%	1,95%	0,30%	0,10%	0,90%	0,07%
310_Other Plastic Metalized	0,00%	0,85%	0,00%	0,00%	0,80%	0,00%
310_Multilayer Flexibles Aluminum	0,75%	1,60%	0,20%	1,47%	2,27%	1,08%
310_Multilayer Flexibles Paper	0,34%	1,19%	0,09%	0,61%	1,41%	0,45%
310_Multilayer Flexibles Other	4,31%	5,15%	1,16%	4,72%	5,53%	3,47%
310_PE Rigid	1,80%	2,65%	0,49%	0,70%	1,50%	0,51%
310_PP Rigid	1,30%	2,15%	0,35%	10,80%	11,60%	7,93%
310_Other Plastics Flexibles Textile	0,82%	1,67%	0,22%	1,22%	2,02%	0,89%
310_Other Plastics Flexibles Nets	0,71%	1,55%	0,19%	1,16%	1,96%	0,85%
310_Other Plastics Flexibles Foamed	1,07%	1,92%	0,29%	1,83%	2,63%	1,34%
310_Paper Print Cardboard	1,17%	2,01%	0,31%	2,83%	3,63%	2,08%
310_Paper Tissue	0,33%	1,18%	0,09%	0,87%	1,68%	0,64%
310_Compound	0,00%	0,85%	0,00%	1,20%	2,00%	0,88%
310_Clogged	3,30%	4,15%	0,89%	8,04%	8,85%	5,91%
310_Others	0,00%	0,85%	0,00%	1,76%	2,56%	1,29%
310_Fine Fraction	4,80%	5,65%	1,29%	11,20%	12,00%	8,23%

Table C.14 The ± 25% values of separation efficiencies of NIR-VIS LDPE Natural.

	NIR-VIS LDPE Natural (targeting PE Film Transparent Clear)					
	To PE Film Bale			To NIR LDPE Cleaner		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	18,7%	24,9%	31,1%	81,3%	75,1%	68,9%
PE Film Transparent Clear Printed	11,2%	9,0%	6,7%	88,8%	91,0%	93,3%
PE Film Transparent Coloured	8,7%	7,0%	5,2%	91,3%	93,0%	94,8%
PE Film Opaque Coloured	14,2%	11,4%	8,5%	85,8%	88,6%	91,5%
PE Film Black	5,0%	4,0%	3,0%	95,0%	96,0%	97,0%
PE Film Metalized	0,0%	0,0%	0,0%	100,0%	100,0%	100,0%
PP Film Transparent Clear	0,4%	0,3%	0,3%	99,6%	99,7%	99,7%
PP Film Transparent Clear Printed	1,5%	1,2%	0,9%	98,5%	98,8%	99,1%
PP Film Transparent Coloured	0,0%	0,0%	2,5%	100,0%	100,0%	97,5%
PP Film Opaque Coloured	1,2%	1,0%	0,7%	98,8%	99,0%	99,3%
PP Film Black	1,5%	1,2%	0,9%	98,5%	98,8%	99,1%
PP Film Metalized	0,3%	0,2%	0,2%	99,7%	99,8%	99,8%
Other Plastic Film	1,2%	1,0%	0,7%	98,8%	99,0%	99,3%
Other Plastic Metalized	1,2%	1,0%	0,7%	98,8%	99,0%	99,3%
Multilayer Flexibles Aluminum	1,3%	1,0%	0,8%	98,7%	99,0%	99,2%
Multilayer Flexibles Paper	0,8%	0,6%	0,5%	99,2%	99,4%	99,5%
Multilayer Flexibles Other	2,8%	2,2%	1,7%	97,2%	97,8%	98,3%
PE Rigid	0,5%	0,4%	0,3%	99,5%	99,6%	99,7%
PP Rigid	5,1%	4,0%	3,0%	94,9%	96,0%	97,0%
Other Plastics Flexibles Textile	3,0%	2,4%	1,8%	97,0%	97,6%	98,2%
Other Plastics Flexibles Nets	2,1%	1,7%	1,3%	97,9%	98,3%	98,7%
Other Plastics Flexibles Foamed	3,6%	2,9%	2,1%	96,4%	97,1%	97,9%
Paper Print Cardboard	1,5%	1,2%	0,9%	98,5%	98,8%	99,1%
Paper Tissue	0,3%	0,2%	0,2%	99,7%	99,8%	99,8%
Compound	0,6%	0,5%	0,3%	99,4%	99,5%	99,7%
Clogged	4,3%	3,5%	2,6%	95,7%	96,5%	97,4%
Others	1,5%	1,2%	0,9%	98,5%	98,8%	99,1%
Fine Fraction	4,5%	3,6%	2,7%	95,5%	96,4%	97,3%

Table C.15 The ± 25% values of separation efficiencies of NIR PE Cleaner.

	NIR LDPE Cleaner (targeting all PE)					
	To PE Flex Bale			To NIR PO Cleaner		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	74,1%	98,8%	100,0%	25,9%	1,2%	0,0%
PE Film Transparent Clear Printed	74,9%	99,9%	100,0%	25,1%	0,1%	0,0%
PE Film Transparent Coloured	73,3%	97,8%	100,0%	26,7%	2,2%	0,0%
PE Film Opaque Coloured	74,3%	99,1%	100,0%	25,7%	0,9%	0,0%
PE Film Black	72,9%	97,2%	100,0%	27,1%	2,8%	0,0%
PE Film Metalized	0,0%	0,0%	0,0%	100,0%	100,0%	100,0%
PP Film Transparent Clear	38,7%	31,0%	23,2%	61,3%	69,0%	76,8%
PP Film Transparent Clear Printed	51,8%	41,4%	31,1%	48,2%	58,6%	68,9%
PP Film Transparent Coloured	0,0%	0,0%	2,5%	100,0%	100,0%	97,5%
PP Film Opaque Coloured	79,1%	63,2%	47,4%	20,9%	36,8%	52,6%
PP Film Black	100,0%	97,6%	73,2%	0,0%	2,4%	26,8%
PP Film Metalized	50,0%	40,0%	30,0%	50,0%	60,0%	70,0%
Other Plastic Film	100,0%	86,8%	65,1%	0,0%	13,2%	34,9%
Other Plastic Metalized	100,0%	86,8%	65,1%	0,0%	13,2%	34,9%
Multilayer Flexibles Aluminum	12,4%	9,9%	7,4%	87,6%	90,1%	92,6%
Multilayer Flexibles Paper	87,9%	70,3%	52,7%	12,1%	29,7%	47,3%
Multilayer Flexibles Other	74,7%	99,6%	100,0%	25,3%	0,4%	0,0%
PE Rigid	100,0%	100,0%	75,0%	0,0%	0,0%	25,0%
PP Rigid	48,4%	38,7%	29,0%	51,6%	61,3%	71,0%
Other Plastics Flexibles Textile	45,5%	36,4%	27,3%	54,5%	63,6%	72,7%
Other Plastics Flexibles Nets	100,0%	97,1%	72,8%	0,0%	2,9%	27,2%
Other Plastics Flexibles Foamed	94,0%	75,2%	56,4%	6,0%	24,8%	43,6%
Paper Print Cardboard	63,8%	51,1%	38,3%	36,2%	48,9%	61,7%
Paper Tissue	19,2%	15,3%	11,5%	80,8%	84,7%	88,5%
Compound	39,6%	31,7%	23,8%	60,4%	68,3%	76,2%
Clogged	100,0%	96,0%	72,0%	0,0%	4,0%	28,0%
Others	100,0%	85,0%	63,7%	0,0%	15,0%	36,3%
Fine Fraction	100,0%	81,5%	61,1%	0,0%	18,5%	38,9%

Table C.16 The ± 25% values of separation efficiencies of NIR PP.

	NIR PP (targeting all PP)					
	To NIR PP Cleaner			To NIR PO Cleaner		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	9,3%	7,5%	5,6%	90,7%	92,5%	94,4%
PE Film Transparent Clear Printed	6,0%	4,8%	3,6%	94,0%	95,2%	96,4%
PE Film Transparent Coloured	12,4%	10,0%	7,5%	87,6%	90,0%	92,5%
PE Film Opaque Coloured	4,4%	3,5%	2,6%	95,6%	96,5%	97,4%
PE Film Black	5,7%	4,6%	3,4%	94,3%	95,4%	96,6%
PE Film Metalized	0,9%	0,7%	0,5%	99,1%	99,3%	99,5%
PP Film Transparent Clear	49,2%	65,7%	82,1%	50,8%	34,3%	17,9%
PP Film Transparent Clear Printed	50,5%	67,4%	84,2%	49,5%	32,6%	15,8%
PP Film Transparent Coloured	1,5%	2,0%	2,4%	98,5%	98,0%	97,6%
PP Film Opaque Coloured	48,6%	64,8%	81,0%	51,4%	35,2%	19,0%
PP Film Black	2,9%	3,9%	4,9%	97,1%	96,1%	95,1%
PP Film Metalized	32,2%	42,9%	53,7%	67,8%	57,1%	46,3%
Other Plastic Film	28,2%	22,6%	16,9%	71,8%	77,4%	83,1%
Other Plastic Metalized	28,2%	22,6%	16,9%	71,8%	77,4%	83,1%
Multilayer Flexibles Aluminum	16,4%	13,1%	9,8%	83,6%	86,9%	90,2%
Multilayer Flexibles Paper	13,2%	10,5%	7,9%	86,8%	89,5%	92,1%
Multilayer Flexibles Other	6,0%	4,8%	3,6%	94,0%	95,2%	96,4%
PE Rigid	3,9%	3,1%	2,3%	96,1%	96,9%	97,7%
PP Rigid	48,5%	64,7%	80,9%	51,5%	35,3%	19,1%
Other Plastics Flexibles Textile	46,7%	37,3%	28,0%	53,3%	62,7%	72,0%
Other Plastics Flexibles Nets	7,9%	6,3%	4,8%	92,1%	93,7%	95,2%
Other Plastics Flexibles Foamed	6,0%	4,8%	3,6%	94,0%	95,2%	96,4%
Paper Print Cardboard	8,8%	7,1%	5,3%	91,2%	92,9%	94,7%
Paper Tissue	17,5%	14,0%	10,5%	82,5%	86,0%	89,5%
Compound	66,6%	53,3%	40,0%	33,4%	46,7%	60,0%
Clogged	29,4%	23,5%	17,7%	70,6%	76,5%	82,3%
Others	16,4%	13,1%	9,9%	83,6%	86,9%	90,1%
Fine Fraction	13,8%	11,0%	8,3%	86,2%	89,0%	91,7%

Table C.17 The ± 25% values of separation efficiencies of NIR PP Cleaner.

	NIR PP Cleaner (targeting all PP)					
	To PP Film Bale			To NIR PO Cleaner		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	24,9%	20,0%	15,0%	75,1%	80,0%	85,0%
PE Film Transparent Clear Printed	4,4%	3,6%	2,7%	95,6%	96,4%	97,3%
PE Film Transparent Coloured	23,6%	18,9%	14,2%	76,4%	81,1%	85,8%
PE Film Opaque Coloured	13,9%	11,1%	8,3%	86,1%	88,9%	91,7%
PE Film Black	80,5%	64,4%	48,3%	19,5%	35,6%	51,7%
PE Film Metalized	0,0%	0,0%	2,5%	100,0%	100,0%	97,5%
PP Film Transparent Clear	68,2%	90,9%	100,0%	31,8%	9,1%	0,0%
PP Film Transparent Clear Printed	65,4%	87,1%	100,0%	34,6%	12,9%	0,0%
PP Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Opaque Coloured	67,6%	90,2%	100,0%	32,4%	9,8%	0,0%
PP Film Black	43,4%	57,9%	72,3%	56,6%	42,1%	27,7%
PP Film Metalized	39,4%	52,5%	65,6%	60,6%	47,5%	34,4%
Other Plastic Film	69,0%	55,2%	41,4%	31,0%	44,8%	58,6%
Other Plastic Metalized	69,0%	55,2%	41,4%	31,0%	44,8%	58,6%
Multilayer Flexibles Aluminum	83,0%	66,4%	49,8%	17,0%	33,6%	50,2%
Multilayer Flexibles Paper	54,4%	43,5%	32,6%	45,6%	56,5%	67,4%
Multilayer Flexibles Other	67,4%	53,9%	40,5%	32,6%	46,1%	59,5%
PE Rigid	5,7%	4,6%	3,4%	94,3%	95,4%	96,6%
PP Rigid	73,4%	97,8%	100,0%	26,6%	2,2%	0,0%
Other Plastics Flexibles Textile	100,0%	91,6%	68,7%	0,0%	8,4%	31,3%
Other Plastics Flexibles Nets	78,0%	62,4%	46,8%	22,0%	37,6%	53,2%
Other Plastics Flexibles Foamed	64,8%	51,8%	38,9%	35,2%	48,2%	61,1%
Paper Print Cardboard	53,6%	42,9%	32,1%	46,4%	57,1%	67,9%
Paper Tissue	42,2%	33,8%	25,3%	57,8%	66,2%	74,7%
Compound	65,3%	52,2%	39,2%	34,7%	47,8%	60,8%
Clogged	93,7%	75,0%	56,2%	6,3%	25,0%	43,8%
Others	73,5%	58,8%	44,1%	26,5%	41,2%	55,9%
Fine Fraction	72,2%	57,8%	43,3%	27,8%	42,2%	56,7%

Table C.18 The ± 25% values of separation efficiencies of NIR PO Cleaner.

	NIR PO Cleaner (targeting all PO)					
	To PO New Bale			To Residue		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	73,1%	97,4%	100,0%	26,9%	2,6%	0,0%
PE Film Transparent Clear Printed	74,1%	98,8%	100,0%	25,9%	1,2%	0,0%
PE Film Transparent Coloured	71,6%	95,4%	100,0%	28,4%	4,6%	0,0%
PE Film Opaque Coloured	72,9%	97,2%	100,0%	27,1%	2,8%	0,0%
PE Film Black	71,5%	95,3%	100,0%	28,5%	4,7%	0,0%
PE Film Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Clear	69,6%	92,8%	100,0%	30,4%	7,2%	0,0%
PP Film Transparent Clear Printed	68,9%	91,9%	100,0%	31,1%	8,1%	0,0%
PP Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Opaque Coloured	71,4%	95,3%	100,0%	28,6%	4,7%	0,0%
PP Film Black	0,0%	0,0%	2,5%	100,0%	100,0%	97,5%
PP Film Metalized	68,5%	91,3%	100,0%	31,5%	8,7%	0,0%
Other Plastic Film	67,2%	53,7%	40,3%	32,8%	46,3%	59,7%
Other Plastic Metalized	67,2%	53,7%	40,3%	32,8%	46,3%	59,7%
Multilayer Flexibles Aluminum	44,4%	35,5%	26,6%	55,6%	64,5%	73,4%
Multilayer Flexibles Paper	47,7%	38,2%	28,6%	52,3%	61,8%	71,4%
Multilayer Flexibles Other	100,0%	96,0%	72,0%	0,0%	4,0%	28,0%
PE Rigid	74,9%	99,9%	100,0%	25,1%	0,1%	0,0%
PP Rigid	72,8%	97,1%	100,0%	27,2%	2,9%	0,0%
Other Plastics Flexibles Textile	40,2%	32,1%	24,1%	59,8%	67,9%	75,9%
Other Plastics Flexibles Nets	100,0%	98,0%	73,5%	0,0%	2,0%	26,5%
Other Plastics Flexibles Foamed	65,2%	52,2%	39,1%	34,8%	47,8%	60,9%
Paper Print Cardboard	41,4%	33,1%	24,8%	58,6%	66,9%	75,2%
Paper Tissue	27,6%	22,1%	16,5%	72,4%	77,9%	83,5%
Compound	57,4%	45,9%	34,4%	42,6%	54,1%	65,6%
Clogged	100,0%	87,9%	65,9%	0,0%	12,1%	34,1%
Others	40,8%	32,6%	24,5%	59,2%	67,4%	75,5%
Fine Fraction	100,0%	83,9%	62,9%	0,0%	16,1%	37,1%

Table C.19 The ± 25% values of separation efficiencies of Cold Washing.

