



JRC SCIENCE FOR POLICY REPORT

Towards a better definition and calculation of recycling

Proposals for calculating recycling yields in multi-output processes and recycling of biodegradable waste, and a quality framework for recycling

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Abstract

Recycling is any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes, excluding energy recovery and reprocessing into materials that are to be used as fuels or for backfilling operations. The development of innovative technologies, such as multi-output recycling technologies (e.g. chemical recycling), calls for increasing clarity on several aspects of the definition. For instance, the current rules for calculating the amount of recycled municipal waste state that, in case of multi-output processes, the quantity recycled shall be determined by a mass balance approach. However, mass balance rules are not provided. Lack of sufficiently clear guidelines also apply to compostable plastic waste and quality of recycling. This lack of clarity is an obstacle to the conception of robust policy measures addressing recycling and circular economy. To close the gaps, this study contains technical proposals for i) calculation rules to perform mass balance for reporting of recycling rate in multi-output processes and ii) clarifications of the recycling calculation rules for biodegradable waste. In addition, the study presents an estimation of the impacts of the changes proposed and a preliminary framework to address quality of recycling.

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Executive summary

The objective of this study is to make a technical proposal for revising the implementing decisions related to recycling, in view of the expected development of new and advanced multi-output recycling technologies, such as chemical recycling technologies that are likely to emerge in the coming years. The report provides (i) a proposal for detailed calculation rules for performing a mass balance to calculate recycling yields in multi-output technologies, such as chemical recycling. Besides, (ii) it also proposes ways to clarify some of the calculation rules applied to the quantification of recycled bio-waste and compostable plastic waste in order to adapt them to the application of different recycling processes. The study also (iii) gives a preliminary assessment of the potential impact (costs and benefits) following the changes proposed and the development of advanced recycling technologies, such as chemical recycling for plastic packaging waste. Last, (iv) a framework proposal for exploring how to revise and extend the definition of quality of recycling and making it quantifiable is presented.

Policy context

One of the main pillars of the European Green Deal is the new Circular Economy Action Plan adopted by the European Commission in March 2020. Among its objectives, key importance is given to achieving the EU's 2050 climate neutrality target, announcing sustainable initiatives along the entire life cycle of products and promoting circular economy, and ensuring that waste is prevented and the resources used are kept in the economy for as long as possible.

In the endeavour to foster material recirculation, Directive 2018/851 amending the Waste Framework Directive (2008/98/EC) (European Commission, 2018a) and Directive 2018/852 amending the Packaging and Packaging Waste Directive (1994/62/EC) (European Commission, 2018b) call for Member States to achieve stringent targets (by weight) on re-use and recycling of municipal solid waste and packaging waste in the coming years, implying a significant effort in implementing best practices. Aiming at verifying whether the preparing for re-use and recycling targets for municipal waste in Directive 2018/851 and recycling targets for packaging waste are attained, two Commission's Implementing Decisions (2019/1004 for municipal waste (European Commission, 2019a) and 2019/665 for packaging waste (European Commission, 2019b)) have provided appropriate calculation rules. One of the rules for calculating recycled municipal waste or packaging waste states that 'where municipal waste/packaging waste materials enter recovery operations whereby those materials are not principally used either as fuel or other means to generate energy, or for material recovery, but result in output that includes recycled materials, fuels or backfilling materials in significant proportions, the amount of recycled waste shall be determined by a mass balance approach'. However, although such mass balance approach is mentioned in both Implementing Decision 2019/1004 (Article 3) and, for the specific case of packaging, Implementing Decision 2019/665 (Article 6c), clear rules on how to perform it are not provided. In relation to this legislation, additional guidance appears also needed to clarify the calculation of recycled waste for the case of compostable plastic waste since its use is encouraged for specific applications by the new EU policy framework within the new circular economy action plan¹. While EC Implementing Decision 2019/1004 provides rules for the calculation of bio-waste recycling, the way compostable plastic waste should be addressed is not sufficiently clear. Finally, the Waste Framework Directive, while encouraging high-quality recycling, does not provide a clear definition of it, thereby calling for an improvement of the understanding of what high- or low- quality actually means.

Key conclusions

Proposal for a mass balance approach

¹ The new EU Circular Economy Action Plan. Available at: https://environment.ec.europa.eu/system/files/2022-12/COM_2022_682_1_EN_ACT_part1_v4.pdf.

The mathematical framework herein presented calculates the recycling yield, the energy recovery yield and the loss yield of the recycling process. We herein use the term recycling yield that is associated with the recycling process itself. Indeed, the recycling process represents the system boundary of our calculation rules as opposed to the end-of-life recycling rate that refers to the yield (also called sometime 'efficiency') of the entire recycling chain, including collection, sorting, and recycling. By defining the system boundaries in this way, we can consider the recycling yield herein calculated as the (input-output) material recovery efficiency of a recycling process. An important aspect of the mass balance approach presented is that the recycling yield is referred only to the waste feedstock in input.

Proposal for Compostable Plastic Waste

Concerning bio-waste and compostable plastic waste, it is proposed to maintain largely the current calculation rules with some adjustments in the formulation of selected articles of the EC implementing decisions 2019/1004 and 2019/665. Notably, it is proposed to clearly open the scope of the technologies that can recycle biodegradable waste apart from aerobic/anaerobic treatments, provided that the amount of output obtained is comparable (in quantity) to the benchmark represented by the typical output of a plant performing composting/anaerobic digestion, and is used as a recycled product, material or substance. Also, it is proposed how to specifically calculate and report the recycling of compostable plastic waste when treated together with biowaste.

Effects of the changes proposed

To estimate the effects, we focused on plastic waste packaging and projected the EU plastic waste packaging flow up to 2030 considering two different management scenarios: a scenario where part of plastic waste is managed through mechanical recycling and the rest is incinerated (this represents the baseline; no chemical recycling), and a scenario where mechanical and chemical recycling are considered together so as to decrease the amount of plastic waste incinerated (this represents an increase of multi-output chemical recycling technologies, as expected by year 2030). The results obtained for the economic assessment highlighted that the development of chemical recycling technologies in the coming years needs substantial investments that cannot be entirely balanced from the associated revenues. With respect to the environmental analysis, which only focused on the quantification of greenhouse gas emissions, the scenario where both chemical and mechanical recycling are considered performs slightly better than the baseline where no chemical recycling takes place. It should be noted that our economic and environmental estimations are very conservative as they do not take into account the potential improvement of the chemical technologies over time (e.g., economy of scale or optimized efficiency in terms of reduced energy consumption). Hence, such results should be seen as preliminary, rather conservative, and used with care.

Proposal for quality of recycling

A framework defining quality of recycling is presented and applied to a case study on PET. The framework is based on three main dimensions, namely the Total Substitution Potential (telling how much primary material can be replaced via secondary material), the Long-Term in-Use Occupation (telling for how long the recycled material remains in the economic system), and the Environmental Impact (here represented only by the carbon footprint). Each of these three dimensions provides additional information relevant for defining the quality of recycling. In general, we conclude that the higher the Total Substitution Potential and Long-Term in-Use Occupation and the lower the Environmental Impact of a certain recycling pathway, the higher the quality of recycling.

Main findings

The main findings of this report can be summarised as follows:

- A compilation of calculation rules for estimating the recycling yield for multi-output technologies based on a mass balance approach.

- A clarification on calculation rules for estimating the recycling yield of biodegradable waste (including both bio-waste and compostable plastic waste).
- The estimation of preliminary costs and environmental burdens/benefits that may arise from implementing calculations rules and changes proposed herein.
- The proposal of a new framework for defining quality of recycling with its application to a case study.

Related and future JRC work

It should be noted that a parallel project on life cycle assessment of chemical recycling from JRC is currently ongoing (Garcia-Gutierrez et al., 2023), with the general aim of assessing the performance of plastic waste management via chemical recycling. Synergies between these two projects have been exploited by estimating the effects related to the development of chemical recycling for plastic packaging waste by 2030. Further connections are expected as this study only shows a few future scenarios with preliminary data that should be further improved.

Concerning future outlooks, this report should be seen as a first attempt to provide guidance on mass balance for calculating recycling and on quality of recycling. The proposals provided in this report may thus be further refined over the coming years. The main limitation of the mass balance herein presented is the traceability of material flows from one operator to another one downstream the recycling value chain.

Especially, the quality of recycling framework should be taken as a first attempt to define quality and needs further testing on case studies at industrial level to verify its feasibility and applicability at industrial level.

Quick guide

This report is composed of two main parts. After the introductory sections 1 and 2, section 3-to-6 focus on recycling and recycling calculation rules for multi-output processes and biodegradable waste recycling. In particular, section 6 summarizes the technical proposals. The second part presents a literature review on quality of recycling (section 7) and a preliminary framework to define it and quantify it (section 8).

Disclaimer

It should be stressed that, at this point, the European Commission has not undertaken any steps for the revision of the definition of recycling. Therefore:

- The present document contains technical proposals and does not constitute the official opinion of the European Commission regarding the revision of the definition of recycling and related calculation rules;
- The present document does not constitute any commitment by the European Commission to start work on the revision of the definition of recycling and related calculation rules.

1 Introduction

1.1 Rules for calculation and reporting of recycling

The new Circular Economy Action Plan adopted by the European Commission in March 2020 is one of the main building blocks of the European Green Deal (European Commission, 2019c) and a prerequisite to achieve the EU's 2050 climate neutrality target. The new action plan announces initiatives along the entire life cycle of products. It targets how products are designed, encourages sustainable consumption, and promotes circular economy, to ensure that waste is prevented and the resources used are kept in the economy for as long as possible. In this context, recycling plays a key role in closing material loops and ensuring lasting value, both for the end-product and the virgin resource.

Directive 2018/851 amending the Waste Framework Directive, calls Member States for achieving a minimum of 55% by weight of re-use and recycling of municipal solid waste (MSW) by 2025, 60% by 2030, and 65% by 2035 (European Commission, 2018a). According to European Environment Agency (2021), the average rate of preparation for re-use and recycling of municipal waste in EU Member States was about 48% in 2019. Therefore, a significant effort in implementing best practices is urgently required in the coming years (Hann et al., 2020).

Recycling is defined by Directive 2008/98 as "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" (European Commission, 2008).

Member States shall take appropriate measures to ensure that waste which has undergone a recycling or other recovery operation ceases to be waste, complying with the following conditions: (i) the substance or object is to be used for specific purposes; (ii) a market or demand exists for such a substance or object; (iii) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and, (iv) the use of the substance or object will not lead to overall adverse environmental or human health impacts.