	Cold Washing (targeting all plastics)					
	To Density Separation/Hot Wa:			To Residue		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Transparent Clear Printed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Opaque Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Black	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Clear	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Clear Printed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Opaque Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Black	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastic Film	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastic Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Aluminum	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Paper	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Other	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Rigid	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Rigid	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Textile	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Nets	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Foamed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Paper Print Cardboard	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Paper Tissue	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Compound	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Clogged	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Others	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Fine Fraction	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%

Table C.20 The ± 25% values of separation efficiencies of Hot Washing.

	Hot Washing (targeting all plastics)					
	To Density Separation			To Residue		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Transparent Clear Printed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Opaque Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Black	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Film Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Clear	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Clear Printed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Transparent Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Opaque Coloured	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Black	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Film Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastic Film	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastic Metalized	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Aluminum	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Paper	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Multilayer Flexibles Other	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PE Rigid	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
PP Rigid	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Textile	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Nets	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Other Plastics Flexibles Foamed	75,0%	100,0%	100,0%	25,0%	0,0%	0,0%
Paper Print Cardboard	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Paper Tissue	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Compound	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Clogged	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Others	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%
Fine Fraction	6,3%	5,0%	3,75%	93,8%	0,0%	96,3%

Table C.21 The ± 25% values of separation efficiencies of Density Separation.

	Density Separation (targeting PO)					
	To Drying			To Residue		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	73,5%	98,0%	100,0%	26,5%	2,0%	0,0%
PE Film Transparent Clear Printed	73,5%	98,0%	100,0%	26,5%	2,0%	0,0%
PE Film Transparent Coloured	73,5%	98,0%	100,0%	26,5%	2,0%	0,0%
PE Film Opaque Coloured	73,5%	98,0%	100,0%	26,5%	2,0%	0,0%
PE Film Black	73,5%	98,0%	100,0%	26,5%	2,0%	0,0%
PE Film Metalized	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
PP Film Transparent Clear	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Film Transparent Clear Printed	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Film Transparent Coloured	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Film Opaque Coloured	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Film Black	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Film Metalized	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Other Plastic Film	62,5%	50,0%	37,5%	37,5%	50,0%	62,5%
Other Plastic Metalized	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Multilayer Flexibles Aluminum	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Multilayer Flexibles Paper	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Multilayer Flexibles Other	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
PE Rigid	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
PP Rigid	74,3%	99,0%	100,0%	25,8%	1,0%	0,0%
Other Plastics Flexibles Textile	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Other Plastics Flexibles Nets	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Other Plastics Flexibles Foamed	37,5%	30,0%	22,5%	62,5%	70,0%	77,5%
Paper Print Cardboard	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Paper Tissue	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Compound	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Clogged	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Others	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Fine Fraction	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%

Table C.22 The ± 25% values of separation efficiencies of Extrusion.

	Extrusion (targeting PO)					
	To Drying			To Residue		
	(-) 25%	Central	(+) 25%	(-) 25%	Central	(+) 25%
PE Film Transparent Clear	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PE Film Transparent Clear Printed	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PE Film Transparent Coloured	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PE Film Opaque Coloured	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PE Film Black	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PE Film Metalized	100,0%	90,0%	67,5%	0,0%	10,0%	32,5%
PP Film Transparent Clear	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Film Transparent Clear Printed	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Film Transparent Coloured	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Film Opaque Coloured	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Film Black	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Film Metalized	100,0%	90,0%	67,5%	0,0%	10,0%	32,5%
Other Plastic Film	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Other Plastic Metalized	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Multilayer Flexibles Aluminum	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Multilayer Flexibles Paper	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Multilayer Flexibles Other	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
PE Rigid	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
PP Rigid	72,8%	97,0%	100,0%	27,3%	3,0%	0,0%
Other Plastics Flexibles Textile	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Other Plastics Flexibles Nets	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Other Plastics Flexibles Foamed	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Paper Print Cardboard	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Paper Tissue	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Compound	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Clogged	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Others	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%
Fine Fraction	6,3%	5,0%	3,8%	93,8%	95,0%	96,3%

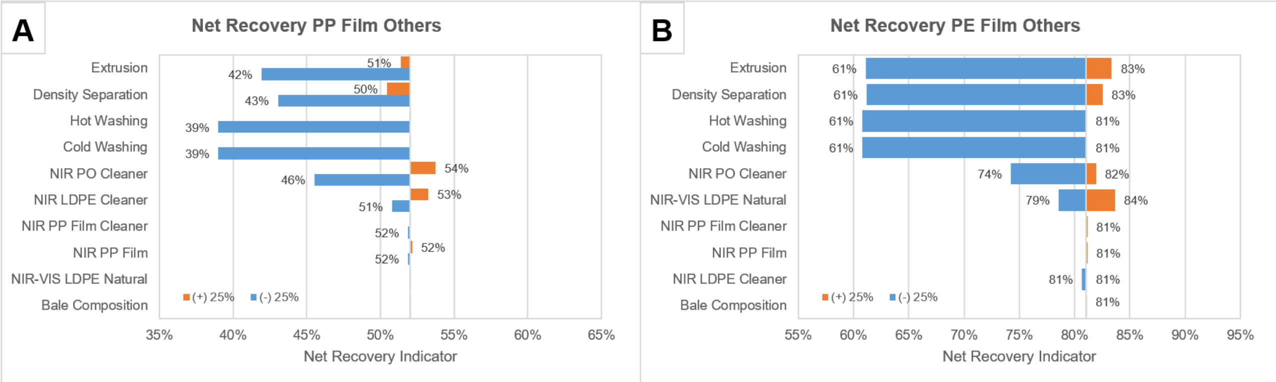


Figure C.5 The results of sensitivity analysis towards the net recovery of PP Film Others (A) and net recovery of PE Film Others (B).

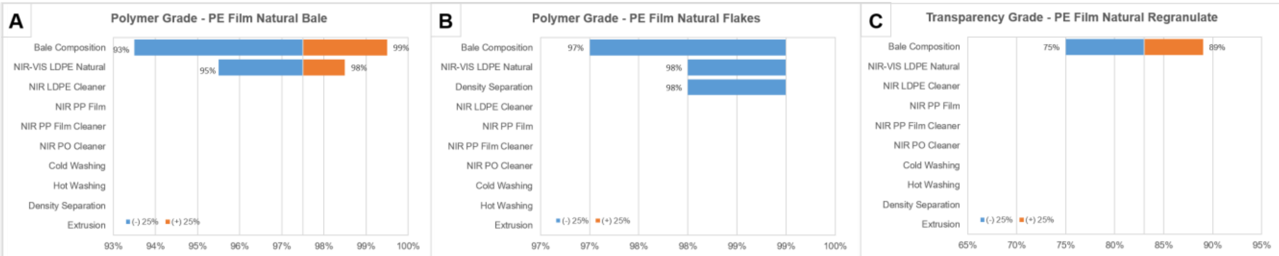


Figure C.6 The results of sensitivity analysis towards the Polymer Grade of PE Film Natural at bale (A) and flakes (B), including the Transparency Grade at regranulates level (C).

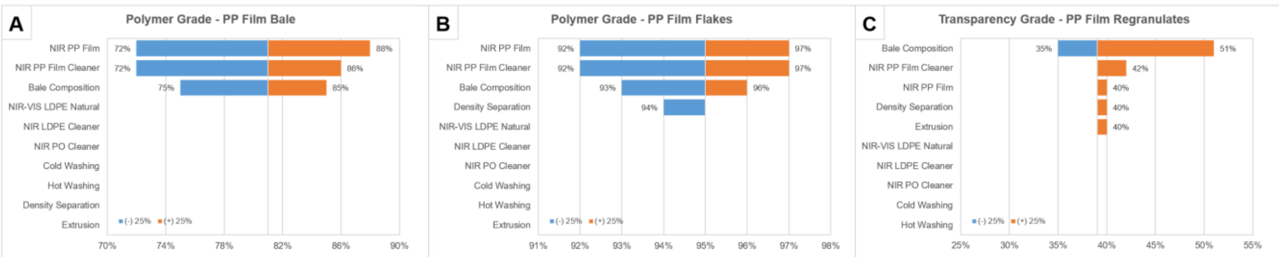


Figure C.7 The results of sensitivity analysis towards the Polymer Grade of PP Film at bale (A) and flakes (B), including the Transparency Grade at regranulates level (C).

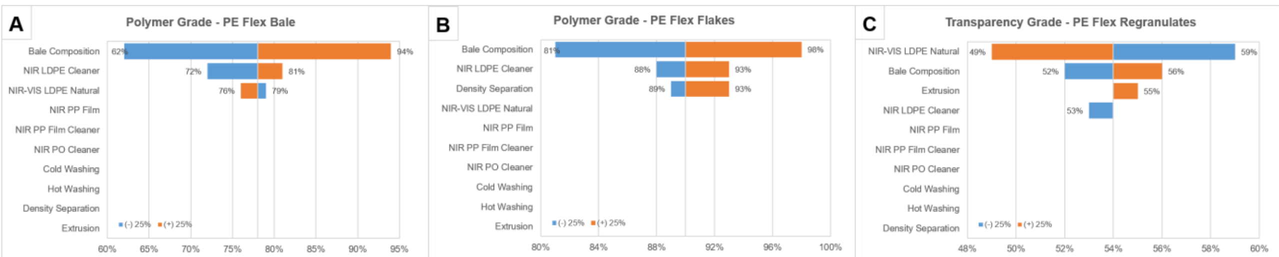


Figure C.8 The results of sensitivity analysis towards the Polymer Grade of PE Flex at bale (A) and flakes (B), including the Transparency Grade at regranulates level (C).

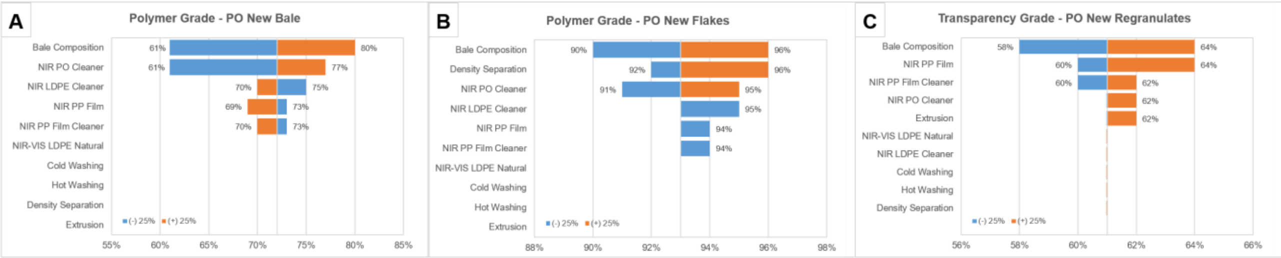


Figure C.9 The results of sensitivity analysis towards the Polymer Grade of PO New at bale (A) and flakes (B), including the Transparency Grade at regranulates level (C). The polymer grade at regranulates level is excluded from the sensitivity results because the PO New regranulates are still expected to consist of 100% polyolefin based on the MFA model in all cases.

SECTION 7: ECONOMIC MODELING PARAMETERS

Table C.23 Plant section, processing type, and equipment category based on the flowsheet in the main text – Improved Mechanical Recycling: Case study of QRP with Tier 1 procedure.

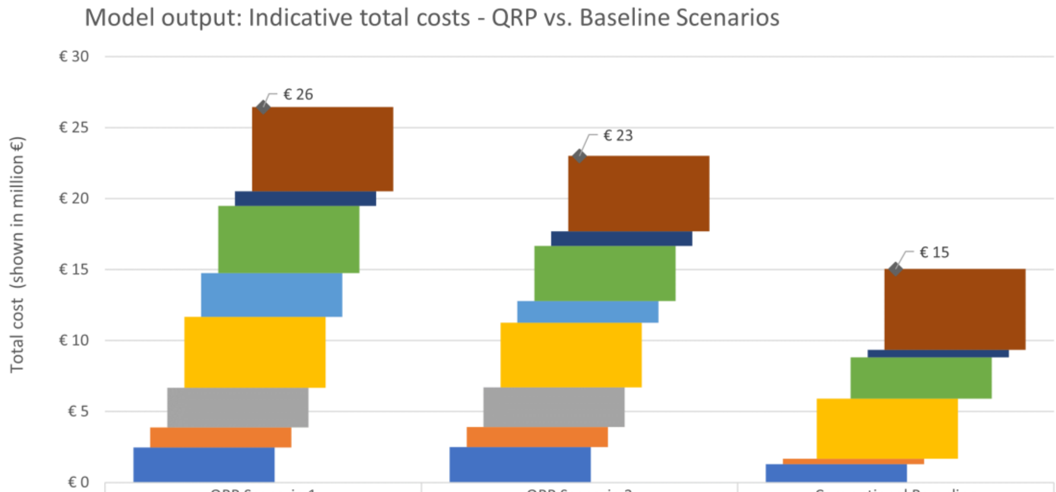
Plant section	**Equipment category**	**Sorting or Recycling Equipment**	**Number of equipment**		
			Improved Mechanical Recycling	Conventional Mechanical Recycling	
			Case study: QRP	DSD 310-1	DSD 323-2
Feeding and conditioning for sorting and recycling	Mobile equipment, processing equipment	Wheel loader	2	1	1
		Debaler	2	1	1
		Dosing unit	2	-	-
		Overbelt Magnet	4	2	-
		Fine screening	4	-	-
		Shredder	4	2	2
		Conveyor belt	50	2	2
Sorting	Processing Equipment	NIR sorter	11	2	-
Washing	Processing Equipment	Classical Washing	4	2	2
		Hot Washing	4	-	-
		Density Separation	4	2	2
		Mechanical and Thermal Dryer	8	2	2
Regranulation	Processing Equipment	Single filter extruder	0	2	2
		Double filter extruder	4	-	-
		Degassing	4	-	-
		Deodorization	4	-	-
Bale and final product handling	Mobile equipment, Processing equipment	Forklift	4	2	2
		Baler	2	-	-
		Sack/Bar Carrier	2	1	1

Table C.24 Plant section, processing type, and equipment category based on the flowsheet in the main text – Improved Mechanical Recycling: Case study of QRP with Tier 1 and Tier 2 procedures.

Plant section	**Equipment category**	**Sorting or Recycling Equipment**	**Number of equipment**		
			Improved Mechanical Recycling	'Conventional' Mechanical Recycling	
			Case study: QRP	DSD 310-1	DSD 323-2
Feeding and conditioning for sorting and recycling	Mobile equipment, processing equipment	Wheel loader	2	1	1
		Debaler	2	1	1
		Dosing unit	2	-	-
		Overbelt Magnet	4	2	-
		Fine screening	4	-	-
		Shredder	4	2	2
		Conveyor belt	50	2	2
Sorting	Processing Equipment	NIR sorter	11	2	-
Washing	Processing Equipment	Classical Washing	4	2	2
		Hot Washing	2	-	-
		Density Separation	4	2	2
		Mechanical and Thermal Dryer	6	2	2
Regranulation	Processing Equipment	Single filter extruder	2	2	2
		Double filter extruder	2	-	-
		Degassing	2	-	-
		Deodorization	2	-	-
Bale and final product handling	Mobile equipment, Processing equipment	Forklift	4	2	2
		Baler	2	-	-
		Sack/Bar Carrier	2	1	1

Table C.25 The central, upper and lower values (± 25%) used in the sensitivity analysis of the net income/loss of QRP scenario 1 and scenario 2.