For the purpose of verifying whether the preparing for re-use and recycling targets for municipal waste for years 2025, 2030 and 2035 of Directive 2008/98/EC as amended by Directive 2018/851 are attained, as well as targets for 2025 and 2030 of Directive 1994/62/EC as amended by Directive 2018/852, the EC Implementing Decisions 2019/1004 and 2019/665 provide appropriate calculation rules. The rules set out specify that, as regards recycling, waste that enters a recycling operation or waste that has achieved end of waste (EoW) status is to be used for the calculation of the targets for 2025, 2030 and 2035. As a general rule, the recycled waste is to be measured at the point where the waste enters the recycling operation. According to Commission Implementing Decision 2019/665 (for packaging) and Commission Implementing Decision 2019/1004 (for MSW) the amount of recycled waste shall be the amount of municipal waste at the calculation point. The current main rules for calculating recycled municipal and packaging waste are summarised herein:

- The amount of municipal waste entering the recycling operation shall include targeted materials. It may include non-targeted materials² only to the extent that their presence is tolerable for the specific recycling operation (i.e., acceptable by the market; Antonopoulos et al., 2021).
- Where the measurement point relates to the output of a facility that sends waste for recycling without further pre-treatment, or to the input to a facility where waste enters the recycling operation without any pre-treatment, the amount of sorted

² 'non-targeted materials' means waste materials that are not reprocessed in a given recycling operation into products, materials or substances that are not waste.

waste that is rejected by the recycling facility shall not be included in the amount of recycled waste.

- Where a facility carries out pre-treatment operations prior to the calculation point in that facility, the waste removed during the pre-treatment shall not be included in the amount of recycled municipal waste reported by that facility.
- Where the humidity rate of packaging waste at the measurement point differs from that of packaging placed on the market, the amount of packaging at the measurement point shall be corrected in order to reflect the natural humidity rate of the packaging waste comparable to the humidity of equivalent packaging placed on the market.
- Where biodegradable packaging that is subjected to aerobic or anaerobic treatment is included in the recycled amounts for the respective packaging material, the amount of biodegradable packaging in biodegradable waste shall be determined by performing regular composition analyses of the biodegradable waste entering those operations. Biodegradable packaging waste that is removed before, during or after the recycling operation shall not be included in the recycled amounts.
- Where waste materials enter recovery operations whereby those materials are used principally as a fuel or other means to generate energy, the output of such operations that is subject to material recovery, such as the mineral fraction of incineration bottom ash or clinker resulting from co-incineration, shall not be included in the amount of municipal waste recycled.
- Where waste materials enter recovery operations whereby those materials are not principally used either as fuel or other means to generate energy, or for material recovery, but result in output that includes recycled materials, fuels or backfilling materials in significant proportions, the amount of recycled waste shall be determined by a mass balance approach which results in accounting only for waste materials that are subjected to recycling.

As for the latter point, although such mass balance approach is mentioned in both Commission Implementing Decision 2019/1004 (Article 3) and, for the specific case of packaging, Commission Implementing Decision 2019/665 (Article 6c), clear rules on how to perform it are not sufficiently provided. This calls for a clarification of the mass balance (and, broadly, the recycling calculation rules), especially considering that the Commission regularly receives requests from the industry and Member States to revisit the definition of recycling to allow for more innovative technologies to be considered recycling or contribute to recycling objectives and secure legal certainty for their investments. It is therefore necessary to increase clarity on how the mass balance should be performed to estimate the recycling rates proposing appropriate mathematical methodologies, e.g., using a mass balance approach in which results of specific recycling rates, energy recoveries and losses are calculated taking into account only waste materials that are subjected to recycling. While recycling needs to be a transparent and verifiable process to ensure material provenance and probity of the system, the mass-balance approach is a proven chain of custody approach, and it appears as essential to assess recycling rates in complex technology systems (e.g., having a mix of materials and fuels in output), to bring transparency and audited traceability process. This particularly affects technologies involving multiple-output products (materials, chemicals, energy and fuels), such as chemical recycling processes but also other emerging recycling processes, for which such mass balance may not be straightforward. Therefore, a sound common approach that ensures a high level of reliability of the reported data should be established. This study aims to close this gap.

1.2 Quality of recycling

Quality of recycling is a rather complex concept, at the same time acknowledged as very important and left undefined in both EU *acquis* and scientific literature. While the recycling

rates certainly provide an indication of the amount of materials recovered by the waste management system in place in the EU, they nevertheless do not give any information on the quality of the material recovered. The quality is important as it determines the type of use of the recyclate and its further recyclability. This in turn affects the closure of material loops in specific sectors/markets, i.e., the circularity. An example is the following: plastic from food-contact materials could be recycled and used for fibres production (e.g., PET for textile market) when the quality is not high enough for food grade applications. However, this prevents closing the loop in the food packaging sector where additional virgin plastic would be needed to compensate the loss of material. A similar case could apply when PP plastic packaging waste is recycled into low value products such as benches in place of higher value applications. In both examples, there is an evident loss of the original product functionality (food-contact) and also a likely decrease in the further recyclability of the new products.

While authors explicitly or implicitly refer to quality of recycling as to a concept well-known and -defined, an in-depth analysis of the technical and scientific literature seems to suggest that this is not the case. It is remarkable that the Waste Framework Directive (European Commission, 2018a) repeatedly mentions that recycling should be steered towards high-quality without providing a rigorous definition of high-quality recycling. Most of the scientific studies, while proposing indicators reflecting quality or improving current recycling rates indicators used at EU level, leave the concept of "quality" undefined (e.g., Eriksen & Astrup, 2019; Haupt et al., 2017; Roithner & Rechberger, 2020). A lack of clarity on what quality means is a crucial obstacle to the conception of robust policy measures addressing recycling and in broader sense circular economy. In this study we aim to close this gap by proposing a framework for the definition of quality of recycling. We do this by (i) reviewing the available studies tackling recycling quality, (ii) synthesizing the approaches available, and (iii) suggesting a possible way forward for the definition and its operationalisation.

2 Scope and objectives

This study focuses on the definition of recycling as provided in the EU Directive 2008/98 and the calculation rules laid out in the subsequent Commission Implementing Decisions 2019/1004 (European Commission, 2019a) for MSW, and 2019/665 (European Commission, 2019b) for packaging. Accordingly, the main focus is on municipal solid waste and its material fractions (notably bio-waste, glass, metals, paper, and plastic), of which packaging waste is considered as a subset of selected fractions of MSW (e.g., packaging forms part of paper, plastic). However, the general principles and guidance drawn in this study are expected to be applicable also to waste streams other than MSW.

The overarching objective of this study is to produce a Technical Proposal to revise or extend the definition of recycling and the existing recycling performance calculation rules to take account of different recycling process and technologies, in particular, chemical recycling. Alongside, an assessment of the impacts as a consequence to the changes in the definition of recycling and related calculation rules proposed is also performed, which can inform the Commission in its policy development work.

To this purpose, the study has the following specific objectives:

- Perform a literature review of the different types of existing and emerging technologies (hybrid recovery/recycling processes that currently fall outside the scope of the definition due to their primary goal being energy/fuel production or that have multiple outputs including fuel or other means to generate energy alongside producing recycled materials), in particular considering the quality of the output materials.
- Perform a literature review of how quality of recycling is currently understood at the technical and scientific level (state-of-the-art).
- Propose recycling calculation rules for hybrid recovery/recycling processes applying them on selected waste streams as case study (e.g., chemical recycling of packaging plastics or plastic waste).
- Propose a definition of quality of recycling and options on how to operationalise it in the definition or by means of supporting measures.
- Estimate the (possible) impacts of changing the calculation rules for measuring recycling performance.

The technical proposals are submitted to and discussed with a wide range of stakeholders during dedicated consultations, which are summarised in this document. Notwithstanding this, the present document cannot be regarded as the official opinion of the European Commission regarding the revision of the definition of recycling and does not constitute any commitment by the European Commission to start work on the revision of the definition.

3 Recycling processes: state-of-the-art

Recycling processes are typically distinguished between mechanical/physical (e.g., mechanical recycling of plastics), chemical (e.g., chemical recycling of plastics, tyres, wood waste, bio-waste), and biological (e.g., anaerobic digestion, composting, other fermentation-based processes) processes. The different types of recycling pathways can be summarised as follows:

- Mechanical/physical recycling: it refers to operations that aim to recover material from the waste using mechanical/physical processes (washing, grinding, separating, drying, etc.). These operations do not alter the polymeric structure of the waste.
- Chemical recycling: it refers to operations that aim to recover material, substances and products from the waste by changes to its chemical structure using chemical processes.
- Biological recycling: it refers to operations that aim to recover material, substances and products from the waste by breaking down its chemical structure using biological processes (e.g., enzymatic hydrolysis, biological oxidation, fermentation).

An overview of the common and emerging recycling techniques for the various material fractions composing MSW are presented in sections 3.1 and 3.2. We focus on bio-waste, glass waste, metal waste, paper and cardboard waste, plastic waste (including biodegradable plastics), wood waste, textile waste, bulky waste, and composite materials. Recycling of batteries, waste electrical and electronic equipment and hazardous household waste is not included here because they are investigated thoroughly in other ongoing JRC studies.

3.1 Established recycling processes

Recycling of bio-waste

Bio-waste (food and garden waste, green cuttings) is typically recycled via biological recycling through composting or anaerobic digestion. Both treatments could be optionally preceded by a pre-treatment stage where the collected bio-waste is sorted (e.g., using magnets and screening technologies) to remove unwanted items, such as packaging, bags, and other miss-thrown waste materials (sorting). Composting is an aerobic process whereby, through mechanical processing (called turning) and/or forced ventilation, the degradable organic matter in the bio-waste is converted into carbon dioxide (CO₂), water vapour, and a stabilised organic matter (compost). The latter, after a refining stage for removal of wood chips and other large non-degraded items (e.g., non-degraded branches or wood fractions of garden waste) that are recirculated into the aerobic process, is then used as organic fertiliser (soil amendment) for agriculture, gardening, horticulture (substituting for fossil carbon peat) or as material for landscaping. In anaerobic digestion, the fermentation occurs in a closed reactor under anaerobic conditions leading to the production of two main outputs, i.e., biogas (composed mainly by methane and CO₂) and digestate (residual unconverted organic feedstock). The former can be used for energy purposes, via upgrading to natural gas quality or direct combustion for heat and electricity generation, while the digestate is used as fertiliser and soil improver either in raw form or after post-treatment. Post-treatments can consist of dewatering, drying, fast composting or more advanced processing techniques to recover high-value NPK fertilisers as detailed in recent work by JRC (Huygens et al., 2019; Tonini et al., 2019). Although not yet widely established at commercial and market level, more advanced technologies for bio-waste biological and chemical recycling are being developed, e.g., for the production of animal feed from bio-waste, bioplastics and other chemical platforms such as lactic acid, succinic acid, etc. (see section 3.2 for more information on emerging recycling technologies).

Recycling of glass waste

Glass manufacturing plants can use glass cullets together with conventional raw materials (limestone, CaCO_3 , sand, SiO_2 , and soda ash, Na_2CO_3) to lower the melting temperature and, therefore, reduce the energy needed for the production process. The glass waste goes through a pre-treatment process (sorting) which removes unwanted material (e.g., paper or plastic) normally using blown air. Further, metal objects are removed with magnets or eddy current system separators. Next, the waste flow is sorted by colour through optical sorting and washed to remove any further impurities. The pre-treated feedstock is then crushed and fed in the furnace (together with the primary material) to be melted, substituting conventional raw materials that would otherwise be used (limestone CaCO_3 , sand, SiO_2 , and soda ash, Na_2CO_3). Then, it is finally moulded into new products such as bottles and jars. Glass does not degrade through the recycling process so it can be recycled indefinite times.