Cost-benefit analysis parameter	Lower value (-25%)	Central value	Upper value (+25%)
Price of rPE Film Natural	€ 900/ton	€ 1200/ton	€ 1500/ton
Price of T1-rPE Flex	€ 375/ton	€ 500/ton	€ 625/ton
Price of T2-rPE Flex	€ 300/ton	€ 400/ton	€ 500/ton
Price of rPP Film	€ 975/ton	€ 1300/ton	€ 1625/ton
Price of T1-rPO New	€ 300/ton	€ 400/ton	€ 500/ton
Price of T2-rPO New	€ 225/ton	€ 300/ton	€ 375/ton
Price of electricity	€ 0.08/kWh	€ 0.10/kWh	€ 0.13/kWh
Depreciation rate	11%	15%	19%
Labor cost	€ 33,750/person.year	€ 45,000/person.year	€ 56,250/person.year
Bale opener	€ 362,500	€ 485,000	€ 600,000
Shredder	€ 130,000	€ 176,000	€ 220,000
Cold Washing	€ 1,350,000	€ 1,804,000	€ 2,250,000
Hot Washing	€ 1,350,000	€ 1,848,000	€ 2,250,000
Single filter extruder	€ 1,350,000	€ 1,760,000	€ 2,250,000
Double filter extruder with degassing and deodorization	€ 2,000,000	€ 2,640,000	€ 3,400,000



	QRP Scenario 1	QRP Scenario 2	Conventional Recycling
Building for Recycling	€ 6	€ 5	€ 6
Bale and final product handling	€ 1	€ 1	€ 1
Regranulation	€ 5	€ 4	€ 3
Hot washing	€ 3	€ 2	€ 0
Cold Washing	€ 5	€ 5	€ 4
Building for additional sorting QRP	€ 3	€ 3	€ 0
Sorting	€ 1	€ 1	€ 0
Feeding, conditioning, and shredding	€ 2	€ 2	€ 1
◆ Total	€ 26	€ 23	€ 15

Figure C.10 Total cost (OPEX) needed for QRP scenario 1, QRP scenario 2, and conventional recycling 310-1 and 323-2 bales. The cost is shown in million € and broken down per plant section or recycling equipment. In QRP scenario 1, all fractions are processed through Tier 1. In QRP scenario 2, PE Film natural and PP Film are processed through Tier 1 and PE Flex and PO New are processed through Tier 2.

D

APPENDIX D. COST-BENEFIT ANALYSIS OF COLLECTING AND RECYCLING NON-HOUSEHOLD END-USE PLASTIC FILM WASTE FROM URBAN AREAS

SECTION 1: PRIMARY DATA POINTS ON THE TOTAL COLLECTED NON-HOUSEHOLD END-USE PLASTIC FILM WASTE FROM WASTE SAMPLING AND TOTAL LISTED COMPANIES IN THE URBAN AREAS OF GHENT – BELGIUM (POSTCODE: 9000 – 9070)

Continued on the next pages.

Table D.1 Data on the non-household end-use plastics film waste collected during waste sampling from urban areas of Ghent–Belgium in 2018. The waste quantity data (in tonne) is shown for different NACE sectors, incl. the number of companies participated in the sampling campaign. The table also provides the total active companies based on the Orbis (2022). *Units*: Waste quantity (*tonne*), Waste generated per company (*tonne/year.company*), Total waste generated (*tonne/year.NACE sector*). For more information, readers are advised to read section 5.2.3 in the main text.

	Data sources			
	Dataset collected during sampling in 2018			*Data from Orbis (2022)*
NACE sectors, names, and codes	*Waste quantity*	*Number of companies*	*Waste generated per company*	*Total active companies*
A – Agriculture, forestry and fishing				
A01 - Crop and animal production, hunting and related service activities	3.38	47	0.07	333
A02 - Forestry and logging	0.00	1	0.00	16
B – Mining and quarrying				
B08 - Other mining and quarrying	0.54	7	0.08	7
C – Manufacturing				
C10 - Manufacture of food products	23.88	173	0.14	372
C11 - Manufacture of beverages	22.32	8	2.79	31
C13 - Manufacture of textiles	73.69	20	3.68	98
C14 - Manufacture of wearing apparel	1.94	8	0.24	82
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	0.92	19	0.05	120
C17 - Manufacture of paper and paper products	20.18	20	1.01	16
C18 - Printing and reproduction of recorded media	60.87	44	1.38	511
C19 - Manufacture of coke and refined petroleum products	0.00	14	0.00	2
C20 - Manufacture of chemicals and chemical products	111.58	69	1.60	59
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	11.95	14	0.85	46
C22 - Manufacture of rubber and plastic products	36.18	20	1.81	32
C23 - Manufacture of other non-metallic mineral products	20.35	42	0.48	77
C24 - Manufacture of basic metals	50.86	16	3.18	21

C25 - Manufacture of fabricated metal products, except machinery and equipment	2.92	52	0.06	203
C26 - Manufacture of computer, electronic and optical products	3.78	11	0.34	41
C27 - Manufacture of electrical equipment	39.29	21	1.87	39
C28 - Manufacture of machinery and equipment n.e.c.	1.24	26	0.05	54
C29 - Manufacture of motor vehicles, trailers and semi-trailers	366.88	28	13.14	24
C30 - Manufacture of other transport equipment	1.20	3	0.40	21
C31 - Manufacture of furniture	0.34	22	0.02	134
C32 - Other manufacturing	0.00	18	0.00	142
C33 - Repair and installation of machinery and equipment	0.48	23	0.02	125
D – Electricity, gas, steam and air conditioning supply				
D35 - Electricity, gas, steam and air conditioning supply	0.65	15	0.04	82
E – Water supply; sewerage; waste management and remediation activities				
E36 - Water collection, treatment and supply	0.00	26	0.00	11
E37 - Sewerage	0.00	10	0.00	10
E38 - Waste collection, treatment and disposal activities; materials recovery	36.59	41	0.89	53
E39 - Remediation activities and other waste management services	0.00	6	0.00	6
F – Construction				
F41 - Construction of buildings	4.93	189	0.03	718
F42 - Civil engineering	0.02	62	0.00	168
F43 - Specialized construction activities	20.91	507	0.04	1,847
G – Wholesale and retail trade; repair of motor vehicles and motorcycles				
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	57.50	261	0.22	484
G46 - Wholesale trade, except of motor vehicles and motorcycles	399.64	564	0.71	2,128
G47 - Retail trade, except of motor vehicles and motorcycles	428.88	1,065	0.40	3,386
Total from NACE A – G,	*1,803.87*	*3,472*	-	*11,498*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*1,608.57*	*3,286*	-	*11,281*

SECTION 2: TOTAL ESTIMATED QUANTITY OF NON-HOUSEHOLD END-USE PLASTIC FILM WASTE GENERATION FROM URBAN AREAS OF GHENT – BELGIUM (POSTCODE: 9000 – 9070)

Table D.2 Total (annual) non-household end-use plastic films waste generation from urban areas of Ghent – Belgium (postal code 9000 – 9070), which extrapolated by multiplying data collected during waste sampling in 2018 (average waste generated per company, in tonne/year.company) with total listed active companies in Table D.1. For more information, readers are advised to read section 5.2.3 in the main text.

NACE Codes and Names per Sector	*Total waste generated (after extrapolation)*
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	23.94
A02 - Forestry and logging	0.00
B – Mining and quarrying	
B08 - Other mining and quarrying	0.46
C – Manufacturing	
C10 - Manufacture of food products	51.36
C11 - Manufacture of beverages	86.49
C13 - Manufacture of textiles	361.06
C14 - Manufacture of wearing apparel	19.83
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	5.81
C17 - Manufacture of paper and paper products	16.14
C18 - Printing and reproduction of recorded media	706.93
C19 - Manufacture of coke and refined petroleum products	0.00
C20 - Manufacture of chemicals and chemical products	94.55
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	39.26
C22 - Manufacture of rubber and plastic products	57.88
C23 - Manufacture of other non-metallic mineral products	37.31
C24 - Manufacture of basic metals	66.75
C25 - Manufacture of fabricated metal products, except machinery and equipment	11.38
C26 - Manufacture of computer, electronic and optical products	14.09

C27 - Manufacture of electrical equipment	72.97
C28 - Manufacture of machinery and equipment n.e.c.	2.57
C29 - Manufacture of motor vehicles, trailers and semi-trailers	315.33
C30 - Manufacture of other transport equipment	8.40
C31 - Manufacture of furniture	2.06
C32 - Other manufacturing	0.00
C33 - Repair and installation of machinery and equipment	2.61
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	3.54
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0.00
E37 - Sewerage	0.00
E38 - Waste collection, treatment and disposal activities; materials recovery	47.30
E39 - Remediation activities and other waste management services	0.00
F – Construction	
F41 - Construction of buildings	18.72
F42 - Civil engineering	0.06
F43 - Specialized construction activities	76.18
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	106.63
G46 - Wholesale trade, except of motor vehicles and motorcycles	1,507.85
G47 - Retail trade, except of motor vehicles and motorcycles	1,353.55
Total from NACE A – G	*5,121.04*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*4,882.04*

SECTION 2: DATA ON THE LISTED ACTIVE COMPANIES (NACE SECTOR A – G) FROM ORBIS (2022) IN SINT-MARTENS-LATEM, MELLE, ZELZATE, WETTEREN, MERELBEKE, DE PINTE, LOKEREN, DEINZE, NAZARETH, LOCHRISTI, EVERGEM, AND EEKLO

Data on active companies from Orbis (2022) that is used to estimate the total quantity of non-household end-use film waste generation per NACE sector in their respective municipality. It is estimated by quantifying (average) film waste generated per company (in tonne/year.company) by number of active companies (after discounted by 20%) (section 5.2.3 in main text). The (average) film waste generated for *NACE sector A03* is assumed to be identical to NACE sector A01 data. The (average) film waste generated for *NACE sector B06, B07,* and *B09* is assumed to be identical to NACE sector B08 data. The (average) film waste generated for *NACE sector C15* is assumed to be identical to NACE sector C16 data.

Table D.3 Total listed active companies (NACE A – G) from Orbis (2022), per municipality.

Sint-Martens-Latem

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	13
A02 - Forestry and logging	0
A03 – Fishing and aquaculture	2
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	11
C11 - Manufacture of beverages	2

C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	6
C14 - Manufacture of wearing apparel	3
C15 – Manufacture of leather and related products	0
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	3
C17 - Manufacture of paper and paper products	1
C18 - Printing and reproduction of recorded media	8
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	2
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	1
C22 - Manufacture of rubber and plastic products	0
C23 - Manufacture of other non-metallic mineral products	4
C24 - Manufacture of basic metals	0
C25 - Manufacture of fabricated metal products, except machinery and equipment	10
C26 - Manufacture of computer, electronic and optical products	3
C27 - Manufacture of electrical equipment	2
C28 - Manufacture of machinery and equipment n.e.c.	7
C29 - Manufacture of motor vehicles, trailers and semi-trailers	2
C30 - Manufacture of other transport equipment	0
C31 - Manufacture of furniture	6
C32 - Other manufacturing	8
C33 - Repair and installation of machinery and equipment	3
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	1
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	0
E38 - Waste collection, treatment and disposal activities; materials recovery	1
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	70
F42 - Civil engineering	3

F43 - Specialized construction activities	67
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	34
G46 - Wholesale trade, except of motor vehicles and motorcycles	156
G47 - Retail trade, except of motor vehicles and motorcycles	132
Total from NACE A – G	*561*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*557*

Melle

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	33
A02 - Forestry and logging	2
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	1
C – Manufacturing	
C10 - Manufacture of food products	11
C11 - Manufacture of beverages	1
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	5
C14 - Manufacture of wearing apparel	6
C15 – Manufacture of leather and related products	3
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	4
C17 - Manufacture of paper and paper products	1
C18 - Printing and reproduction of recorded media	14
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	0

C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	1
C22 - Manufacture of rubber and plastic products	2
C23 - Manufacture of other non-metallic mineral products	5
C24 - Manufacture of basic metals	0
C25 - Manufacture of fabricated metal products, except machinery and equipment	21
C26 - Manufacture of computer, electronic and optical products	1
C27 - Manufacture of electrical equipment	5
C28 - Manufacture of machinery and equipment n.e.c.	5
C29 - Manufacture of motor vehicles, trailers and semi-trailers	1
C30 - Manufacture of other transport equipment	2
C31 - Manufacture of furniture	9
C32 - Other manufacturing	7
C33 - Repair and installation of machinery and equipment	12
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	5
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	2
E38 - Waste collection, treatment and disposal activities; materials recovery	5
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	79
F42 - Civil engineering	12
F43 - Specialized construction activities	120
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	33
G46 - Wholesale trade, except of motor vehicles and motorcycles	114
G47 - Retail trade, except of motor vehicles and motorcycles	119
Total from NACE A – G	*641*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*631*

Zelzate

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	40
A02 - Forestry and logging	0
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	19
C11 - Manufacture of beverages	1
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	7
C14 - Manufacture of wearing apparel	0
C15 – Manufacture of leather and related products	0
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	3
C17 - Manufacture of paper and paper products	0
C18 - Printing and reproduction of recorded media	4
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	2
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	3
C22 - Manufacture of rubber and plastic products	0
C23 - Manufacture of other non-metallic mineral products	3
C24 - Manufacture of basic metals	2
C25 - Manufacture of fabricated metal products, except machinery and equipment	20
C26 - Manufacture of computer, electronic and optical products	2
C27 - Manufacture of electrical equipment	0
C28 - Manufacture of machinery and equipment n.e.c.	2
C29 - Manufacture of motor vehicles, trailers and semi-trailers	0
C30 - Manufacture of other transport equipment	1

C31 - Manufacture of furniture	7
C32 - Other manufacturing	2
C33 - Repair and installation of machinery and equipment	16
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	5
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	1
E38 - Waste collection, treatment and disposal activities; materials recovery	3
E39 - Remediation activities and other waste management services	1
F – Construction	
F41 - Construction of buildings	33
F42 - Civil engineering	7
F43 - Specialized construction activities	118
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	40
G46 - Wholesale trade, except of motor vehicles and motorcycles	66
G47 - Retail trade, except of motor vehicles and motorcycles	123
Total from NACE A – G	*531*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*521*

Wetteren

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	121
A02 - Forestry and logging	3
A03 – Fishing and aquaculture	1
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	2

C – Manufacturing	
C10 - Manufacture of food products	40
C11 - Manufacture of beverages	7
C12 – Manufacture of tobacco products	1
C13 - Manufacture of textiles	7
C14 - Manufacture of wearing apparel	4
C15 – Manufacture of leather and related products	1
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	13
C17 - Manufacture of paper and paper products	5
C18 - Printing and reproduction of recorded media	27
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	4
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	2
C22 - Manufacture of rubber and plastic products	8
C23 - Manufacture of other non-metallic mineral products	3
C24 - Manufacture of basic metals	1
C25 - Manufacture of fabricated metal products, except machinery and equipment	45
C26 - Manufacture of computer, electronic and optical products	6
C27 - Manufacture of electrical equipment	10
C28 - Manufacture of machinery and equipment n.e.c.	8
C29 - Manufacture of motor vehicles, trailers and semi-trailers	5
C30 - Manufacture of other transport equipment	2
C31 - Manufacture of furniture	11
C32 - Other manufacturing	10
C33 - Repair and installation of machinery and equipment	27
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	2
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	1
E37 - Sewerage	3
E38 - Waste collection, treatment and disposal activities; materials recovery	9
E39 - Remediation activities and other waste management services	4

F – Construction	
F41 - Construction of buildings	103
F42 - Civil engineering	17
F43 - Specialized construction activities	286
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	86
G46 - Wholesale trade, except of motor vehicles and motorcycles	196
G47 - Retail trade, except of motor vehicles and motorcycles	329
Total from NACE A – G	*1,410*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*1,379*

Merelbeke

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	77
A02 - Forestry and logging	2
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	1
C – Manufacturing	
C10 - Manufacture of food products	26
C11 - Manufacture of beverages	3
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	9
C14 - Manufacture of wearing apparel	5
C15 – Manufacture of leather and related products	0
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	17
C17 - Manufacture of paper and paper products	1