Recycling of metal waste

The reprocessing of steel is typically occurring via electric arc furnace (EAF) or basic oxygen furnace (BOF) (Damgaard et al., 2009). Prior to EAF or BOF, pre-treatment (sorting) operations take place to remove unwanted items. The BOF process accepts only 25–30% of scrap steel, while the EAF process accepts 100% steel scrap and this is where the majority of the post-consumer steel scrap ends up. The main steps of the EAF process are as follows. The scrap is first preheated with the off gas generated at latter steps in order to conserve energy (and optionally additional fossil energy can be added). Next, the scrap is loaded in baskets together with lime, which is used as a flux. The furnace anodes are then lowered into the scrap. The initial energy to the arcs is kept low, until they are fully submerged in the scrap at which point the energy is increased until complete melting. Oxygen can be added to the early stages of the melting to boost the process. When the final temperature has been reached, the liquefied steel is tapped into a ladle, and alloying and deoxidizing compounds are added. After this, the steel is sent for casting to produce any kind of final product.

Aluminium recycling mainly takes place in rotary or reverberatory furnaces; for very clean aluminium grades, induction furnaces can be used but these take up a very small part of the aluminium recycling (Damgaard et al., 2009). For the aluminium collected via MSW, (e.g., beverage cans and foils), it is necessary to pre-treat the aluminium to remove contaminants and de-coat or de-oil the scrap. This improves the thermal efficiency of recycling and reduces potential emissions from the melting process. After pre-treatment, the scrap is loaded into the furnaces. There are a number of different furnace setups depending on the quality of the aluminium scrap. From the furnace the melted aluminium is tapped for either direct casting or sent to another furnace where alloys can be made. In this process the aluminium is also refined to remove the remaining impurities in the product. Typically, the aluminium recycling process only uses around 5% of the energy needed for the virgin aluminium production, as the alumina conversion in virgin production is held responsible for the majority of the energy consumption (Damgaard et al., 2009).

Recycling of paper and cardboard waste

There are two main groups of reprocessing of paper and cardboard into pulp: mechanical and chemical-mechanical re-pulping (Merrild et al., 2009). Mechanical re-pulping consists of re-pulping, mechanical removal of large contaminants, refining by washing, sorting, and milling, mechanical removal of finer contaminants, thickening and optional bleaching, and final drying. Mechanical pulping is used for production of paper of lower grades. Chemical re-pulping, in addition to the steps listed above, includes also de-inking to brighten up the pulp for use in higher value paper grades such as printing and copy paper for which such parameter is important. The process of de-inking involves a chemical step where agents are added to free the ink from the pulp and a mechanical step of flotation where the removed ink is finally physically separated from the rest of the pulp. De-inking normally occurs after the refining step.

Recycling of plastic waste

Plastic waste can either be recycled through mechanical/physical (also referred to as 'material recycling' in literature) or chemical recycling. With the former, the molecular structure of plastic is preserved, while with the latter the polymer chains are converted into its oligomers, monomers or other basic chemicals such as carbon monoxide, carbon dioxide, methane, and hydrogen (Delva et al., 2019). In a recent publication, Collias et al. (2021) distinguishes material recycling into mechanical and physical recycling (i.e., dissolution or solvent-based recycling), and chemical recycling into depolymerisation, gasification and pyrolysis as summarised in Figure 1. As displayed in Figure 1, these recycling processes can be further classified into polymer loops, monomer loops, and molecular loops. Material recycling belongs to the polymer loop as the output obtained from this reprocessing is the purified form of the same input plastic waste that was originally fed into the process (Collias et al., 2021). Depolymerisation is classified as a monomer loop as the input plastic waste is converted into its constitutive monomers, while pyrolysis and gasification are classified as molecular loops as the input plastic waste is converted into smaller molecules or group of molecules (e.g., carbon monoxide, carbon dioxide, hydrogen, methane) prior to further reprocessing into monomers/polymers (Collias et al., 2021).

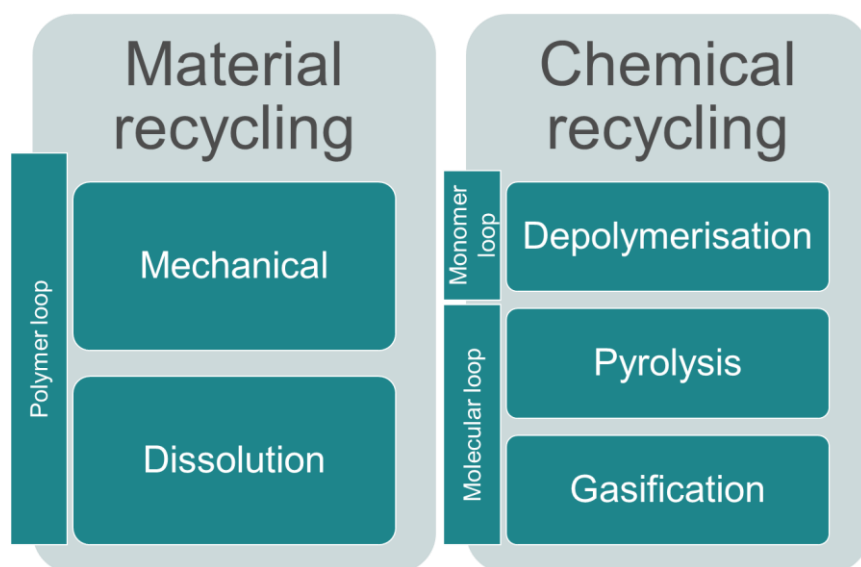


Figure 1: Classification of recycling technologies for plastic waste, adapted from Collias et al. (2021).

It is worth mentioning that plastic waste may also follow other routes for recovery, which might involve material recovery or recycling to a certain extent along with energy recovery. Notably, it can be either pelletized or crushed and substitute coke and pulverized coal in blast furnaces for steel and iron production (Ogaki et al., 2001). Plastic waste can be utilised in blast furnaces to produce heat via combustion or as reducing agent in gasification due to the production of reducing gases, as carbon monoxide and hydrogen, that reduce the iron ore into iron oxides (Devasahayam et al., 2019). When utilised in this application, both thermoplastics and thermosets can be employed. This type of application benefits the iron and steel making industry as plastic can increase the productivity, reduce the coke ratio, decrease the process temperature and, therefore, the energy inputs required, and cut both harmful and CO₂ emissions (Devasahayam et al., 2019). Yet, only a share of the input plastic feedstock is incorporated in the final iron cast as the remaining share is turned into energy and off-gas.

Mechanical recycling of plastic waste

Mechanical recycling (under material recycling; Figure 1) is only suitable for thermoplastic materials as thermoset plastic cannot be re-melted. This recycling technology involves physical processes that can occur either at all or multiple times, and are as follows:

cutting/shredding into small flakes; contaminant separation (removal of impurities such as paper and dust via a cyclone); floating (separation into different types of plastic according to their density); milling (for separate, single-polymer plastic); washing and drying; agglutination (after the addition of pigments or additives, the product can either be stored and sold at a later stage or sent to further processing); extrusion (extrusion to strand); pelletizing; and, quenching (water cooling to granulate the plastic and sell it as a final product) (Al-Salem et al., 2009).

Mechanical recycling is highly dependent on the quality of the input waste, which can be reprocessed into the same product or a similar one if the quality of the input is sufficiently high after collection and sorting. Furthermore, mechanical recycling can highly be affected by the presence of contamination of one polymer or another polymer material. Indeed, if such contaminations are present, the mechanical properties of the final products are hindered; this can also result into reprocessing problems (Delva et al., 2019). As for mechanical properties, such as crystallinity and mechanical strength, operational parameters (i.e., high temperatures and shear forces) during melting and reprocessing can cause mechanical and thermal degradation of the polymers, affecting the polymer chain length and distribution (European Bioplastics, 2020). Other possible problems related to mechanical recycling are (i) the lack of adequate capacity to process complex materials into their purified components for their subsequent use, (ii) mixing different polymers, and (iii) the degradation of the plastic during its lifetime due to long-time exposure to external factors (Crippa et al., 2019; Ragaert et al., 2017).

It should be mentioned that a number of recently published scientific studies and articles consider dissolution (solvent-based purification) as a type of physical (material) recycling because the structure of the polymer is not altered (as mentioned earlier, this is the main discriminator between chemical and mechanical process). The dissolution process consists in using heat and solvents to dissolve the plastic into a solution of polymers and additives from which it was originally made from. In the final step, new additives are added to produce the recycled plastic.

Chemical recycling of plastic waste

Chemical recycling (sometimes also referred to as “advanced recycling”) is a process where polymer chains are degraded into monomers or other basic chemicals. Chemical recycling is considered to be less impacted by the presence of impurities and mixed plastics. On this basis, plastic waste streams that cannot be currently recycled via mechanical recycling could be processed via chemical processes. Furthermore, the loss of quality that occurs during mechanical recycling can be overcome with chemical recycling, as impurities can be removed and polymers equivalent to the virgin ones can be potentially obtained opening the possibility for infinite recycle loops. Regardless of the environmental superiority of one technology over the other (still under study), it appears clear that whenever possible synergies between these two types of technologies should be envisioned (Collias et al., 2021). Based on current state-of-the-art knowledge (Ragaert et al., 2017), chemically recycled polymers appear to be more expensive than the virgin ones as the plants are not yet optimised cost-wise and would function only at large-scale. Yet, this may change with technology development and economy of scale. More details on the technologies for chemical recycling are given in section 3.2.

Recycling of compostable plastic waste

Compostable plastic refers to plastic materials that can be converted by microorganisms into natural substances such as water, CO₂ or methane, and biomass, without the need of additional additives. The most important feature that distinguishes compostable plastics from non-compostable plastic is the degradation time, which can span from several months to several years for the former and last for hundreds of years for the latter. It is noticeable to mention that the degradation time strongly depends on the conditions of exposure during disposal (e.g., temperature, pressure). It is important to highlight that compostable does not mean necessarily bio-based (i.e., produced from renewable resources), since

there are (both fully and partially) fossil-based plastics that are compostable, such as polybutylene adipate terephthalate (PBAT) and polycaprolactone (PCL) (Di Bartolo et al., 2021).

Compostable plastics can be recycled similarly to plastic materials via mechanical and chemical recycling, but additionally also via biological recycling since they can be broken down into simple compounds through microbial action. Mechanical recycling is rarely used on a commercial scale for compostable polymers. It can cause thermal and mechanical degradation of the material leading to a lowering of molar mass or cross-linking compromising certain properties of the products, such as the tensile strength, tensile strength at break, melt flow index, impact strength or the thermal stability (all decreasing normally with increasing number of extrusion cycles). Possible solutions might come from composites of compostable polymers as additives to a neat material (natural fillers) (Sikorska et al., 2021). Also, compostable plastics might cause problems when they interfere (unintentionally) in the mechanical recycling of non-compostable plastics (e.g., polypropylene, PP), decreasing mechanical and thermal properties due to immiscibility and incompatibility of blends. Thus, it is important to separate compostable plastics properly. This can be achieved by employing novel processes to detect compostable polymers (e.g., in the PP recycling process) such as the Fourier transform infrared spectroscopy. It is important to highlight that mechanical recycling of bio-based polymers (i.e., chemically identical to their fossil-based counterparts – referred as “drop-in”) does not influence on recycled petrochemical polymer properties and can be managed in the conventional processing and recycling streams without adaptation.

Chemical recycling is an interesting end-of-life option for materials that cannot be mechanically recycled, and depends on the affordability of processes and the efficiency of catalysts (see section 3.2). It has not been implemented for large-scale postconsumer recycling. This option applied to compostable polymers includes thermal and chemical processes, being chemolysis the most common one. Chemical depolymerisation requires reactants (such as solvents to break the bonds of polymers), heat and catalysts, and a clean and homogeneous polymer waste, such as single stream of polylactic acid or polyhydroxyalkanoate (PLA or PHA, respectively). Yet, it is almost impossible to differentiate and separate compostable polymer materials (e.g., PLA) from non-compostable plastics (e.g., PET) using visual discrimination since they are very similar. The use of labels is crucial for overcoming this issue; by employing them, PLA can be collected as a separate waste stream and recycled in a profitable way. Thermochemical processes (and in general waste-to-energy) with energy recovery have already been implemented for compostable plastics (Di Bartolo et al., 2021).

Biological recycling includes composting, either industrial or domestic (home-composting), and anaerobic digestion (either wet or dry) . The requirements on industrial compostability of plastic packaging and non-packaging plastics have been introduced by two harmonised standards (EN 13432:2000 and EN 14995:2006, respectively) that set the criteria (disintegration, thickness, chemical composition, heavy metals, etc.) for assessing the suitability. The testing methodologies to evaluate biodegradability are also set in official standards (e.g., EN ISO 14855) (Di Bartolo et al., 2021). For home composting, it is yet to be specifically described by EN harmonised standards (i.e., EN 17427 and EN 17428), since conditions are not as controlled as in industrial composting and they vary greatly depending on the installation. The use of labels and logos that identify compostable products are common, and different independent certification bodies carry out certification (Hann et al., 2020). Thus, those products may be collected with bio-waste, if legally allowed in the Member State³, and directed to biological recycling without removing the remains of their content. It is important to highlight that each component of the compostable product should be compostable, and materials combining non-compostable and compostable polymers should not be collected together with bio-waste. The use of compostable additives in their composition is also required. While anaerobic digestion may

³ For example in Germany only compostable plastic bags are allowed to be collected with biowaste (no other compostable packaging).

be appealing for compostable plastics because of the energy recovery (Di Bartolo et al., 2021), as opposite to direct composting, the effective degradation of the material in the digester is not guaranteed (it depends on the retention time) and the plastics may be sorted out prior to entering the digester to avoid clogging or failures.

Recycling of wood waste

Wood waste is typically recycled via mechanical processes. The collected wood waste is transported to the recycling plant where, at its arrival, the input stream is weighted and a visual inspection is performed to assess the degree of quality of the waste and the type of wood. Then, the wood waste is cleaned, shredded into chips and then flakes, dried, and cleaned again. The flakes are screened according to their size, which depends on their future use. For example, larger pieces are used to make furnishings, whereas small wood fibres are utilised to produce animal beddings. It is noticeable that 80-90% of the flakes are later used to produce particleboard. Specifically, particleboard is produced from wood waste suitable for recycling, which is shredded and dried. Then, the dried chips are pressed and organic resins are added to form a solid mat, known as particleboard (Faraca, Edjabou, et al., 2019).

Wood waste can undergo also biological recycling, specifically through composting. When composting wood waste, it is necessary to add to the waste a source of moisture and nitrogen. Before being composted, in general, the waste is shredded; this is particularly important for wood waste as it is inherently difficult to break it down due to the high levels of lignin (WRAP, 2007). Furthermore, wood waste can also contain preservatives and biocides that further slowdown the composting process, ultimately affecting the technical feasibility and economic cost of the plant itself (WRAP, 2007).

Recycling of textile waste

Prior to its recycling, textile waste undergoes a first pre-treatment which consists in identifying the textile materials. This is considered as the most challenging task since fabrics and textiles have a complicated structure. The most promising technologies implemented to perform this task are near-infrared spectroscopy and chemometric, and nuclear magnetic resonance (Damayanti et al., 2021). After being sorted, textile waste can be recycled via mechanical, chemical, biological, and thermal processes.

Mechanical recycling is among the easiest and cheapest technologies. It can be classified into several different methods according to the degree of breakdown that the textile waste has to undergo, such as fibre, fabric, polymer, and monomer recycling (Damayanti et al., 2021). For natural fibres (such as wool and cotton), textile waste is shredded, blended and combed, and, finally, spun into a yarn. However, the yarn obtained is generally of lower quality relative to the corresponding virgin one as shredding results in short fibres of lower quality and strength. Because of this, recycled cotton needs to be mixed with virgin cotton fibres to improve its quality (Damayanti et al., 2021). For synthetic fibres (such as polyester), the process consists in shredding the textile waste, granulate it and form plastic pellets (nurdles). The plastic pellets undergo further processes so that they can be melted, extruded and spun into new fibres. However, textile waste contains a large variety and amount of different fibres, which cannot be effectively recycled via mechanical processes. Finally, specifically for the case of blended fabrics (such as polyester-cotton), mechanical recycling seems to cause deterioration of the natural fibres, while not provoking the same detrimental effects on the synthetic ones. For this, chemical recycling appears as a possible alternative treatment (Damayanti et al., 2021).

Chemical recycling of textile waste comprehends pyrolysis, gasification, ammonolysis, and glycolysis, which are described as follows (Damayanti et al., 2021):

- Pyrolysis: this technology does not require pre-treatment and, therefore, can accept multi-material textile waste. The outputs of this process are syngas (used as direct fuel or as raw material to produce hydrocarbon and alcohols); a liquid product (mono-polyaromatic and oxygen compounds containing hydrocarbons such as

alcohols, aldehydes, ketones, and carboxylic acids), which composition depends on the operational temperature; and a solid product (the char can be used as a primary filler or a hybrid filler, or as graphene oxide in concrete composite applications). However, pyrolysis requires high temperatures and, therefore, high energy consumptions (Damayanti et al., 2021).

- Gasification: this technology can potentially be fed with mixed textile waste. The desired product of the process is syngas, which includes compounds such as hydrogen, CO₂, carbon monoxide, and methane, while ethylene and ash can be formed as by-products. As for pyrolysis, the reactions occurring during gasification require high temperature, resulting in significant energy consumptions (Damayanti et al., 2021).
- Ammonolysis: this process is the primary depolymerisation method used for nylon 6,6 and nylon 6 mixtures. The process consists in heating the mixture with ammonia at high temperature and pressure to obtain monomers and water, the latter inhibiting the conversion of amides formed as intermediates to nitriles. Therefore, the water is removed to allow the full conversion into nitriles. This may be achieved by passing ammonia through the reaction zone and then passes to a subsequent chamber as the ammonia is not condensed with the monomers (McKinney, 1994). The downsides of this process are that it generates a mixture of primary and secondary amines, it applies a toxic solvent (i.e., ammonia), and requires high temperature and pressure (Damayanti et al., 2021).
- Glycolysis: this process can be used to convert large molecules into small molecules and it is largely used for PET fibres and polyurethane. The process is characterised by low energy consumption, but it has low selectivity and can result in a slow process if no catalysts are used. Efforts are focused on developing an eco-friendly and high-performance catalyst (Damayanti et al., 2021).

Biological recycling of textile waste consists in enzymatic hydrolysis. The recycling process is preceded by a pre-treatment, which is a crucial step as it affects the yield of the recycling process. The pre-treatment can be done employing acid, alkaline or ionic liquids. Acid pre-treatments utilise sulphuric and phosphoric acid and can break the polymeric structures into monomers, thus increasing the availability of cellulose and, therefore, increasing biodegradability. However, this process can lead to the production of side products (such as furfural), it is costly and requires corrosion-resistant equipment. Alkali pre-treatments use bases such as calcium, ammonium hydroxides, sodium, and potassium. This type of pre-treatment improves the solubilisation of lignin and decreases the crystallinity of cellulose by increasing its digestibility, resulting in high glucose yields and reduced formation of fermentation inhibitors. Finally, ionic pre-treatments are considered to be more environmentally friendly as they dissolve the cellulose at moderate temperature without degrading the solvent or the cellulose. After the pre-treatment, hydrolysis occurs in the presence of large amounts of water and cellulases (catalyst), which are usually a mixture of three types of enzymes each of which has a different role in the enzymatic process: (i) endoglucanases degrades the cellulose chain by increasing the number of accessible end parts; (ii) exoglucanases degrades cellobiose at both ends of the chain, and (iii) beta-glucosidases depolymerises the disaccharide cellobiose into monosaccharides units. Afterwards, textile waste as cotton, polyester, nylon, and silk can be degraded through five different fermentation processes: (i) simultaneous saccharification and fermentation to convert sugars into ethanol; (ii) separated hydrolysis fermentation to produce bioethanol; (iii) semi-simultaneous saccharification and fermentation to produce bioethanol; (iv) consolidated bioprocessing; and, (v) submerged fermentation (Damayanti et al., 2021). It should be noticed that biological recycling converts textile into products other than textiles (see above: bioethanol, cellulose, and derivatives such as biogas).

Thermal recycling of textile waste occurs via hydrothermal processes. The hydrothermal process is among the most promising technologies to degrade carbon-polymer waste and organic components into a liquid, solid and gas phase. These can be later used to produce industrial chemicals. Hydrothermal recycling does not require any pre-treatment and

utilises water as the main constituent of the reaction. According to the method employed to utilise water, five different processes can be distinguished: (i) hot water extraction, (ii) pressurised hot water extraction, (iii) hot liquid water treatment, (iv) hydrothermal carbonisation, and (v) hydrothermal liquefaction. The main drawbacks of the hydrothermal process are high temperatures and pressures (despite being relatively lower than the ones required for pyrolysis and gasification) and low reaction times (Damayanti et al., 2021).

Recycling of bulky waste

Bulky waste refers to all MSW that cannot be managed through the collection system in place due to size, shape or weight issues. Being bulky waste highly heterogeneous, this is sorted at dedicated centres to segregate recyclables such as plastics, paper and cardboard, metals, wood waste through manual sorting, manual sorting belts and/or automatic processes (e.g., air flow or optical separators) (URBANREC, 2020). As follows, examples based on the URBANREC project are provided (ECOFRAG, 2019):

- Wood from furniture undergoes manual separation, cutting and fragmentation and it is then used for wood plastic composite applications.
- Foam mattresses can be composed of polyurethane or latex. If present, springs and metal parts are detached and sold, then the foam is fragmented to obtain a product of high quality that can be used for fabricating new mattresses, while polyurethane foam can be converted into adhesives through glycolysis, and textile parts can be used to produce textile applications, such as needlefelts and composites.
- Tyres are fragmented, and are then separated into metal parts, textile parts and rubber. Textile parts can be used in textile applications.
- Jute carpets and polyamide carpets are separated into their front and back parts. As back parts contain great amounts of glue, only front parts can be used for textile applications.
- Artificial grass has a similar composition to that of carpets. This waste is separated into a front and a back part; the front parts composed of polyethylene (PE) can be used for synthetic textile applications.