C18 - Printing and reproduction of recorded media	24
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	5
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	12
C22 - Manufacture of rubber and plastic products	1
C23 - Manufacture of other non-metallic mineral products	11
C24 - Manufacture of basic metals	1
C25 - Manufacture of fabricated metal products, except machinery and equipment	41
C26 - Manufacture of computer, electronic and optical products	4
C27 - Manufacture of electrical equipment	9
C28 - Manufacture of machinery and equipment n.e.c.	9
C29 - Manufacture of motor vehicles, trailers and semi-trailers	5
C30 - Manufacture of other transport equipment	2
C31 - Manufacture of furniture	11
C32 - Other manufacturing	6
C33 - Repair and installation of machinery and equipment	21
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	6
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	1
E38 - Waste collection, treatment and disposal activities; materials recovery	2
E39 - Remediation activities and other waste management services	1
F – Construction	
F41 - Construction of buildings	99
F42 - Civil engineering	19
F43 - Specialized construction activities	252
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	58
G46 - Wholesale trade, except of motor vehicles and motorcycles	209
G47 - Retail trade, except of motor vehicles and motorcycles	207
Total from NACE A – G	*1,157*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*1,135*

De Pinte

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	19
A02 - Forestry and logging	0
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	10
C11 - Manufacture of beverages	1
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	2
C14 - Manufacture of wearing apparel	1
C15 – Manufacture of leather and related products	0
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	7
C17 - Manufacture of paper and paper products	1
C18 - Printing and reproduction of recorded media	7
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	0
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	1
C22 - Manufacture of rubber and plastic products	2
C23 - Manufacture of other non-metallic mineral products	1
C24 - Manufacture of basic metals	0
C25 - Manufacture of fabricated metal products, except machinery and equipment	11
C26 - Manufacture of computer, electronic and optical products	4
C27 - Manufacture of electrical equipment	1

C28 - Manufacture of machinery and equipment n.e.c.	3
C29 - Manufacture of motor vehicles, trailers and semi-trailers	0
C30 - Manufacture of other transport equipment	0
C31 - Manufacture of furniture	2
C32 - Other manufacturing	10
C33 - Repair and installation of machinery and equipment	3
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	0
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	1
E37 - Sewerage	0
E38 - Waste collection, treatment and disposal activities; materials recovery	2
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	25
F42 - Civil engineering	4
F43 - Specialized construction activities	62
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	9
G46 - Wholesale trade, except of motor vehicles and motorcycles	79
G47 - Retail trade, except of motor vehicles and motorcycles	88
Total from NACE A – G	*356*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*350*

Lokeren

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	170
A02 - Forestry and logging	2
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0

B07 – Mining of metal ores	0
B08 - Other mining and quarrying	1
B09 – Mining support service activities	2
C – Manufacturing	
C10 - Manufacture of food products	84
C11 - Manufacture of beverages	2
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	15
C14 - Manufacture of wearing apparel	5
C15 – Manufacture of leather and related products	3
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	31
C17 - Manufacture of paper and paper products	4
C18 - Printing and reproduction of recorded media	29
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	10
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	4
C22 - Manufacture of rubber and plastic products	12
C23 - Manufacture of other non-metallic mineral products	9
C24 - Manufacture of basic metals	4
C25 - Manufacture of fabricated metal products, except machinery and equipment	83
C26 - Manufacture of computer, electronic and optical products	9
C27 - Manufacture of electrical equipment	15
C28 - Manufacture of machinery and equipment n.e.c.	18
C29 - Manufacture of motor vehicles, trailers and semi-trailers	7
C30 - Manufacture of other transport equipment	4
C31 - Manufacture of furniture	17
C32 - Other manufacturing	15
C33 - Repair and installation of machinery and equipment	43
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	11
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	1

E37 - Sewerage	7
E38 - Waste collection, treatment and disposal activities; materials recovery	18
E39 - Remediation activities and other waste management services	1
F – Construction	
F41 - Construction of buildings	119
F42 - Civil engineering	143
F43 - Specialized construction activities	501
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	126
G46 - Wholesale trade, except of motor vehicles and motorcycles	336
G47 - Retail trade, except of motor vehicles and motorcycles	412
Total from NACE A – G	*2,273*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*2,220*

Deinze

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	485
A02 - Forestry and logging	4
A03 – Fishing and aquaculture	2
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	2
B09 – Mining support service activities	1
C – Manufacturing	
C10 - Manufacture of food products	96
C11 - Manufacture of beverages	4
C12 – Manufacture of tobacco products	1
C13 - Manufacture of textiles	23
C14 - Manufacture of wearing apparel	20
C15 – Manufacture of leather and related products	3

C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	46
C17 - Manufacture of paper and paper products	3
C18 - Printing and reproduction of recorded media	60
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	8
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	12
C22 - Manufacture of rubber and plastic products	8
C23 - Manufacture of other non-metallic mineral products	30
C24 - Manufacture of basic metals	5
C25 - Manufacture of fabricated metal products, except machinery and equipment	80
C26 - Manufacture of computer, electronic and optical products	6
C27 - Manufacture of electrical equipment	10
C28 - Manufacture of machinery and equipment n.e.c.	24
C29 - Manufacture of motor vehicles, trailers and semi-trailers	5
C30 - Manufacture of other transport equipment	2
C31 - Manufacture of furniture	34
C32 - Other manufacturing	21
C33 - Repair and installation of machinery and equipment	55
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	23
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	3
E38 - Waste collection, treatment and disposal activities; materials recovery	8
E39 - Remediation activities and other waste management services	6
F – Construction	
F41 - Construction of buildings	275
F42 - Civil engineering	28
F43 - Specialized construction activities	698
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	131
G46 - Wholesale trade, except of motor vehicles and motorcycles	441

G47 - Retail trade, except of motor vehicles and motorcycles	574
Total from NACE A – G	*3,237*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*3,192*

Nazareth

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	115
A02 - Forestry and logging	0
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	1
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	15
C11 - Manufacture of beverages	2
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	7
C14 - Manufacture of wearing apparel	3
C15 – Manufacture of leather and related products	1
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	12
C17 - Manufacture of paper and paper products	1
C18 - Printing and reproduction of recorded media	17
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	3
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	6
C22 - Manufacture of rubber and plastic products	5
C23 - Manufacture of other non-metallic mineral products	2
C24 - Manufacture of basic metals	0

C25 - Manufacture of fabricated metal products, except machinery and equipment	25
C26 - Manufacture of computer, electronic and optical products	6
C27 - Manufacture of electrical equipment	6
C28 - Manufacture of machinery and equipment n.e.c.	6
C29 - Manufacture of motor vehicles, trailers and semi-trailers	5
C30 - Manufacture of other transport equipment	0
C31 - Manufacture of furniture	8
C32 - Other manufacturing	4
C33 - Repair and installation of machinery and equipment	14
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	5
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	0
E37 - Sewerage	3
E38 - Waste collection, treatment and disposal activities; materials recovery	1
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	58
F42 - Civil engineering	9
F43 - Specialized construction activities	123
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	25
G46 - Wholesale trade, except of motor vehicles and motorcycles	150
G47 - Retail trade, except of motor vehicles and motorcycles	82
Total from NACE A – G	*720*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*702*

Lochristi

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	196
A02 - Forestry and logging	1

A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	11
C11 - Manufacture of beverages	2
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	6
C14 - Manufacture of wearing apparel	7
C15 – Manufacture of leather and related products	1
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	6
C17 - Manufacture of paper and paper products	1
C18 - Printing and reproduction of recorded media	25
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	2
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	1
C22 - Manufacture of rubber and plastic products	0
C23 - Manufacture of other non-metallic mineral products	4
C24 - Manufacture of basic metals	0
C25 - Manufacture of fabricated metal products, except machinery and equipment	26
C26 - Manufacture of computer, electronic and optical products	1
C27 - Manufacture of electrical equipment	2
C28 - Manufacture of machinery and equipment n.e.c.	2
C29 - Manufacture of motor vehicles, trailers and semi-trailers	0
C30 - Manufacture of other transport equipment	0
C31 - Manufacture of furniture	4
C32 - Other manufacturing	2
C33 - Repair and installation of machinery and equipment	14
D – Electricity, gas, steam and air conditioning supply	

D35 - Electricity, gas, steam and air conditioning supply	1
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	1
E37 - Sewerage	0
E38 - Waste collection, treatment and disposal activities; materials recovery	0
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	53
F42 - Civil engineering	5
F43 - Specialized construction activities	172
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	37
G46 - Wholesale trade, except of motor vehicles and motorcycles	166
G47 - Retail trade, except of motor vehicles and motorcycles	188
Total from NACE A – G	*937*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*933*

Evergem

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	108
A02 - Forestry and logging	2
A03 – Fishing and aquaculture	0
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	1
B09 – Mining support service activities	0
C – Manufacturing	
C10 - Manufacture of food products	38
C11 - Manufacture of beverages	4
C12 – Manufacture of tobacco products	0

C13 - Manufacture of textiles	3
C14 - Manufacture of wearing apparel	2
C15 – Manufacture of leather and related products	0
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	15
C17 - Manufacture of paper and paper products	0
C18 - Printing and reproduction of recorded media	18
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	11
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	2
C22 - Manufacture of rubber and plastic products	1
C23 - Manufacture of other non-metallic mineral products	10
C24 - Manufacture of basic metals	2
C25 - Manufacture of fabricated metal products, except machinery and equipment	52
C26 - Manufacture of computer, electronic and optical products	3
C27 - Manufacture of electrical equipment	5
C28 - Manufacture of machinery and equipment n.e.c.	10
C29 - Manufacture of motor vehicles, trailers and semi-trailers	6
C30 - Manufacture of other transport equipment	1
C31 - Manufacture of furniture	13
C32 - Other manufacturing	5
C33 - Repair and installation of machinery and equipment	23
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	0
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	1
E37 - Sewerage	8
E38 - Waste collection, treatment and disposal activities; materials recovery	8
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	89
F42 - Civil engineering	10
F43 - Specialized construction activities	259

G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	58
G46 - Wholesale trade, except of motor vehicles and motorcycles	130
G47 - Retail trade, except of motor vehicles and motorcycles	149
Total from NACE A – G	*1,047*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*1,016*

Eeklo

NACE Codes and Names per Sector	Number of companies
A – Agriculture, forestry and fishing	
A01 - Crop and animal production, hunting and related service activities	112
A02 - Forestry and logging	1
A03 – Fishing and aquaculture	1
B – Mining and quarrying	
B06 – Extraction of crude petroleum and natural gas	0
B07 – Mining of metal ores	0
B08 - Other mining and quarrying	0
B09 – Mining support service activities	1
C – Manufacturing	
C10 - Manufacture of food products	43
C11 - Manufacture of beverages	3
C12 – Manufacture of tobacco products	0
C13 - Manufacture of textiles	9
C14 - Manufacture of wearing apparel	3
C15 – Manufacture of leather and related products	2
C16 - Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	19
C17 - Manufacture of paper and paper products	7
C18 - Printing and reproduction of recorded media	15
C19 - Manufacture of coke and refined petroleum products	0
C20 - Manufacture of chemicals and chemical products	3
C21 - Manufacture of basic pharmaceutical products and pharmaceutical preparations	1

C22 - Manufacture of rubber and plastic products	3
C23 - Manufacture of other non-metallic mineral products	10
C24 - Manufacture of basic metals	7
C25 - Manufacture of fabricated metal products, except machinery and equipment	68
C26 - Manufacture of computer, electronic and optical products	0
C27 - Manufacture of electrical equipment	4
C28 - Manufacture of machinery and equipment n.e.c.	7
C29 - Manufacture of motor vehicles, trailers and semi-trailers	2
C30 - Manufacture of other transport equipment	1
C31 - Manufacture of furniture	24
C32 - Other manufacturing	6
C33 - Repair and installation of machinery and equipment	18
D – Electricity, gas, steam and air conditioning supply	
D35 - Electricity, gas, steam and air conditioning supply	4
E – Water supply; sewerage; waste management and remediation activities	
E36 - Water collection, treatment and supply	2
E37 - Sewerage	3
E38 - Waste collection, treatment and disposal activities; materials recovery	17
E39 - Remediation activities and other waste management services	0
F – Construction	
F41 - Construction of buildings	92
F42 - Civil engineering	11
F43 - Specialized construction activities	237
G – Wholesale and retail trade; repair of motor vehicles and motorcycles	
G45 - Wholesale and retail trade and repair of motor vehicles and motorcycles	84
G46 - Wholesale trade, except of motor vehicles and motorcycles	156
G47 - Retail trade, except of motor vehicles and motorcycles	268
Total from NACE A – G	*1,244*
Total from NACE A – G, excluding C20, C21, C22, E36, E37, E38, and E39	*1,215*

SECTION 4: WASTE COMPOSITIONAL ANALYSES FROM THE WASTE SAMPLING CONDUCTED IN 2021 – 2022 IN URBAN AREAS OF GHENT – BELGIUM

Figure D.1 Images of the collected samples from Retail sector (e.g., NACE sector G.47) from urban areas of Ghent.

Figure D.2 Images of the collected samples from Wholesale sector (e.g., NACE sector G.46) from urban areas of Ghent.

Figure D.3 Images of the collected samples from 'Other' sectors (e.g., NACE sector C.10, C.18, etc.) from urban areas of Ghent.

Figure D.4 Images of the collected samples from logistic sectors (e.g., NACE sector H.49) from urban areas of Ghent, which is predominantly Transparent PE Film (13 kg sample).

SECTION 5: LOGISTIC SIMULATIONS DATA POINTS

Table D.4 Number of garbage trucks used in the non-household end-use plastic film waste collection from urban areas considered in this study.

Municipalities	**Collection frequencies**		
	Weekly	*Fortnightly*	*Monthly*
Ghent	49	28	16
Sint-Martens-Latem	2	1	1
Melle	2	1	1
Zelzate	2	1	1
Merelbeke	4	2	2
De Pinte	1	1	1
Lokeren	6	4	3
Nazareth	2	2	1
Deinze	7	4	3
Lochristi	3	2	1
Evergem	3	2	1
Eeklo	4	2	2
Wetteren	4	3	2

SECTION 6: MECHANICAL RECYCLING OF NON-HOUSEHOLD END-USE PLASTIC FILM DATA POINTS

Table D.5 Separation efficiency (shown in %) of each waste category (in Table 5.2) in different equipment (as shown in Figure 5.3) used in MFA modeling, averaged based on Brouwer et al. (2018), Lase et al. (2022), and Kleinhans et al. (2021).

Waste Category	**NIR PE Film Cleaner**		**Cold & Hot Washing**		**Density Separation**		**Extrusion**	
Output	*Next step*	*Residue*	*Next step*	*Residue*	*Float*	*Sink*	*Regranulate*	*Residue*
PE film transparent	86	14	98	2	89	11	96	4
PE film colored	86	14	98	2	89	11	96	4
PP film transparent	40	60	98	2	89	11	96	4
PP Film colored	40	60	98	2	89	11	96	4
Other films	40	60	98	2	30	70	5	95
Residue	60	40	10	90	5	95	5	95

Table D.6 Cost modeling parameters, capital investment per recycling equipment

Recycling equipment	**Total investment (in €)**, *incl. project management cost, installation, transportation, and site infrastructure*	**Source**
Bag opener	€ 528,000	Cimpan et al. (2016); Bashirgonbadi et al. (2022)
Shredder	€ 176,000	Bashirgonbadi et al. (2022)
NIR LDPE Cleaner (-)	€ 228,800	Cimpan et al. (2016) ; Bashirgonbadi et al. (2022)
Cold washing, including density separation and dryers	€ 1,350,000	Bashirgonbadi et al. (2022)
Hot washing, including dryers	€ 1,350,000	Bashirgonbadi et al. (2022)
Extruder	€ 1,350,000	Bashirgonbadi et al. (2022)
Handling station	€ 352,000	Cimpan et al. (2016)

Table D.7 Cost benefit analysis parameters to quantify the total investment and annual costs

Cost-benefit analysis parameters	**Value**	**Source**
Capital Investment		
Price of Equipment (PoC)	Table D.6	Bashirgonbadi et al., 2022
Additional Costs		
• Procuring, installation and running test (IC)	60% of PoC	Bashirgonbadi et al., 2022; Larrain et al. 2021; Sinnott and Towler, 2019; Cimpan et al. 2016
• Engineering and project management (EPMC)	10% of PoC and 10% of IC	
• Building construction (BC)	25% of total capital investment	
Total capital investment = PoC + IC + EPMC + BC		
Annual costs		
Labor use	1,5 person/kt processing capacity for recycling	Bashirgonbadi et al., 2022
Labor cost	64,200 €/person.year	Larrain et al. 2021 ; Larrain et al. 2020 ; STATBEL, 2019
Electricity	0.074 €/kWh	PwC, 2019; STATBEL, 2019; OVAM, 2019; Sinnott and Towler, 2019
Fuel	1,310 €/m3	
Gas	0.07€/kWh	
Water	1.2 €/m3	
Repair and maintenance	4% × Total capital investment	Bashirgonbadi et al., 2022; Larrain et al. 2021; Cimpan et al. 2016; Sinnott and Towler, 2019; OVAM, 2019
Insurance	1.5% × Total capital investment	
Depreciation	10–15% × Total capital investment	
Residual treatment	132.5 €/ton residue	
General overhead plant	10% of total cost	

Table D.8 Cost modeling parameters, energy usage per recycling equipment. The amount of NaOH used during hot washing is fixed at 2% of the total throughput (in tonne/year) into hot washing process.