Composite materials

Composites materials are polymers that are reinforced with high stiffness and strength fibers, such as carbon fibers and glass fibers (Cembureau & EuCIA, 2019). Composite materials can be recycled through either mechanical, thermal, or chemical technologies.

Mechanical recycling can be applied to glass fibres (Technology readiness level - TRL - 9) and carbon fibres (TRL 6-7). This technology is effective, it requires low energy inputs and runs at low costs. Yet, this process drastically reduces the value of the recycled materials, namely short fibres and ground matrix powder that can be used as reinforcement or fillers, respectively (Assocompositi, 2022; WindEurope, 2020).

The thermal technologies that can be employed for composite materials' recycling can be classified in mature (i.e., pyrolysis) and emerging (i.e., gasification, depolymerisation, and high voltage pulse fragmentation). Pyrolysis allows the recovery of the fibres as ash and of the polymer matrix as hydrocarbon products. The recycled outputs can be used as additives and fillers, yet the value of these recycled materials decreases due to the high temperatures employed in the process. Further, pyrolysis requires high investment and running costs and is currently economically viable for carbon fibres only. High voltage pulse fragmentation (TRL 6) is an electro-mechanical process that effectively divides fibres from matrices with the use of electricity. Compared to mechanical recycling, high voltage pulse fragmentation allows obtaining longer and cleaner fibres that, hence, have higher value. However, only short fibres can be recovered at high quality, which requires high levels of energy. Depolymerisation (TRL 7) is a two-step process: in the first step, CO₂ is used to recover the fibres, while in the second step organic residues are removed while recovering the resin. Gasification (TRL 5-6) has the peculiarity of being a process able to treat mixed

materials; however, the fibres obtained as recycled material are more degraded than the ones obtained through pyrolysis (Assocompositi, 2022; WindEurope, 2020).

Finally, composite materials can be also treated through cement co-processing (TRL 9). The glass fibres can be used as a component of cement mixes (or cement clinkers), while the polymer matrix is burned as fuel for the process (refuse-derived fuel) reducing the carbon footprint of cement production (Assocompositi, 2022; WindEurope, 2020). As the process both recovers energy and partially mass, it is not yet clear from the current definition of recycling whether the output of such a technology can be defined as recycled (and the related processing as recycling), to some extent.

3.2 Emerging multi-output recycling processes

This section summarises the literature of studies dealing specifically with emerging multi-output processes (earlier introduced in the section 3.1) for MSW treatment. Multi-output technologies are here intended as technologies that convert input-waste into multiple products, e.g., chemicals, materials, fuels and energy. To this category belong different types of emerging technologies, which could be based on chemical, biological or thermal processing. In the following sub-sections we focus specifically on chemical recycling (mainly treating polymeric waste such as plastic, wood and tyres) and advanced biological recycling (i.e., biorefineries). Notice that the scope of the review is on literature studies published after year 2010 and that provide quantitative technology data to detail the processes in terms of input of resources, materials, energy and outputs of materials, energy and emissions based on either primary or secondary technology data. Studies providing a general overview/review of pathways and processes, while considered as valuable background information (e.g., to define qualitatively the processes), have not been considered for retrieving quantitative data in the following sections. Furthermore, studies focused on very specific agro-industrial biomasses or biomass residues (e.g., cassava, spent coffee ground, citrus waste) or lab-based experimental studies have also not been considered in the review.

Chemical and other advanced recycling of plastic waste

Chemical recycling can broadly be defined as the collection of thermal and/or chemical techniques that break down polymeric feedstock (e.g., plastic, tyres, but also wood or food waste) into its constituent parts, i.e., monomers, oligomers or heterogeneous hydrocarbon mixes for different applications (e.g., production of virgin-like polymers, chemicals, or fuels). Notice that, while chemical recycling is largely employed in reprocessing plastic waste, the literature review conducted on life cycle assessments (LCAs) of chemical recycling also revealed studies concerning other waste streams, i.e., food and wood waste, which were also included. Finally, notice that as thermal techniques are included within chemical recycling, hydrothermal carbonisation is also reported as a possible re-processing technique for food waste in Table 2.

With respect to plastic waste, the use of chemical recycling is often seen as an alternative to mechanical recycling, which suffers from technical downsides such as its inability to treat contaminated or low quality streams, or to separate the additives that are present in plastic waste, or the limited number of recycling loops that a polymer can undergo without showing structure degradation. Plastic chemical recycling is therefore an “umbrella” term that comprises a number of technologies, which can be divided into different categories depending on the level of decomposition that the polymer will be subject to: (i) chemical depolymerisation; (ii) thermal depolymerisation; and, (iii) cracking. Solvent-based purification, while often classified as chemical recycling (some literature classifies it as a type of physical (material) recycling as mentioned in section 3.1), does not alter the polymer structure of the waste. On top of the above-mentioned consolidated chemical recycling processes, a number of emerging technologies are currently investigated. A summary of the different recycling technologies for plastic waste is shown in Table 1.

Table 1. Summary of different recycling technologies for plastic waste. Adapted from Zero Waste Europe (2019). PA: polyamide; PC: polycarbonate; PE: polyethylene; PET: polyethylene terephthalate; PHA: polyhydroxyalkanoate; PLA: polylactic acid; PMMA: poly(methyl methacrylate); PP: polypropylene; PS: polystyrene; PU: polyurethane; PVC: polyvinyl chloride.

Technology	Feedstock	Output	Decontamination	Can treat mixed plastic?	Maturity
Mechanical recycling	PE, PET, PP, PS	Polymer	No	Yes	Industrial scale
Solvent-based purification (Dissolution)	PE, PP, PVC, PS	Polymer	Yes	No	Pilot scale
Chemical depolymerisation	PET, PU, PA, PLA, PC, PHA, PEF	Monomer	Yes	No	Pilot scale
Thermal depolymerisation (pyrolysis)	PMMA, PS	Monomer	Yes	No	Pilot scale (TRL 9)
Cracking (pyrolysis or gasification)	Plastic mix	Hydrocarbon mix	Yes	Yes	Pilot scale
Pyrolysis with in-line reforming	Different types of plastic	Hydrocarbon mix		Yes	Development stage (TRL 4)
Microwave-assisted pyrolysis		Hydrocarbon mix		Yes	Laboratory and pilot scale (TRL 4)
Plasma pyrolysis		Monomer		Yes	Laboratory scale (TRL 4)
Plasma gasification	All types of plastic	Hydrocarbon mix		Yes	Commercial scale (TRL 8)

Solvent-based purification (Dissolution)

This type of technologies works by dissolving the polymer in a specific solvent followed by the removal of additives and other contaminants through filtration or phase extraction. The purification process does not change the polymer structure itself but does bring physical and thermal stress to it. Additionally, residual additives or solvents that are not removed during the purification process may be incorporated in the output, which may affect the material quality. As mentioned earlier, while being often classified under chemical

recycling, it has been recently argued that solvent-based purification should be considered as physical recycling, thus equivalent to mechanical recycling, because it goes down to the polymer level, i.e., not to its monomers or hydrocarbon mix (Collias et al., 2021; Zero Waste Europe, 2019).

Chemical depolymerisation

Depolymerisation is the reverse process of polymerisation, i.e., the decomposition of a polymer into its monomers. The process usually employs a solvent, which also acts as a reagent, and it is carried out in the presence of heat and a catalyst. Different types of depolymerisation processes exist, depending on the type of solvent used. Furthermore, sub- or super-critical fluids can be used as reaction media for depolymerisation, in which case the reaction proceed rapidly and selectively (Ragaert et al., 2017). Figure 2 summarises the different types of depolymerisation reactions.

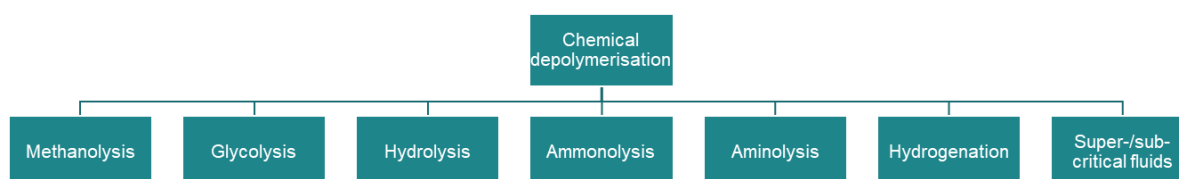


Figure 2. Schematic summary of chemical depolymerisation reactions.

Thermal depolymerisation

This type of processes involve heating the polymer under specific conditions. They can be classified into four main technologies, namely: (i) Pyrolysis; (ii) Gasification; (iii) Catalytic cracking; and, (iv) Hydrogenation (hydro-cracking).

Pyrolysis is the process whereby a hydrocarbon mix or, less commonly, a monomer is obtained from heating the polymer between 400-600°C in the absence of oxygen. Water may be present in the pyrolysis process (hydro-pyrolysis). In principle, pyrolysis can handle any type of plastic feedstock; however, maintaining a certain constant quality of the output may require selected pre-treatments and sorting of the input to reach a certain feedstock quality. From the literature it appears that polyolefins are best suited to this application.

In gasification processes, the polymer is also heated but in this case at higher temperatures (700-1500°C) and in the presence of a controlled amount of both oxygen and water (Zero Waste Europe, 2019). The main product of gasification is syngas ($H_2 + CO$), plus small amounts of other gases such as methane and CO_2 . Similarly to pyrolysis, gasification can virtually handle any type of plastic feedstock.

A third group of processes within the thermal depolymerisation family falls into catalytic cracking, which involves adding a catalyst to a pyrolysis process thereby increasing the rate at which the pyrolysis reactions proceed while reducing the process temperature to 300-350°C (Solis & Silveira, 2020). At the same time, the yield of products with higher added value can be increased by employing the right catalyst. The main drawbacks of this technology lies in the cost of the catalyst and its tendency to suffer from poisoning by contaminants present in the mixed waste plastic stream.

Finally, hydrogenation, also known as hydro-cracking, involves the addition of hydrogen to the cracking process at elevated pressure up to 70 atm (Solis & Silveira, 2020), which increases the yield of certain products. The biggest obstacles in implementing this technology are the cost of hydrogen and high capital and operational expenditures (Manžuch et al., 2021).

Among the emerging technologies, we acknowledge: (i) microwave-assisted pyrolysis, (ii) plasma pyrolysis, (iii) pyrolysis with in-line reforming, and (iv) plasma gasification. Microwave-assisted pyrolysis involves the addition of dielectric material or absorbents (e.g., activated carbon, graphene or silicon dioxide) to the plastic waste. This technology can solve two main drawbacks of conventional pyrolysis, namely high energy requirements

and slow reaction times. Yet, this emerging technology suffers from imprecise temperature measurements, difficult to disperse the microwaves properly, non-uniformity of the heating process, it requires large amounts of feedstock volumes, and, finally, it is still unknown the role in the heating efficiency of the dielectric material as well as the efficiency of the microwave design. The technology has been developed at laboratory and pilot scale only and, therefore, it has been assigned a TRL of 4 (Manžuch et al., 2021).