Recycling equipment	**Electricity (in kWh)**	**Natural gas (in kWh)**	**Water (in m^3/tonne throughput)**	**Source**
Bale opener	100	-	-	Cimpan et al. (2016)
Shredder	160	-	-	Larrain et al. (2021)
NIR LDPE Cleaner	26	-	-	Cimpan et al. (2016)
Cold washing, incl. density separation and mechanical dryers	400	-	8.8	Larrain et al. (2021); WRAP (2009b)

Hot washing, incl. dryers	300	200	2.2	Primary data; Bashirgonbadi et al. (2022)
Extruder	75	-	-	Civancik-Uslu et al. (2021)
Handling station	100	-	-	Cimpan et al. (2016)

SECTION 7: LIFE CYCLE ASSESSMENT DATABASES USED IN THE STUDY

Table D.9 Name of datasets from Ecoinvent v3.8

Process	Ecoinvent profile	Emission factor
Waste collection	Transport, freight, lorry 16 – 32 metric ton, EURO6 {RER} | transport, freight, lorry 16-32 metric ton, EURO6 | Cut-off, U	0.165 kg CO2-eq/tkm
Mechanical recycling process		
Electricity	Electricity, medium voltage {BE}| market for | Cut-off, U	0.261 kg CO2-eq/kWh
Natural gas	Heat, district or industrial, natural gas {BE}| heat and power co-generation, natural gas, combined cycle power plant, 400MW electrical | Cut-off, U	0.078 kg CO2-eq/kWh
Fuel	Diesel, burned in building machine {GLO}| market for | Cut-off, U	0.091 kg CO2-eq/MJ
Water	Tap water {Europe without Switzerland}| market for | Cut-off, U	0.000329 kg CO2-eq/kg
Wastewater treatment	Wastewater, average {Europe without Switzerland}| treatment of wastewater, average, capacity 1E9l/year | Cut-off, U	0.49 kg CO2-eq/m^3
Detergents for hot washing	Sodium hydroxide, without water, in 50% solution state {GLO}| market for | Cut-off, U	1.29 kg CO2-eq/kg
Incineration of PE	Waste polyethylene {RoW}| treatment of waste polyethylene, municipal incineration | Cut-off, U	3.03 kg CO2-eq/kg
Incineration of PP	Waste polypropylene {RoW}| treatment of waste polypropylene, municipal incineration | Cut-off, U	2.56 kg CO2-eq/kg
Incineration of other mixed film fraction	Waste polyethylene terephthalate {RoW}| treatment of waste polyethylene terephthalate, municipal incineration | Cut-off, U	2.07 kg CO2-eq/kg
Virgin production		
Virgin PE granulate	Polyethylene, low density, granulate {RER} | production | cut-off, U	2.02 kg CO2-eq/kg

SECTION 8: LOGISTIC SIMULATION RESULTS FROM OPTIFLOW SOFTWARE

The results of waste collection simulation using OptiFlow© software are presented in Figure D.5 – Figure D.25. It is assumed that the mechanical recycling hub is located in the Port of Ghent, thus the garbage trucks leave and return to the depot during the waste collection. The OptiFlow© software can only simulate the waste collection from 2,000 addresses (data points) per simulation, hence the logistic simulation in urban areas of Ghent is divided into eight different simulations (in Figure D.6 – Figure D.13). The division of areas in the City of Ghent is based on the latitude and longitudinal coordinates. Note that Figure D.5 is a typical result obtained from the logistic simulation from OptiFlow© software (as an example from Ghent sub-area 1) containing average time span of waste collection, total distance, number of routes made, and costs (in € per km).

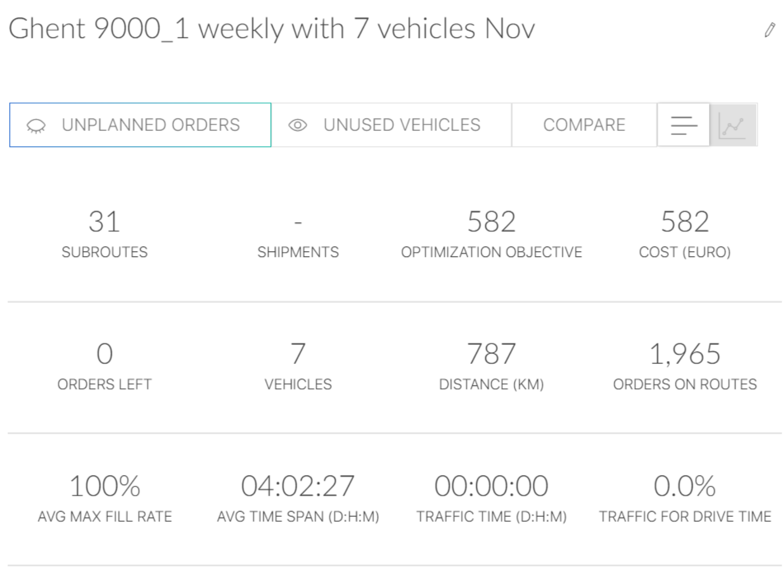
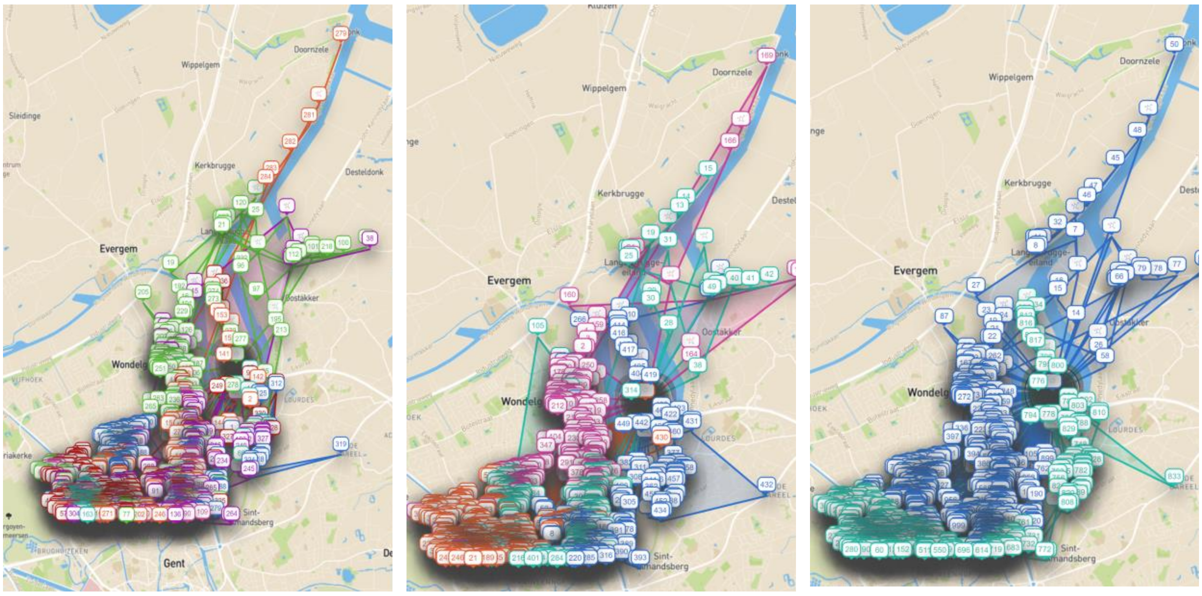
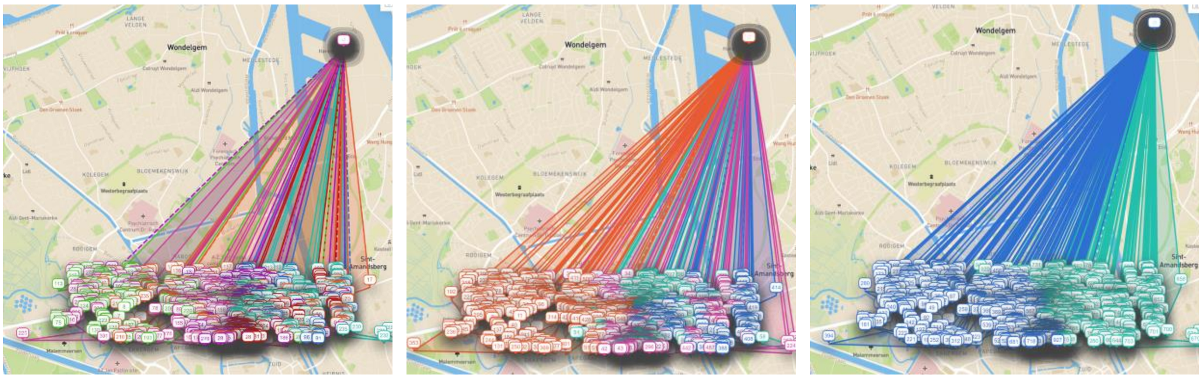


Figure D.5 A typical logistic simulation result from OptiFlow© software. As an example, the logistic simulation from Ghent (sub-area 1) is presented.



Weekly collection Fortnightly collection Monthly collection

Figure D.6 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 1.

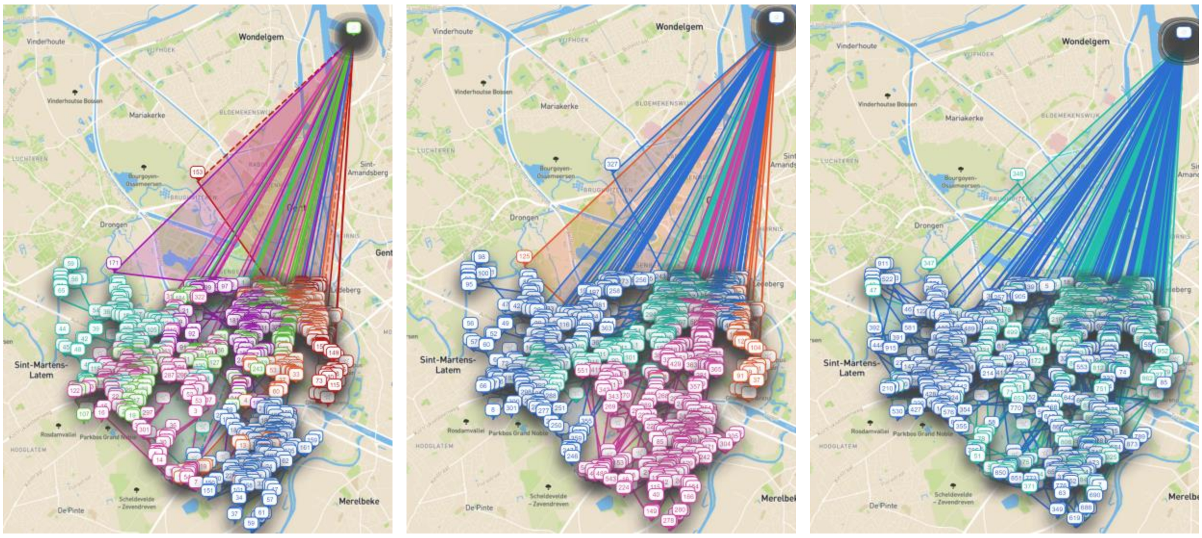


Weekly collection Fortnightly collection Monthly collection

Figure D.7 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 2.

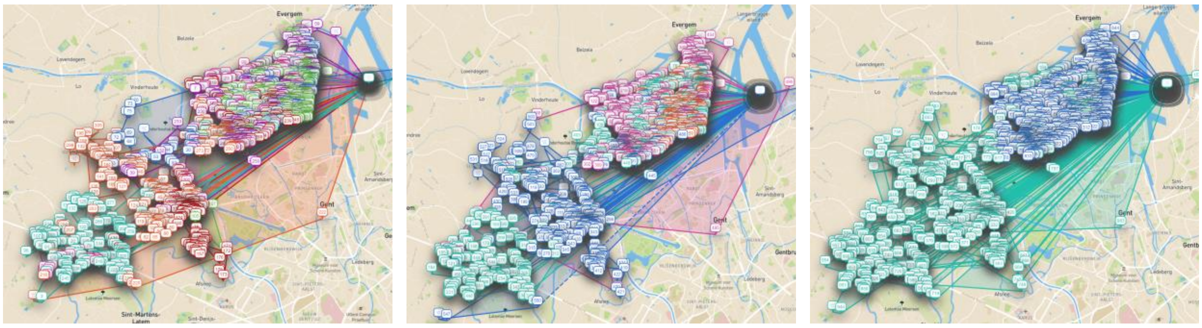
Weekly collection | Fortnightly collection | Monthly collection

Figure D.8 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 3.



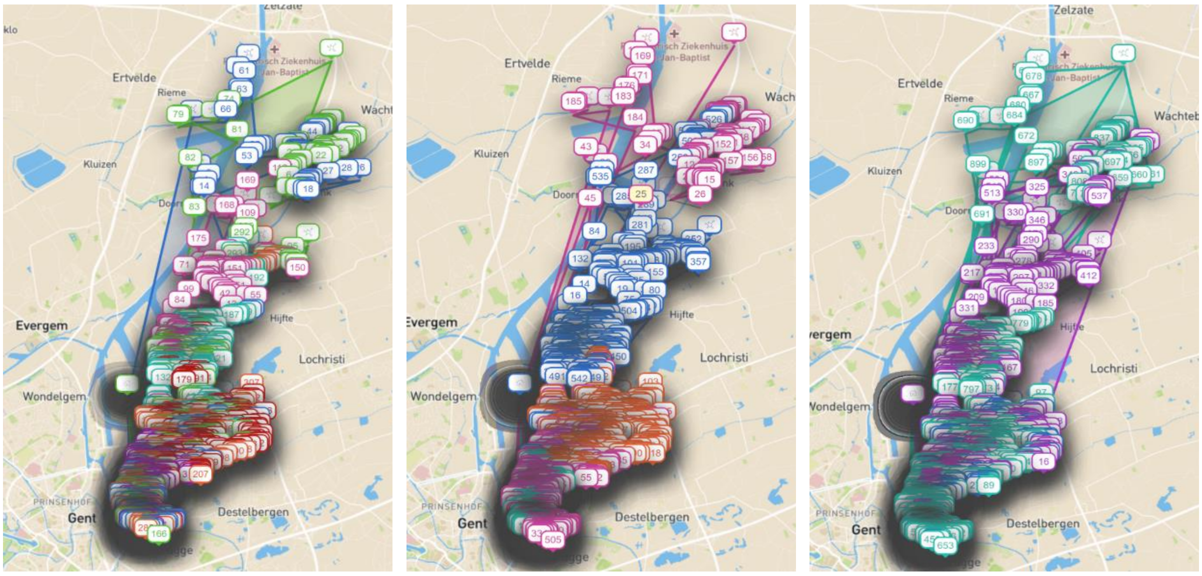
Weekly collection | Fortnightly collection | Monthly collection

Figure D.9 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 4.



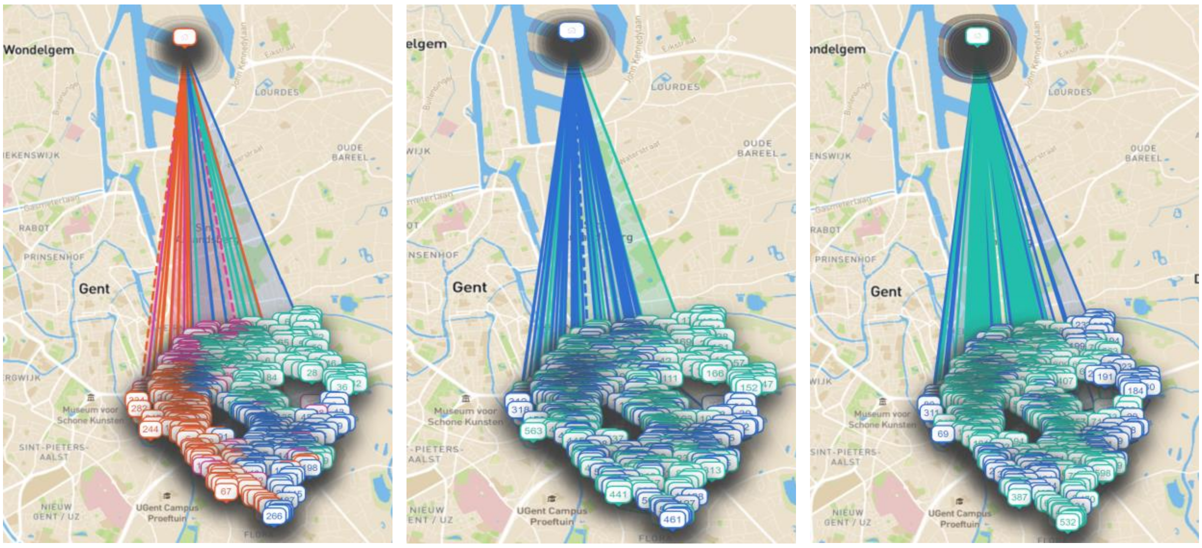
Weekly collection | Fortnightly collection | Monthly collection

Figure D.10 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 5.



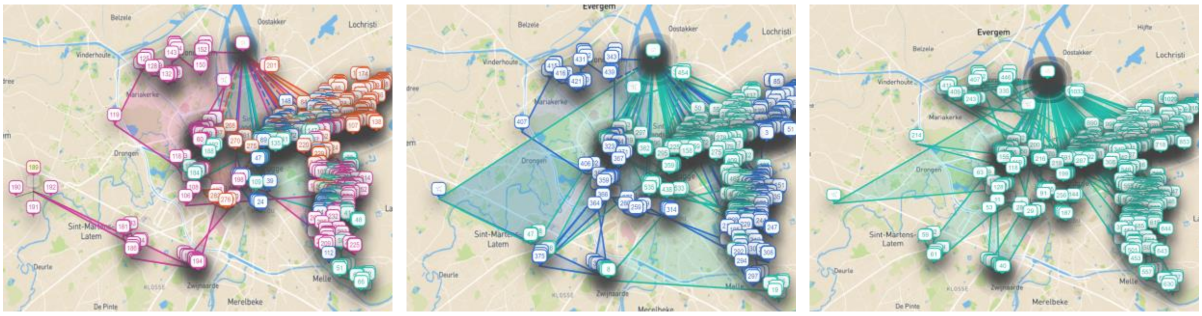
Weekly collection | Fortnightly collection | Monthly collection

Figure D.11 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 6.



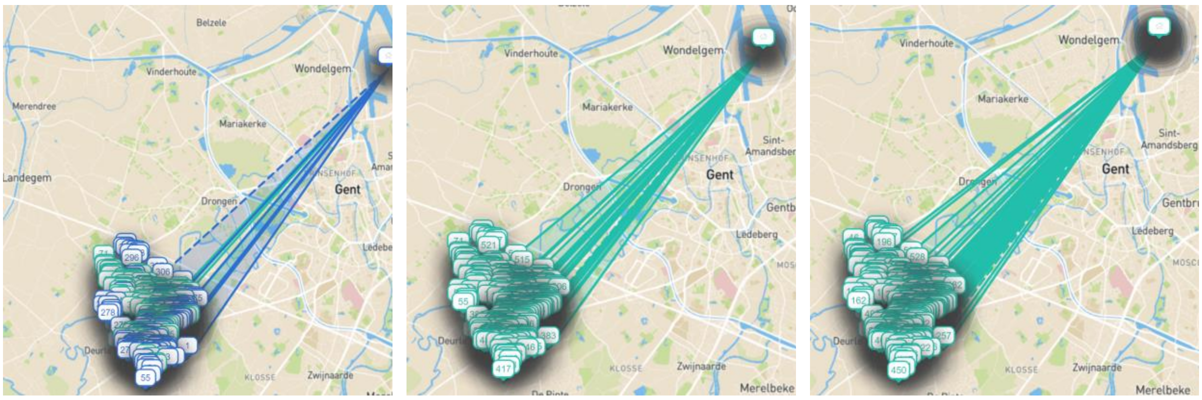
Weekly collection | Fortnightly collection | Monthly collection

Figure D.12 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 7.



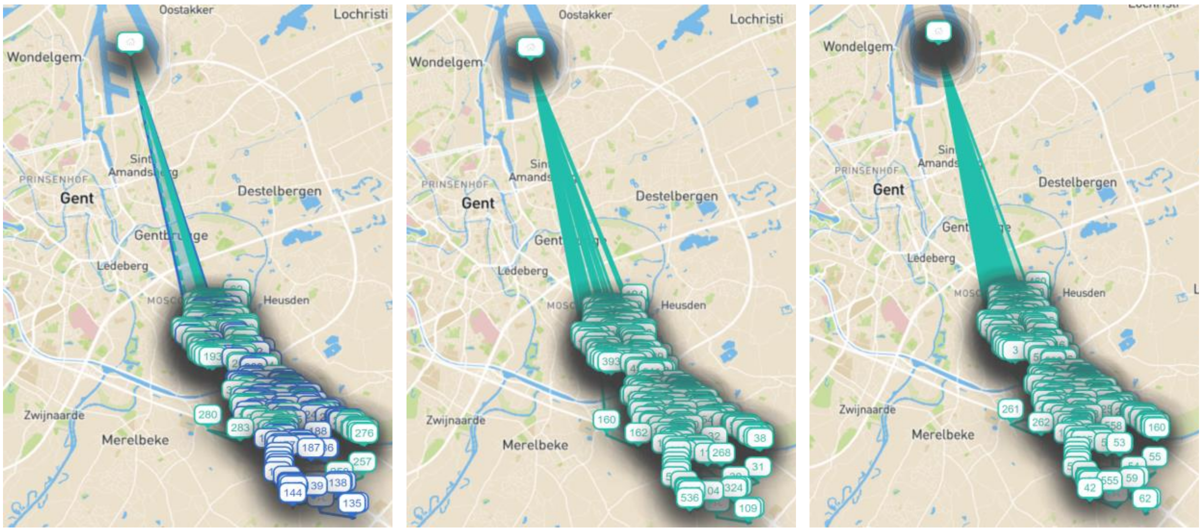
Weekly collection | Fortnightly collection | Monthly collection

Figure D.13 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Ghent – Belgium, sub-area 8.



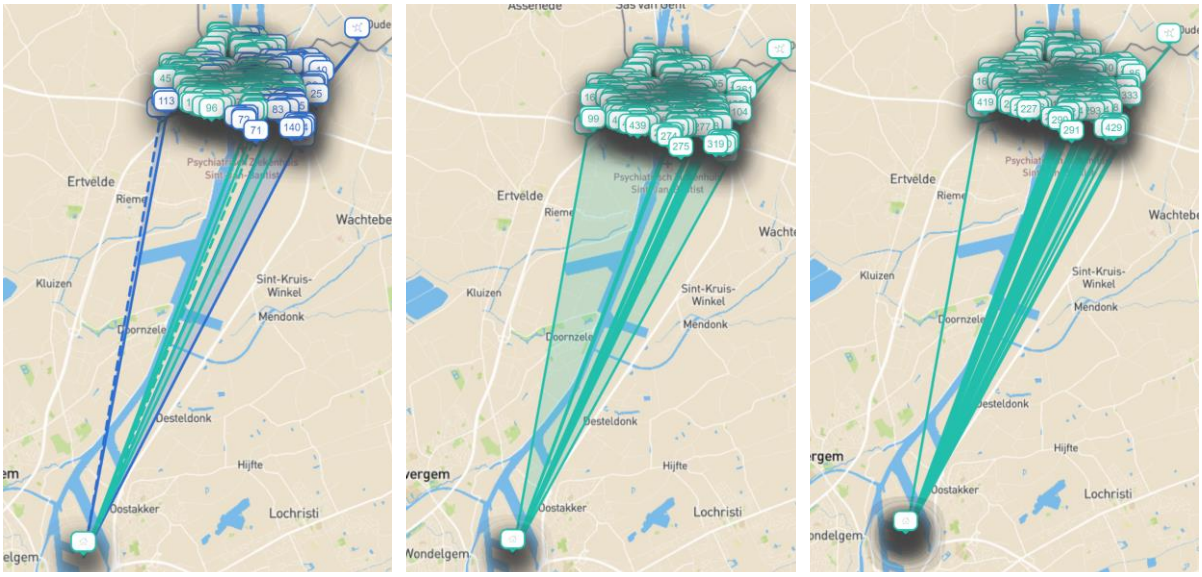
Weekly collection | Fortnightly collection | Monthly collection

Figure D.14 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Sint-Martens-Latem.



Weekly collection Fortnightly collection Monthly collection

Figure D.15 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Melle.

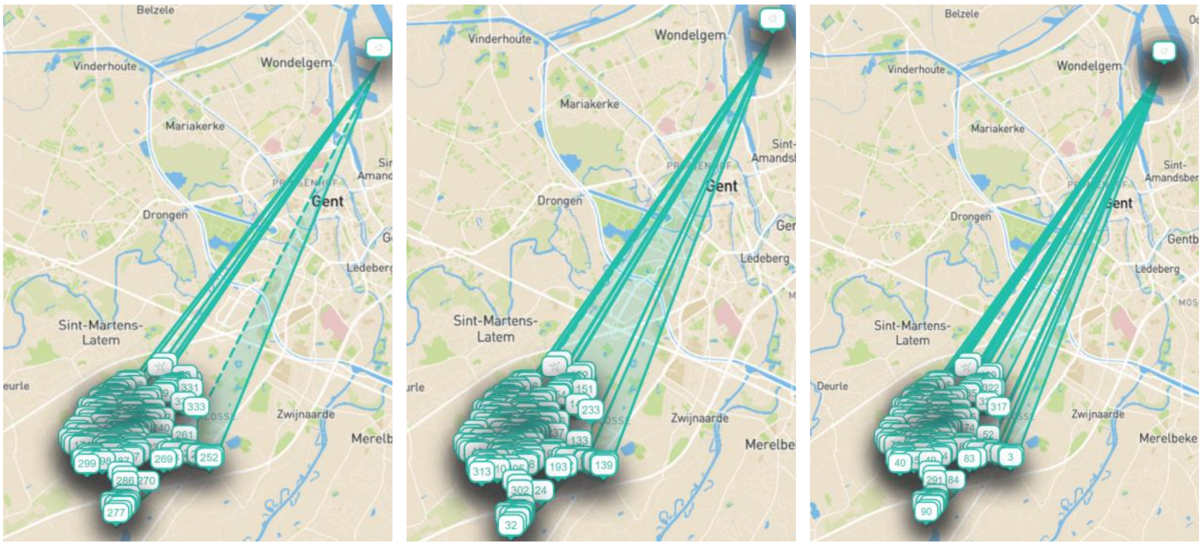


Weekly collection Fortnightly collection Monthly collection

Figure D.16 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Zelzate.

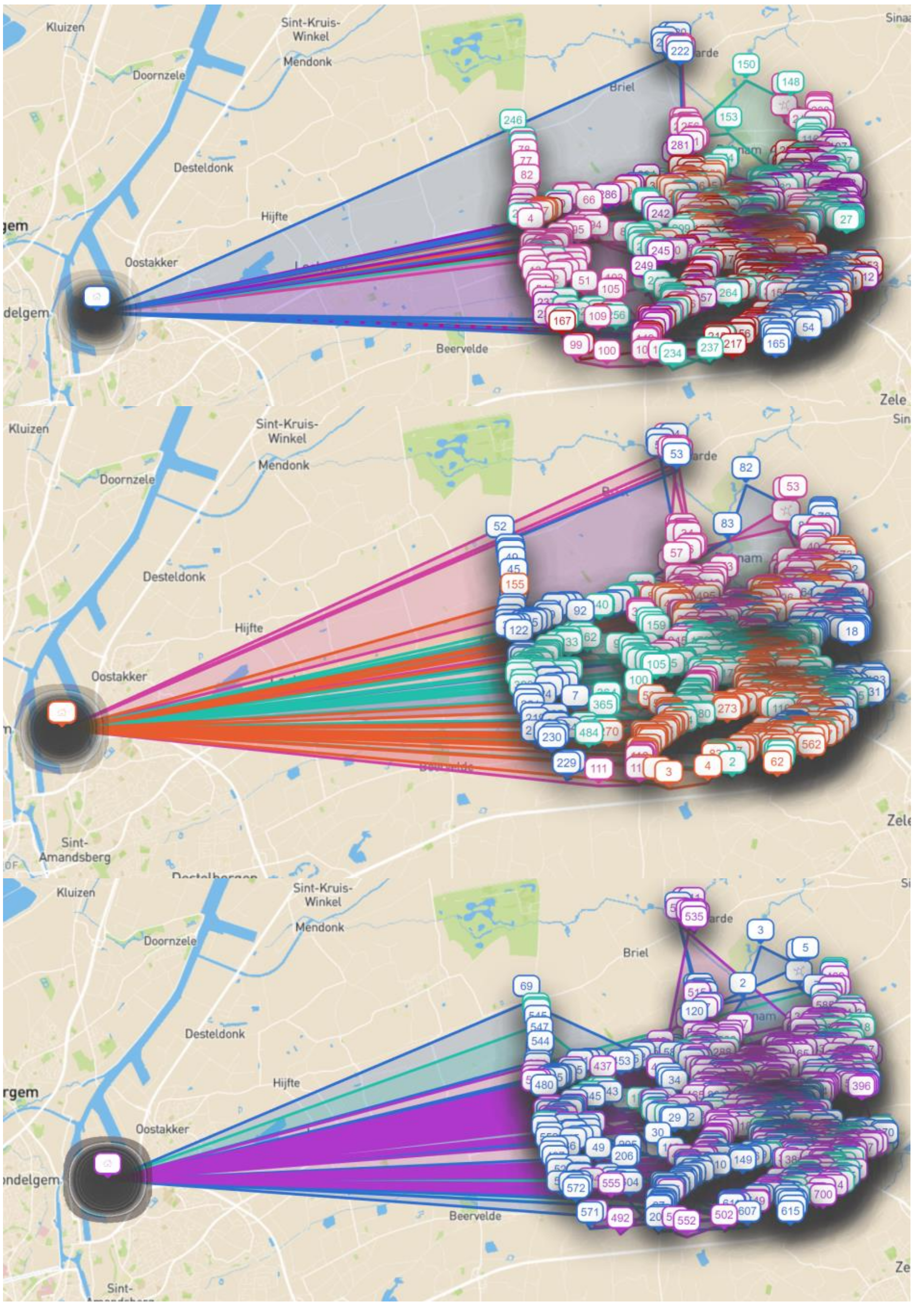
Weekly collection | Fortnightly collection | Monthly collection

Figure D.17 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Merelbeke.



Weekly collection | Fortnightly collection | Monthly collection

Figure D.18 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in De Pinte.