Plasma pyrolysis exploits the thermochemical properties of plasma into conventional pyrolysis to break down entirely plastic waste into monomers to produce syngas (composed of CO and H₂ mainly) and small amounts of higher hydrocarbons. The process is very fast (0.01-0.05 seconds) and requires high temperatures (1730-9730°C). The high temperatures employed are able to decompose toxic compounds that may be present in gas and prevent the formation of HCl. The technology has mostly been applied for hazardous waste and not for recycling of plastic, where it has been only investigated at laboratory scale (TRL 4) (Manžuch et al., 2021).

Pyrolysis with in-line reforming optimises the production of tar-free hydrogen from plastic waste. The process entails the pyrolysis of plastic in the first reactor and the reforming of the pyrolysis products in the subsequent one. Compared to gasification, the process requires lower temperatures (500-900°C) with corresponding decreased costs in production, and avoids the contact between the impurities in plastic waste and the catalyst, thus minimising the costs of catalysts in the reforming step. The main disadvantage of this technology is the absence of its application at the industrial scale (TRL 4) (Manžuch et al., 2021).

Plasma gasification exploits the heat produced by thermal plasma, which is usually generated by direct current non-transferred arc plasma torches. The reaction time ranges between 30 minutes to 3 hours while the reaction temperatures can reach 14000°C, and the process of the plasma gasification is highly dependent on the flow rates of the oxidant, plasma gas, and steam streams. The process can tolerate low-quality feedstock while resulting in high purity syngas with low tar content. Due to the high investment and operating costs, on top of high energy intensity, needs for adequate waste sorting systems and limited understanding of the process, the process cannot be currently scaled up to an industrial scale (TRL 8) (Manžuch et al., 2021).

Outcome of the literature review on LCA of chemical and other advanced recycling

Twelve LCA studies that provide disaggregated input-output inventory of chemical recycling technologies for plastic waste reprocessing have been found in the scientific and technical literature. The assessed technologies include pyrolysis, gasification, hydrocracking, chemical depolymerisation and solvent-based purification, with TRLs ranging from 4 to 7. Different input plastic waste fractions were considered, but most of them deal with mixed plastic waste, polyolefins, PET or PLA. As for the outputs from the chemical recycling process, a wide range of products have been identified. Hydrocarbon mixes are typically the main products of thermochemical technologies, i.e., pyrolysis, gasification and hydro-cracking, while the polymer's constituent monomers are most often the products of depolymerisation and solvent-based purification technologies. On top of these studies, Table 2 includes assessments on chemical recycling of food waste (Albizzati et al., 2021a) and wood waste (Ajao et al., 2021; Papageorgiou et al., 2021) by employing pyrolysis, chemical extraction and hydrothermal carbonisation characterised by TRLs spanning from 3 to 7. Recycling food waste through pyrolysis or hydrothermal carbonisation leads to the production of biochar (amending material) and hydrochar or coal-like char (multiple utilisations, e.g., as fuel or amending material), respectively (Albizzati et al., 2021a). In the study by Ajao et al. (2021) a mass balance and techno-economic evaluation of chemical extraction of wood waste and production of tannins extracts, lignin-based polyurethane foam and cellulose-based composites was performed. While the technology is at a lab-scale, the analysis was complemented with the AspenPlus simulator to estimate full-scale production flows and costs. Little information is however provided on the outputs quality. Finally, Papageorgiou et al. (2021) studied the production

through pyrolysis of biochar (25% of the wood waste in input) and syngas (75% of the wood waste in input) from wood waste for soil remediation; literature data and simulations were used by the authors to perform the mass-balance exercise (see Table 2).

Table 2 summarises the reviewed studies with details on the type of waste treated, the chemical recycling technology and its TRL, as well as the main inputs/outputs of the process. Table 2 is followed by a brief summary of the environmental assessment results, when available from the studies analysed.

Table 2. Overview of literature studies providing technical data on chemical and other advanced recycling technologies treating plastic waste, food waste and wood waste as input-feedstock. MPO: mixed polyolefins; MPW: mixed plastic waste; PE: polyethylene; PET: polyethylene terephthalate; PLA: polylactic acid; PP: polypropylene; PS: polystyrene; TRL: Technology Readiness Level (estimated, when possible). Data are expressed on a wet weight basis, unless differently specified.

Study	Input waste	Technology and TRL	Outputs (kg/kg waste input)
Al-Salem et al. (2014)	MPW	Low temperature pyrolysis, TRL 6-7	Gases (C ₃ -C ₄) 0.147 kg/kg Liquid (Naphtha) 0.265 MJ/kg Wax 0.448 kg/kg CaO 0.04 kg/kg CaCl ₂ 0.017 kg/kg Steam 1.48 MJ/kg Sand and coke 0.076 kg/kg Waxy filter (deposit) 0.046 kg/kg
Al-Salem et al. (2014)	MPW	Hydro-cracking, TRL 6-7	Syncrude 0.822 kg/kg Methane 0.09 kg/kg HCl 0.005 kg/kg CaCl ₂ 0.0041 kg/kg Solid waste 0.05 kg/kg Residue 0.066 kg/kg
Civancik-Uslu et al. (2021)	PP, PE, PS, MPO	Hydro-cracking, TRL 6	Dependant on input waste. Mainly hydrocarbon mix.
Cosate de Andrade et al. (2016)	PLA	Chemical depolymerisation, TRL 4	PLA 0.97 kg/kg
Cossu et al. (2017)	MPW	Gasification, TRL 6-7	Electricity 24% eff. LHV Heat 22% eff. LHV
Demetrious & Crossin (2019)	MPW	Two-stage pyrolysis-	Syngas 0.82 kg/kg Residue 0.048 kg/kg

		gasification, TRL 6-7	
Faraca, Martinez-Sanchez, et al. (2019)	MPW	Pyrolysis, TRL 6-7	Pyrolysis oil 0.66 kg/kg Steam 0.103 kg/kg Residues 0.069 kg/kg
Jeswani et al. (2021)	MPW	Pyrolysis, TRL 6-7	Pyrolysis oil 0.637 kg/kg Steam 1.2 MJ/kg Electricity 0.28 MJ/kg Residues 0.07 kg/kg
Khoo (2019)	MPW	Pyrolysis, TRL 6-7	Diesel 0.65 kg/kg
Khoo (2019)	MPW	Gasification, TRL 6-7	Ethanol 0.278 kg/kg
Maga et al. (2019)	PLA	Chemical depolymerisation, TRL 4	PLA 0.8 kg/kg
Maga et al. (2019)	PLA	Solvent-based purification, TRL 4	PLA 0.9kg/kg
Perugini et al. (2005)	MPO	Low temperature pyrolysis, TRL 6-7	Gas fraction 0.147 kg/kg Heavy fraction (waxes) 0.448 kg/kg Light fraction (liquid) 0.265 kg/kg CaO/CaCl ₂ 0.057 kg/kg Sand 0.076 kg/kg Residues 0.046 kg/kg
Perugini et al. (2005)	MPO	Hydro-cracking, TRL 6-7	Syncrude 0.822 kg/kg Methane 0.09 kg/kg HCl 0.005 kg/kg CaCl ₂ 0.0041 kg/kg Residues 0.11 kg/kg
Shen et al. (2010)	PET	Glycolysis, TRL 4	PET flakes 0.98kg/kg Residue 0.02 kg/kg
Shen et al. (2010)	PET	Methanolysis, TRL 6-7	PET flakes 0.9kg/kg Residue 0.1 kg/kg

Shonfield (2008)	MPO	Pyrolysis	Gas fraction 0.05 kg/kg Heavy fraction (waxes) 0.45 kg/kg Light fraction (liquid) 0.26 kg/kg Residue 0.18 kg/kg
Shonfield (2008)	PP, PE, PS	Pyrolysis	Diesel oil 0.79 kg/kg Residue 0.16 kg/kg
Ajao et al. (2021)	Wood waste	Chemical extraction TRL 3	Cellulose-rich residue 0.6-0.7 kg, Effluent 0.03-0.05 kg, Lignin 0.1-0.2 kg, Tannin 0.03-0.06 kg
Albizzati et al. (2021a)	Food Waste	Pyrolysis TRL 6-7	Biochar (0.23 kg), syngas (0.77 kg)
Albizzati et al. (2021a)	Food Waste	Hydrothermal carbonization TRL 3-4	Coal-like char (0.45 kg)
Papageorgiou et al. (2021)	Wood waste	Pyrolysis TRL 2-3	Biochar (0.25 kg), syngas (0.75 kg)

Environmental performance of chemical recycling

In terms of its environmental performance, there is not a clear trend indicating that chemical recycling is more or less favourable compared with other management options for plastic waste such as mechanical recycling or energy recovery. It appears clear that further research is required to support any conclusion with respect to such treatment hierarchy. Often, the ranking between technologies appears to be strongly dependent upon the type of waste feedstock processed (i.e., quality, purity, contamination, etc.), which call for waste fraction-specific investigations. These shall investigate on how and where chemical recycling could well complement mechanical recycling, rather than supporting a supremacy of one route over the other (as the feedstock processed are often different, i.e., the two technologies fulfil different functions/services).

With respect to the specific studies short-listed within our review, the study by Al-Salem et al. (2014) showed that plastic waste management scenarios including hydro-cracking and pyrolysis had lower global warming potential (GWP) than those including just mechanical recycling. A similar conclusion was given by Civancik-Uslu et al. (2021), who argued that hydro-cracking performed better than incineration with energy recovery and mechanical recycling for the analysed environmental impacts (resource consumption, GWP, terrestrial acidification). However, mechanical recycling performed better than hydro-cracking when the recycling products can substitute virgin materials in a 1:1 ratio, highlighting that the quality of the derived products is critical to the LCA results. The study by Shonfield (2008) goes in the same direction since it demonstrated that the two pyrolysis scenarios that were assessed had broadly comparable environmental performance in most impact categories and generally performed better than landfill and incineration but worse than the mechanical recycling options. Similarly, Faraca, Martinez-Sanchez, et al. (2019) concluded that advanced mechanical recycling provided the largest savings in the highest number of environmental impact categories, including GWP. In the same study by Faraca, Martinez-Sanchez, et al. (2019), conventional mechanical recycling (a very simple material recovery facility complemented with mechanical recycling) typically ranked as the second

best option, apart from the case of GWP where the pyrolysis alternative appeared instead to be more beneficial because of the high losses involved in such simple mechanical pathway. In line with this outcome, Perugini et al. (2005) claimed that the mechanical recycling option assessed was always environmentally preferable to pyrolysis with the only exception of energy consumption. It was also found, though, that chemical recycling (particularly the hydro-cracking process) had a number of valuable environmental indices. Jeswani et al. (2021) demonstrated that chemical recycling of mixed plastic waste via pyrolysis had about 50% lower climate change impact and life cycle energy use relative to energy recovery from plastic waste. However, for most of the remaining impact categories, such as acidification and eutrophication, waste-to-energy (WtE) recovery performed better than pyrolysis due to the higher credits received for the recovered energy. Similar results were obtained by Khoo (2019). A controversial conclusion was given by Demetrious & Crossin (2019) who argued that the treatment of mixed plastic waste was better met environmentally by not following the waste hierarchy, and disposal in landfill was preferable to thermal treatments such as two-stage pyrolysis-gasification.