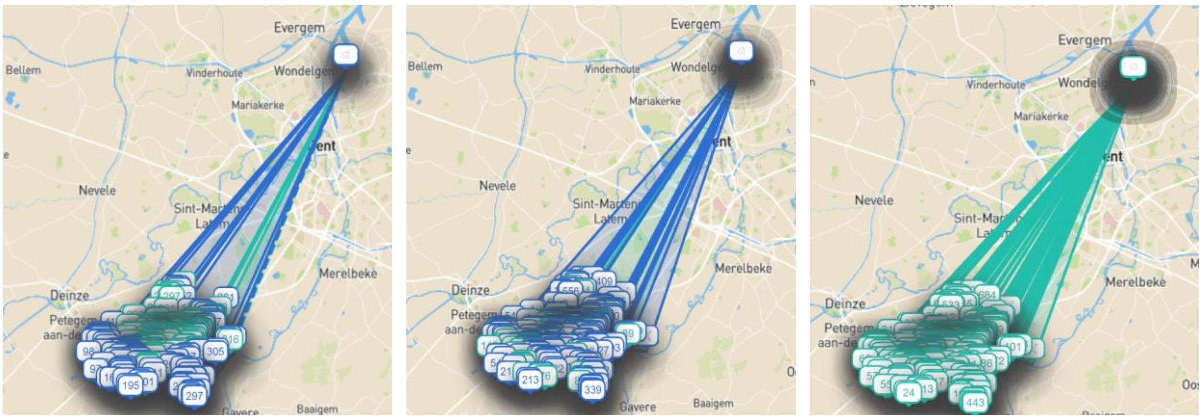


Weekly collection

Fortnightly collection

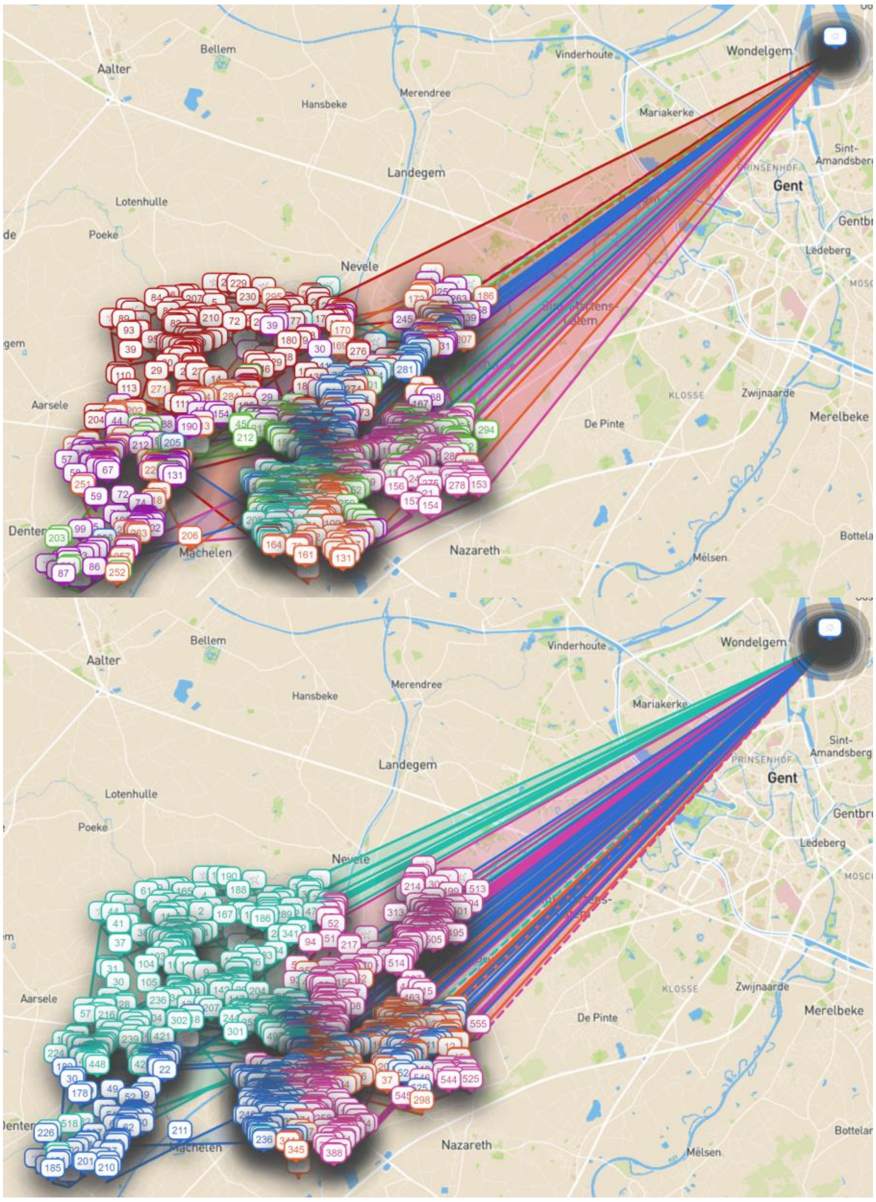
Monthly collection

Figure D.19 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Lokeren.



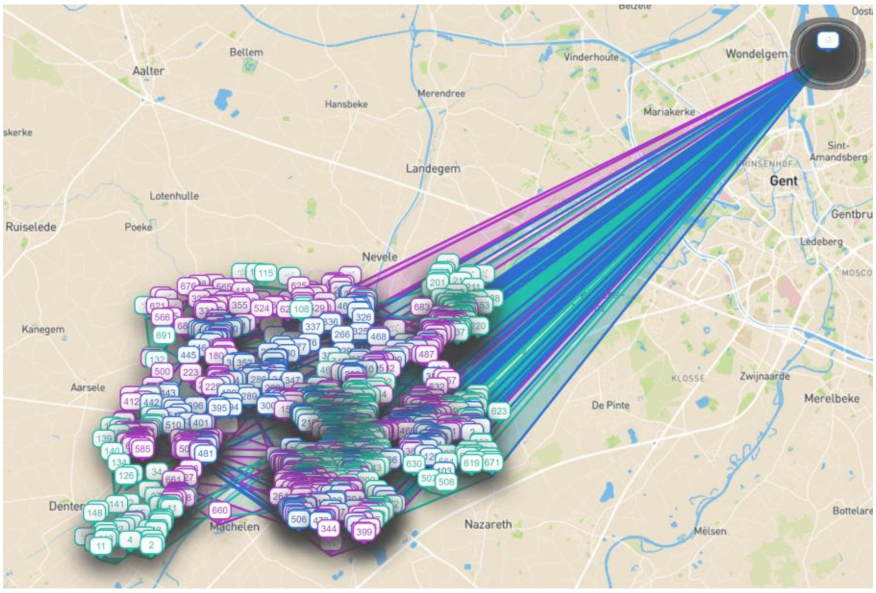
Weekly collection | Fortnightly collection | Monthly collection

Figure D.20 Images of the logistic simulation results of weekly (left), fortnightly (center), and monthly (right) collection in Nazareth.



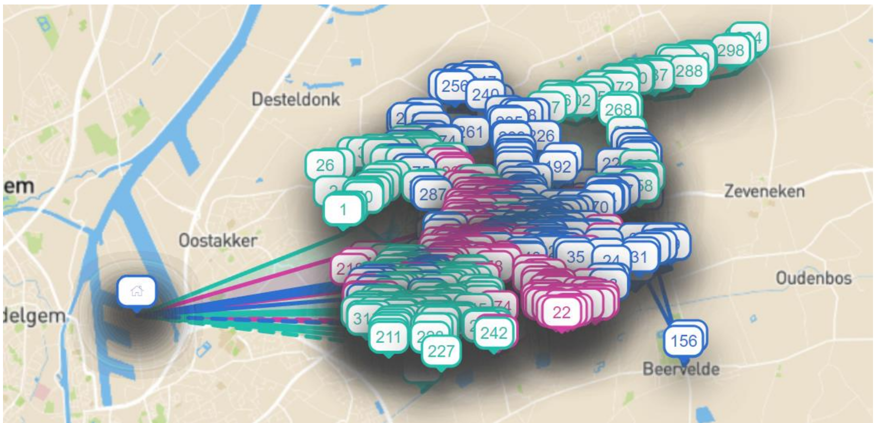
Weekly collection

Fortnightly collection

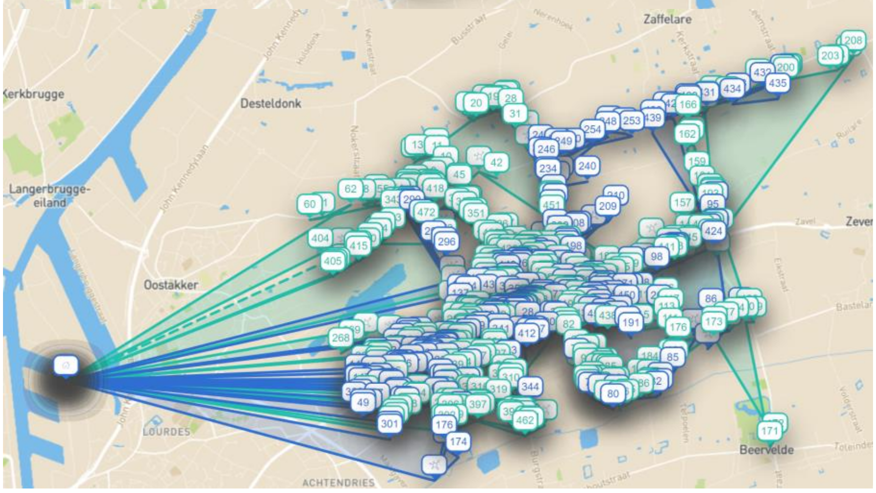


Monthly collection

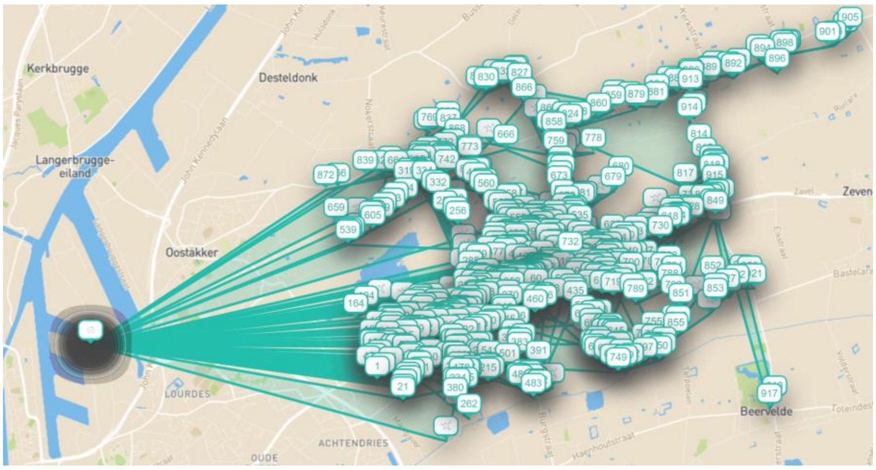
Figure D.21 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Deinze.



Weekly collection

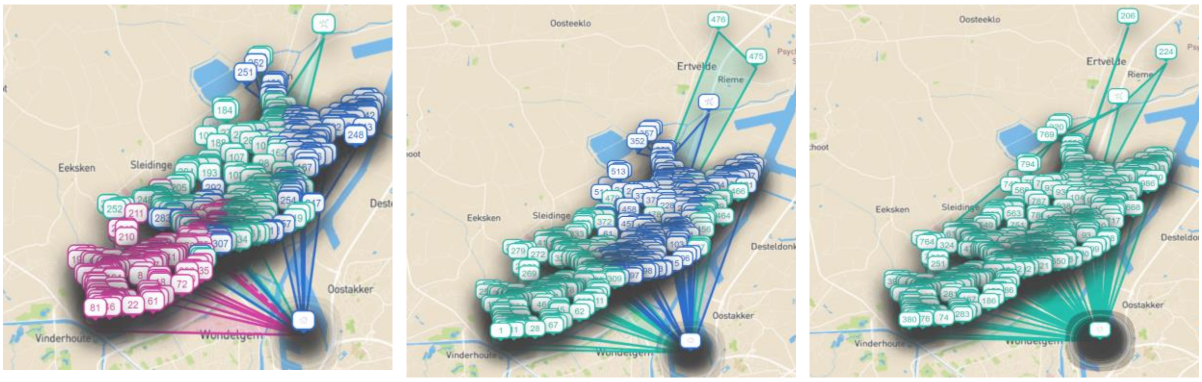


Fortnightly collection



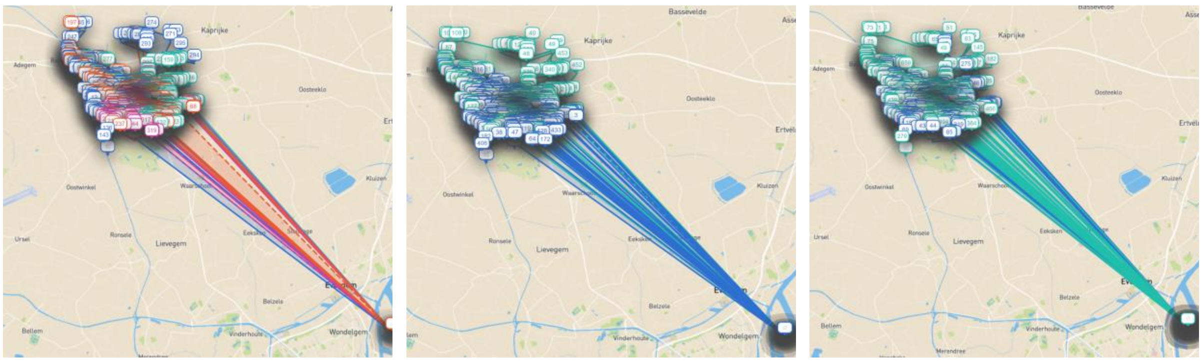
Monthly collection

Figure D.22 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Lochristi.



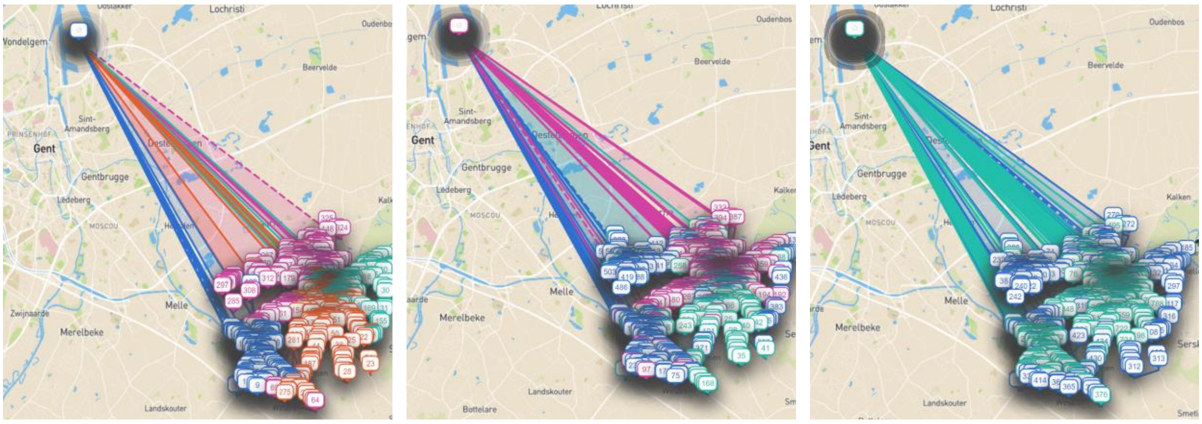
Weekly collection | Fortnightly collection | Monthly collection

Figure D.23 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Evergem.



Weekly collection | Fortnightly collection | Monthly collection

Figure D.24 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Eeklo.



Weekly collection | Fortnightly collection | Monthly collection

Figure D.25 Images of the logistic simulation results of weekly (top), fortnightly (middle), and monthly (bottom) collection in Wetteren.

SECTION 9: MATERIAL FLOW ANALYSIS OF NON-HOUSEHOLD END-USE PLASTIC FILM WASTE RECYCLING

Table D.10 Results on rPE Film production annually (in tonne/year) in different scenarios (S1–S4) depending on various recycling capacities (i.e., from 2,500 – 20,500 tonne/year).