As for chemical depolymerisation and solvent-based purification, Cosate de Andrade et al. (2016) found that chemical depolymerisation of PLA performed better than incineration with energy recovery for the analysed environmental impacts. However, mechanical recycling seemed preferable to chemical recycling when the recycling products could substitute virgin materials in a 1:1 ratio. Similar conclusions were reached by Maga et al. (2019), who found that mechanical recycling of PLA resulted in lower GWP (277 kg CO₂-eq./tonne waste managed) than both chemical depolymerisation (700 kg CO₂-eq./tonne waste managed) and solvent-based purification (521 kg CO₂-eq./tonne waste managed). In line with this trend, Shen et al. (2010) claimed that mechanical recycling of PET waste had lower impacts than chemical recycling via depolymerisation. However, the authors acknowledged that fibres produced from chemical recycling could be applied more widely (in terms of market applications) than fibres produced from mechanical recycling.

Focusing on the case of food waste as feedstock for chemical recycling technologies, recent studies showed that pyrolysis and hydrothermal carbonisation of food waste lead to environmental savings (a gain of 130 kg CO₂-eq/tonne food waste and 2 kg CO₂-eq/tonne food waste, respectively; Albizzati et al., 2021a) despite not being full-scale. However, the authors also stressed that such savings are lower than what could be achieved by managing food waste with alternative recycling technologies such as anaerobic digestion. For wood waste, Ajao et al. (2021) only reported economic costs of the technology and Papageorgiou et al. (2021) assessed the use of biochar produced from wood waste to remediate contaminated soil, i.e., the studies do not provide a comparison between mechanical, chemical recycling and energy recovery. In general, it should be noticed that, while few LCA studies are available specifically on wood waste, many deal with more clean and homogenous streams of lignocellulosic (agricultural or forestry) biomasses.

Advanced biological recycling

Technologies that employ innovative biological recycling technologies are herein intended as 'advanced' ones. In the context of this project, we focus on studies that investigated MSW, municipal organic waste (food and garden waste), and wood waste as input feedstock⁴. Please, notice that only studies providing detailed input-output inventory (in terms of input of resources, materials, energy and outputs of materials, energy and emissions, either based on primary or secondary technology data) were retained for the analysis. Studies focusing on very specific agro-industrial biomasses or biomass residues (e.g., cassava, spent coffee grounds, citrus waste), lab-based experimental studies that do not provide technology data as well as qualitative reviews were also excluded.

⁽⁴⁾ Key words used for search: Wood waste refinery, Wood waste biorefinery, Organic waste refinery, Organic waste biorefinery, Biowaste refinery, Biowaste biorefinery, Food waste high-value products, Food waste refineries, Food waste biorefineries, Waste refineries.

Outcome of the literature review on LCA of advanced biological recycling

With respect to food waste, to date the most up-to-date and comprehensive study is that of Albizzati et al. (2021a) that, collecting data from all the food waste-focused studies published prior to year 2021, assessed the techno-economic, environmental, and economic performance of advanced biological recycling technologies using food waste as input-feedstock. The study provides a detailed disaggregated input-output inventory of the technologies (i.e., quantitative inputs in terms of fuel, resource, materials and outputs in terms of products and environmental emissions; estimates of costs and labour are also provided). With respect to biological recycling, the technologies covered in the above-mentioned study included processes with TRLs ranging from 3 to 6 (see Table 3). The outputs of these technologies were animal feed⁵, levulinic acid, lactic acid, succinic acid, sophorolipid, dymethylfuran (DMF) and hydroxymethylfurfural (HMF) (all food grade), polyhydroxyalkanoate (PHA) and polylactic acid (PLA) (both plastic grade). A common denominator of all the biological recycling technologies studied in Albizzati et al. (2021a) is that the abovementioned high-value products are produced alongside biogas and digestate, i.e., a fuel carrier and an organic fertilising material. Additionally, the amount of chemicals produced is typically low compared with the amount of biogas and residual digestate (Albizzati et al., 2021a, 2021b), as shown in Table 3. From a sustainability perspective, the results of the study from Albizzati et al. (2021a) highlighted that the technologies producing chemicals (levulinic acid, lactic acid, succinic acid, sophorolipid, HMF, DMF, sophorolipid, PHA, and PLA) still need optimisation to be economic and environmental competitive with those using conventional fossil feedstock. Instead, biological recycling technologies producing protein-feed appeared to be already more sustainable compared with their traditional counterparts (e.g., typically soymeal production). Furthermore, Andreasi Bassi, Boldrin, et al. (2021) investigated the production of PHA from municipal food waste and wastewater sewage quantifying the material flows (Table 3), but also the corresponding environmental and economic impacts. Finally, Tonini et al. (2013, 2014) studied an enzyme-based liquefaction technology capable to take mixed MSW (e.g., household waste) and produce a portfolio of output products, notably metals, plastics, bioliquid (liquefied organic and paper/cardboard waste to be sent to anaerobic digestion), and solid residues (remaining non-recyclable combustible materials, e.g., wood, shoes, non-recyclable plastics)⁶. According to the EU legislation (e.g., Waste Framework Directive as amended in 2018 and Commission Implementing Decision EC 2019/1004), the digestate derived from treatment of not source-separated MSW cannot be used in agriculture, but may be used for land restoration purposes.

Table 3. Overview of literature studies providing techno-economic data on multi-output biorefineries treating food and wood waste as input-feedstock. DMF: dymethylfuran; HMF: hydroxymethylfurfural; MSW: Municipal Solid Waste; PHA: polyhydroxyalkanoate; PLA: polylactic acid. TRL: Technology Readiness Level (estimated). Data are expressed on a wet weight basis, unless differently specified.

Study	Input waste	Technology	Outputs (kg/kg waste input)
Albizzati et al. (2021a)	Food Waste	Black soldier flies - based treatment TRL 6	Protein-feed (0.018 kg), compost, digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 4	HMF (0.04 kg), levulinic acid (0.0079 kg), digestate and biogas

⁽⁵⁾ As targets are intended for re-use and recycling, animal feed derived from food waste is herein considered despite being a technology belonging to the 're-use' portfolio.

⁽⁶⁾ See full-scale waste refinery established in UK to treat household waste; available at <https://orsted.com/en/our-business/bioenergy/renescience>.

Albizzati et al. (2021a)	Food Waste	Fermentation TRL 4	DMF (0.022 kg), levulinic acid (0.009 kg), digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 3-4	Sophorolipid (0.1183 kg), energy-feed (0.214 kg), digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 6	PHA (0.009 kg), digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 3-4	Lactic acid (0.055 kg), digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 3-4	Polylactic acid (0.0275 kg), digestate and biogas
Albizzati et al. (2021a)	Food Waste	Fermentation TRL 3-4	Succinic acid (0.062 kg), digestate and biogas
Tonini et al. (2013, 2014)	Mixed MSW	Enzymatic hydrolysis (liquefaction) TRL 6	Metals 0.07 kg, Plastics 0.17 kg, Bioliquid 0.56 kg (for further anaerobic digestion), and solid fuel 0.2 kg (remaining non-recyclable materials; for energy) ⁷
Andreas Bassi, Boldrin, et al. (2021)	Municipal food waste and wastewater sludge	Fermentation, TRL 6	PHA (0.02-0.024 kg/kg municipal food waste), digestate and biogas

Environmental performance of advanced biological recycling

Overall, the results of the environmental assessments show that advanced biological recycling handling food waste have the potential to bring increased environmental savings relative to alternative waste treatment technologies such as anaerobic digestion or incineration with energy recovery, especially under decarbonised energy systems. This will likely be the case for the EU in next decades (Albizzati et al., 2021a, 2021b; Tonini et al., 2013, 2014). The reason for this lies in the larger environmental savings achieved via material and resource recovery, as opposite to maximising the recovery of electricity and heat in energy-oriented waste installations, as it is the case for incineration and anaerobic digestion. On the other hand, the assessments performed by Albizzati et al. (2021a, 2021b) also stress the low maturity of some of these processes (notably the biological processes producing chemicals such as lactic acid, polylactic acid, succinic acid, sophorolipid, HMF, DMF, and levulinic acid). The authors suggest the need for further research and optimisation to reduce the consumption of energy (mainly industrial steam for distillation) and increase the yield of products from food waste (currently poor), which appear to be the main limitations for making these technologies environmentally and economically competitive with the conventional production processes (see especially the analyses of Albizzati et al., 2021b). Finally, the results obtained in Andreas Bassi, Boldrin, et al. (2021) highlighted that it is more beneficial to produce PHA from municipal food waste (and wastewater sludge) than from polyurethane or first-generation biomass; yet, this conclusion highly depends on the avoided alternative treatment of municipal food waste,

⁽⁷⁾ The mass balance is expressed on a dry matter basis from Tonini et al. (2014). Plastics include soft and hard plastics.

the management of the residues generated at the refinery and the local framework conditions making these results not generally applicable.

4 Proposals for recycling calculation rules

4.1 Policy background

According to the Commission Implementing Decision 2019/1004 (European Commission, 2019a) “where municipal waste materials enter recovery operation whereby those materials are not principally used either as fuel or other means to generate energy, or for material recovery but result in output that includes recycled materials, fuels or backfilling materials in significant proportions, the amount of recycled waste is determined by a mass balance approach which results in taking account only of waste materials that are subject to recycling”. While the mass balance approach is mentioned in the Commission Implementing Decision 2019/1004 for MSW and similarly in Commission Implementing Decision 2019/665 for packaging waste (European Commission, 2019b), how to practically calculate the recycling material that results from multi-output operations such as chemical recycling processes is still not sufficiently clear. This calls for an improvement of the calculation rules to calculate the share of recycling for such processes. In this chapter we propose a framework and a set of rules for calculating the share of recycling for processes that are not well-addressed in the rules provided in Commission Implementing Decisions 2019/1004 and 2019/665, such as chemical recycling processes.

Pragmatic definitions of the calculation points for most of the materials have been provided in both Commission Implementing Decisions 2019/1004 and 2019/665:

- For glass, the calculation point is defined as (i) sorted glass that does not undergo further processing before entering a glass furnace, or (ii) the production of filtration media, abrasive materials, glass fibre insulation and construction materials.
- For paper and cardboard, the calculation point is defined as sorted paper/cardboard that does not undergo further processing before entering a pulping operation.
- For metals, the calculation point is defined as sorted metal that does not undergo further processing before entering a metal smelter or furnace.
- For textiles, the calculation point is defined as sorted textile that does not undergo further processing before its utilisation for the production of textile fibres, rags or granulates.
- For wood, the calculation point is defined as sorted wood that does not undergo further treatment before utilisation in particleboard manufacture.
- For plastic, the calculation point is defined as plastic separated by polymer that does not undergo further processing before entering pelletisation, extrusion, or moulding operations, and plastic flakes that do not undergo further processing before their use in a final product.
- For waste items composed of multiple materials, the calculation point is defined as plastic, glass, metal, wood, textile, paper and cardboard and other individual component materials resulting from the treatment of waste items composed of multiple materials that do not undergo further processing before reaching the calculation point established for the specific material.

Besides, for specific materials, the definition of the calculation point is provided in Commission Implementing Decision 2019/1004:

- For waste electric and electronic equipment (WEEE), the calculation point is defined as the WEEE entering a recycling facility after proper treatment and completion of preliminary activities.
- For batteries, the calculation point is defined as the input fractions entering the battery recycling process.