S1: *77% recycling yield*

Recycling capacity	rPE Film production
2,500	1,932
4,500	3,478
6,500	5,024
8,500	6,570
10,500	8,115
12,500	9,661
14,500	11,207
16,500	12,752
18,500	14,298
20,500	15,844

S2: 61*% recycling yield*

Recycling capacity	rPE Film production
2,500	1,526
4,500	2,746
6,500	3,967
8,500	5,188
10,500	6,408
12,500	7,629
14,500	8,849
16,500	10,070
18,500	11,291
20,500	12,511

S3: 61*% recycling yield*

Recycling capacity	rPE Film production
2,500	1,533
4,500	2,759
6,500	3,985
8,500	5,212
10,500	6,438
12,500	7,664
14,500	8,891
16,500	10,117
18,500	11,343
20,500	12,570

S4: 48*% recycling yield*

Recycling capacity	rPE Film production
2,500	1,210
4,500	2,179
6,500	3,147
8,500	4,115
10,500	5,083
12,500	6,052
14,500	7,020
16,500	7,988
18,500	8,957
20,500	9,925

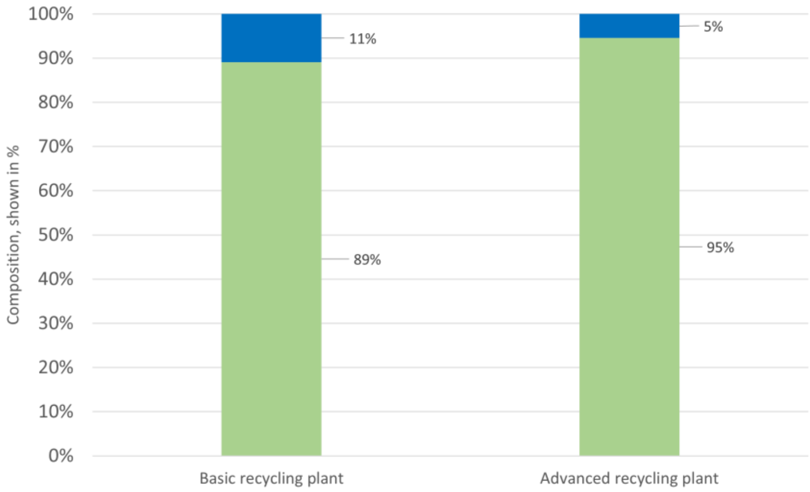
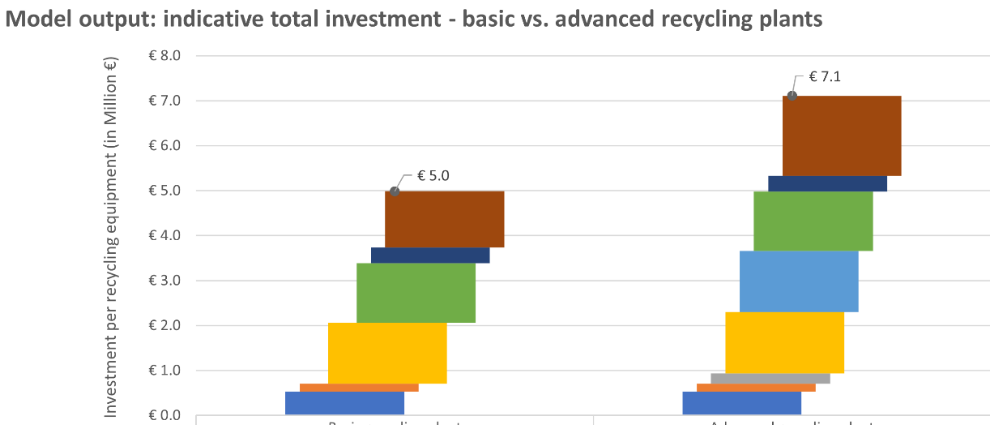


Figure D.26 The expected composition of rPE based on the MFA model

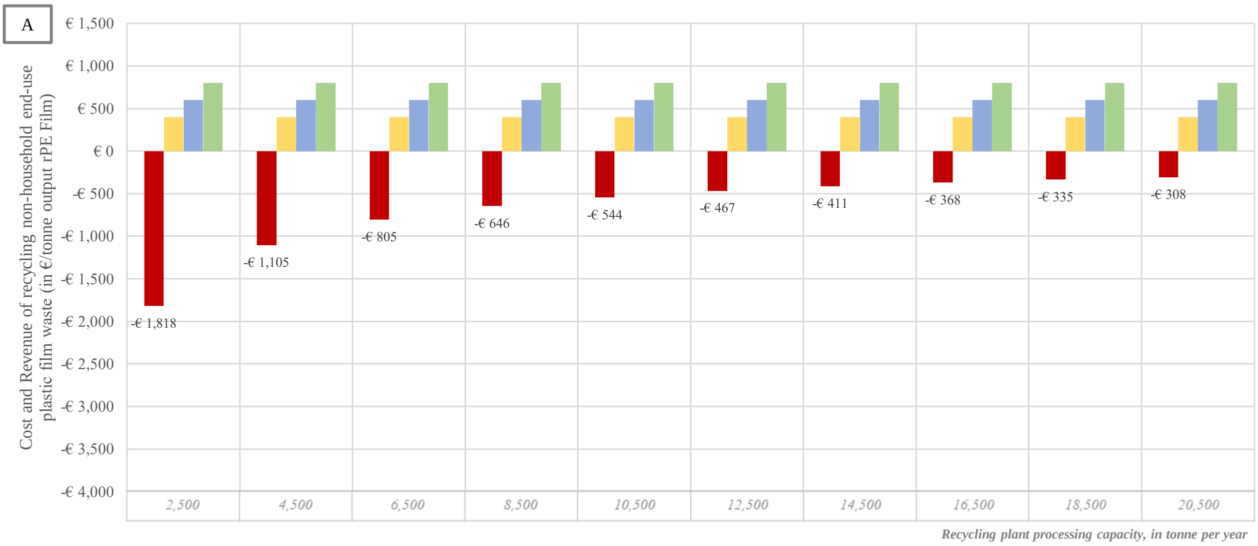
SECTION 10: ECONOMIC ASSESSMENT RESULTS – TOTAL CAPITAL INVESTMENT AND ANNUAL COSTS OF NON-HOUSEHOLD END-USE PLASTIC FILM RECYCLING

Figure D.6 shows the estimated capital investment needed for basic and advanced recycling plant. Figure D.7 shows the estimated annual costs and revenue (shown in €/tonne output rPE) of non-household end-use plastic film waste in different scenarios (S1–S4). Notice that the revenue (i.e., positive value; yellow, blue, and green bars in Figure D.7) are constant regardless of the recycling plant capacity because the results in Figure D.7 are shown in tonne/output (per 1 tonne rPE produced). On the other hand, the cost (i.e., negative value; red bars) changes. From Figure D.7, it can be observed that the costs can be reduced significantly as the recycling plant processes more waste. In the basic recycling plant (Figure D.7a and D.7b), the annual costs per tonne can be reduced up to €308 – €428 (when 20,500 tonne/year processing capacity is reached), depending on the feedstock quality (lower price per tonne is reached when we process higher feedstock quality). Similarly, the annual costs per tonne can be reduced up to €492 – €658 (when 20,500 tonne/year processing capacity is reached) for the advanced recycling plant, similarly depending on the feedstock quality (Figure D.7c and D.7d).



	Basic recycling plant	Advanced recycling plant
Recycling plant	€ 1.2	€ 1.8
Handling station	€ 0.4	€ 0.4
Single filter extruder	€ 1.3	€ 1.3
Hot Washing, incl. dryers	€ 0.0	€ 1.4
Cold Washing, incl. density separation and dryers	€ 1.4	€ 1.4
NIR PE Film Cleaner	€ 0.0	€ 0.2
Primary shredder	€ 0.2	€ 0.2
Bag opener	€ 0.5	€ 0.5
Total	€ 5.0	€ 7.1

Figure D.27 Model output – indicative total investment per recycling equipment for basic and advanced recycling plant configurations.


A
Cost and Revenue of recycling non-household end-use plastic film waste (in €/tonne output rPE Film)
€ 1,500
€ 1,000
€ 500
€ 0
-€ 500
-€ 1,000
-€ 1,500
-€ 2,000
-€ 2,500
-€ 3,000
-€ 3,500
-€ 4,000
-€ 1,818
-€ 1,105
-€ 805
-€ 646
-€ 544
-€ 467
-€ 411
-€ 368
-€ 335
-€ 308
2,500
4,500
6,500
8,500
10,500
12,500
14,500
16,500
18,500
20,500
Recycling plant processing capacity, in tonne per year

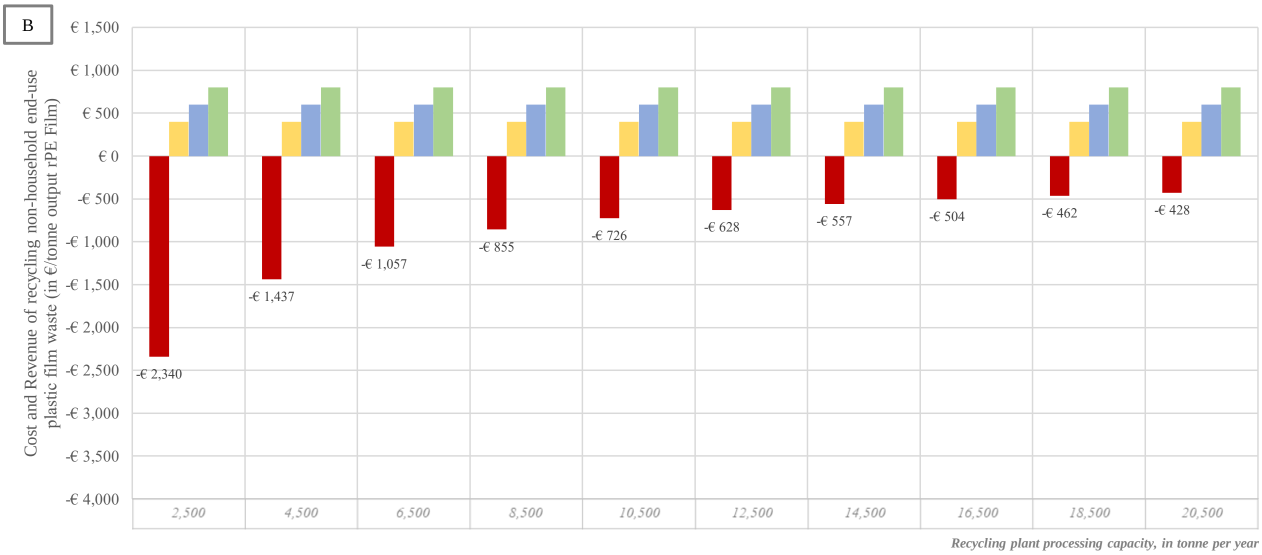

B
Cost and Revenue of recycling non-household end-use plastic film waste (in €/tonne output rPE Film)
€ 1,500
€ 1,000
€ 500
€ 0
-€ 500
-€ 1,000
-€ 1,500
-€ 2,000
-€ 2,500
-€ 3,000
-€ 3,500
-€ 4,000
-€ 2,340
-€ 1,437
-€ 1,057
-€ 855
-€ 726
-€ 628
-€ 557
-€ 504
-€ 462
-€ 428
2,500
4,500
6,500
8,500
10,500
12,500
14,500
16,500
18,500
20,500
Recycling plant processing capacity, in tonne per year

Figure D.28 Estimated annual costs (red bar) and revenue (green bar, high regranulate price; blue bar, central regranulate price; yellow bar, low regranulate price) of non-household end-use plastic film waste recycling in S1(A), S2(B), S3(C), and S4(D). The costs and revenue are shown in €/tonne rPE (y-axis) across different recycling plant processing capacity (x-axis, from 2,500 tonne/year up to 20,500 tonne/year capacity).

Table D.11 Model output – indicative total annual cost for all scenarios (S1–S4) depending on various recycling capacities (i.e., from 2,500 – 20,500 tonne/year).

Recycling plant capacity	**Estimated annual cost** (*in million €/year*)	**Recycling plant capacity**	**Estimated annual cost** (*in million €/year*)
S1		**S2**	
2,500	4.1	2,500	4.1
4,500	4.2	4,500	4.3
6,500	4.2	6,500	4.4
8,500	4.3	8,500	4.5
10,500	4.4	10,500	4.7
12,500	4.5	12,500	4.8
14,500	4.6	14,500	4.9
16,500	4.7	16,500	5.0
18,500	4.8	18,500	5.2
20,500	4.9	20,500	5.4
Recycling plant capacity	**Estimated annual cost** (*in million €/year*)	**Recycling plant capacity**	**Estimated annual cost** (*in million €/year*)
S3		**S4**	
2,500	4.9	2,500	5.0
4,500	5.0	4,500	5.1
6,500	5.2	6,500	5.3
8,500	5.3	8,500	5.5
10,500	5.5	10,500	5.6
12,500	5.6	12,500	5.8
14,500	5.7	14,500	6.0
16,500	5.9	16,500	6.2
18,500	6.0	18,500	6.4
20,500	6.2	20,500	6.5

CURRICULUM VITAE

Irdanto Saputra Lase holds a bachelor's degree in Geological Engineering from Universitas Padjadjaran, Bandung – Indonesia. He also holds a master's degree in Master of Science (MSc) in Sustainable and Innovative Natural Resources Management from Ghent University (Ghent – Belgium), Uppsala University (Uppsala – Sweden) and Technische Universität Bergakademie Freiberg (Freiberg – Germany). He started working on his Joint PhD research at Ghent University and Maastricht University in July 2020. During his PhD, he was involved in interdisciplinary research with a Circular Economy for Flexible Packaging (CEFLEX) consortium, Interreg 2 Seas Program PlastiCity project, and Perfect Sorting consortium project. His main tasks on these projects are material flow modeling of plastic waste throughout the end-of-life recycling systems, a techno-economic assessment of different recycling options, business development of new recycling technologies, and a circular economy evaluation for plastic in Europe. He also worked as a visiting scientist at the Joint Research Centre (JRC) European Commission (Sevilla office) to investigate the potential contribution of emerging plastic recycling technologies to the plastic recycling systems in Europe. His main tasks at the JRC European Commission are material flow modeling of current and future plastic waste treatment in Europe and economic and environmental impact assessment of various plastic waste recycling options in Europe. During his PhD he guided four master students investigating potential improvements of plastic waste treatment and life cycle assessment of different recycling scenarios in Europe.

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2020 – 2023	Doctor of Bioscience Engineering by Ghent University and Doctor by Maastricht University *Thesis: Modeling material flows through plastic recycling chains*

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2012 – 2016	Bachelor of Geological Engineering from Universitas Padjadjaran *Thesis: Seismic facies and hydrocarbon distribution mapping using neural network*

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2022	Visiting scientist at Joint Research Centre (JRC) European Commission – Sevilla, Spain
2019	Internship Trainee at Jan De Nul Group – Taipei, Taiwan
2017 – 2018	Site Engineer at GeoHarbor Group – Jakarta, Indonesia
2016 – 2017	Junior Geologist at Patra Nusa Data – Elnusa Group – Jakarta, Indonesia

Key Publications

Lase, I.S., Tonini, D., Caro, D., Albizzati, P. F., Cristóbal, J., Roosen, M., Kusenberg, M., Ragaert, K., Van Geem, K. M., Dewulf, J., De Meester, S. How much can chemical recycling contribute to plastic waste recycling in Europe? An assessment using material flow analysis modelling. *Resource Conservation and Recycling* 192, 106916. https://doi.org/10.1016/j.resconrec.2023.106916

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Roosen, M., Tonini, D., Albizzati, P. F., Caro, D., Cristobal, J., **Lase, I. S.**, Ragaert, K., Dumoulin, A., De Meester, S. 2023. An operational framework to quantify 'quality of recycling' across different material types. *Environmental Science & Technology*. In Press

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