Thus, the description of calculation point above mentioned for plastic clearly refers to mechanical recycling and does not fit chemical recycling since not all the materials or substances derived from chemical recycling may necessarily be used to synthesize new plastics resulting in a closed loop material recycling. Thus, the concept of calculation point appears not to be appropriate for chemical recycling and similar multi-output technologies. Instead, the mass of material accounted for as being ‘recycled’ for the purpose of achieving

the targets is the result of a mass balance that relies on appropriate measurement points⁸ on inputs and outputs.

Starting from this, it is needed to establish the system boundaries of the mass balance (i.e., defining a mass balance beginning point and a mass balance ending point), and to set in place rules which would allow to determine the amount of material recycled. Note that process losses shall not be accounted as part of recycled material, except for inherent process losses in the recycling process, which would occur regardless of the nature of the input-feedstock (i.e., waste or virgin material). These inherent losses shall not be deducted from the recycling yield similarly to what proposed in European Commission (2019a) and European Commission (2019b) for mechanical recycling.

The selection of the system boundaries of the mass balance depends on the definition of recycling and recycled material, and for certain chemical recycling processes (e.g., pyrolysis) they are open to interpretation due to the complexity of the recycling chain. For example, in a pyrolysis process, it can be claimed by operators that the pyrolysis oil is a recycled product if it achieves EoW status (pyrolysis oil from waste tyres is currently considered a product by REACH (status: intermediate substance, not waste)). Having this in mind, one may argue that the system boundaries of the mass balance could be placed early in the recycling value chain, i.e., after the pyrolysis when the pyrolysis oil is produced. This would clearly lead to a higher yield when compared to a mass balance that includes the whole waste-to-polymer (or -monomer) process, and it would neglect possible losses and fuel production that certainly occur later in the conversion process (at the refinery). Thus, the selection of the boundaries have clear implications in the recycling yield calculation and the reported recycled material.

Having in mind the objectives of the circular economy and the spirit of recycling, aiming at maximising material recovery, the best practice would be to include the whole recycling value chain in the system boundaries and report the recycled quantities at the level of the 'final' transformation to monomers or polymers. In case this is not possible, the traceability of the material have to be acknowledged and one of the options explained later could be implemented (section 4.3). The example depicted in Figure 3 shows the ideal mass balance for a pyrolysis process where the system boundaries include the whole recycling value chain and five different measurement points are needed to quantify the inputs of both waste and virgin feedstock, as well as the different outputs of the process. It is necessary to distinguish between waste and virgin feedstock so as to correctly estimate the recycling yield from waste (see section 4.2).

⁸ Note that herein measurement point does not strictly refer to the definition stated at the Commission Implementing Decision 2019/1004 (European Commission, 2019a) where it refers to "the mass of waste materials measured with a view to determining the amount of waste at the calculation point". Instead, this refers to a general measurement point for different materials (i.e., waste, non-waste, intermediates) necessary to conduct the mass balance.

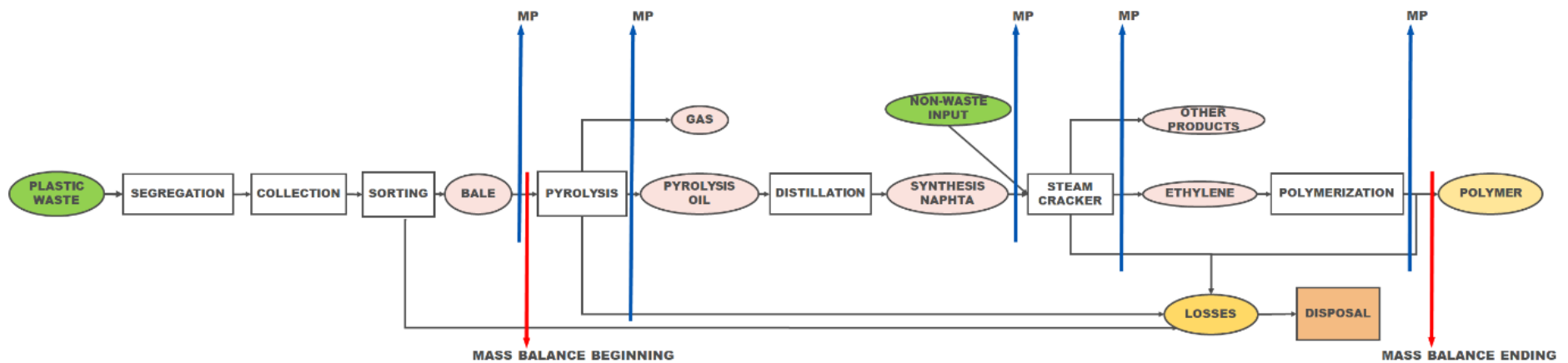


Figure 3: The recycling value chain of plastic waste thermally treated with pyrolysis. The figure shows the mass balance beginning and ideal ending points and the measurement points (MP) at different positions of the value chain. MPs are the points where the mass of materials is measured with a view to determining the amount of recycled material at the mass balance ending point. Notice that the ending point of the mass balance may as well be positioned earlier if the monomers are sold as recycled material (i.e., recycled outputs include also materials other than polymers)

In the following sections, different chain of custody models are described (ISO 22095) and a suggestion for the mathematical implementation of the mass balance model is provided, setting the rules for calculating the share of recycling. While this is generally applicable to all processes, it is especially relevant for complex multi-output chemical recycling processes, such as pyrolysis and gasification.

4.1.1 Chain of custody models

The following sections are largely taken from ISO 22095 (ISO, 2020) and are intended to provide an overview of the different models available. ISO 22095 details terminology and principles of the models that can be used to control inputs and outputs and associated information in a particular chain of custody system. ISO 22095 states that the organizations conforming to ISO 22095 shall establish and implement one or more of the chain of custody models for all materials or products with specified characteristics and shall be transparent about the model chosen. The organization shall only use the same chain of custody model as its supplier or a model with lower physical presence (of the specified characteristic in the output; see Figure 4). The list of (chain of custody) models, ranked from highest to lowest physical presence of the specified characteristics is illustrated in Figure 4. It should be noticed that in the “book and claim model” the administrative flow is not connected to the physical flows throughout the chain of custody. Therefore, within the same chain of custody, it is not possible to switch from the “book and claim model” to other chain of custody models.

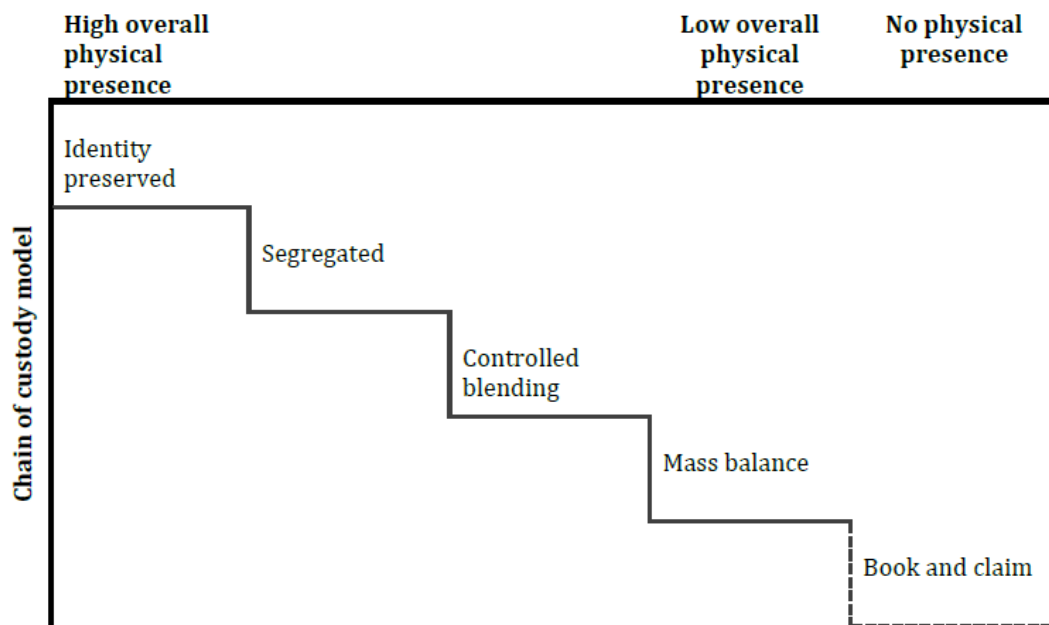


Figure 4. Chain of custody models ranked according to the physical presence of specified characteristics (taken from ISO, 2020).

The different models are defined as follows:

- *Identity preserved model*: Chain of custody model in which the materials or products originate from a single source and their specified characteristics are maintained throughout the supply chain. The material or product can be traced all the way back to the source from which it originates. This model is applicable when there is no mixing of materials in input.
- *Segregated model*: Chain of custody model in which specified characteristics of a material or product are maintained from the initial input to the final output. Addition of material with different characteristics and/or grade to the input is not allowed. Commonly, material from more than one source contributes to a chain of custody

under the segregated model. This model is applicable when there is no mixing of materials in input.

- *Controlled blending model*: Chain of custody model in which materials or products with a set of specified characteristics are mixed according to certain criteria with materials or products without that set of characteristics resulting in a known proportion of the specified characteristics in the final output (also called 'single percentage method').

- *Mass balance model*: Chain of custody model in which materials or products with a set of specified characteristics are mixed according to defined criteria with materials or products without that set of characteristics. The proportion of the input with specified characteristics might only match the initial proportions on average and will typically vary across different outputs. It derives that there is no way to confirm the physical presence of the material with "specified characteristics" (e.g., the recycled material) in the output-product from the process. Two implementation methods are specified: (i) rolling average percentage method; (ii) credit method. The rolling average percentage method is based on the use of a fluctuating proportion of input, bearing specified characteristics, entering the organization over a defined claim period, allowing a claim of an average percentage to be made for the output over the claim period. The organization (i.e., company running the processing facility) shall calculate the average percentage of the inputs and outputs for each material or product. For each material or product, the organization shall define claim periods, which shall reflect the input in relation to the output. These input and output claim periods shall not exceed the specified timeframe. In the credit method the recorded output amount of each type shall be equivalent to the physical input, taking into account a conversion factor. Such conversion factor shall be defined within each material or product at each processing site. The credit account balance shall be calculated for each balancing period (see details in ISO 22095). The balancing period shall not exceed the evaluation period and should be as short as possible. The length of the balancing period shall be evaluated considering the varying needs of different sectors and the desired effectiveness of the system.

- *Book and claim model*: Chain of custody model in which the administrative record flow is not necessarily connected to the physical flow of material or product throughout the supply chain. This chain of custody model is also referred to as "certificate trading model" or "credit trading". The book and claim model aims to ensure that for each purchase for which a claim is made, materials or products with the same specified characteristics have been produced. The book and claim model is most suitable for intangible physical materials or products (e.g., green electricity) and in circumstances where the entire market is controlled.

4.1.2 Application of chain of custody models for recycling technologies

Based on the feedback from researchers and stakeholders, Eunomia Research & Consulting Ltd (2022) summarised the relevant chain of custody models applications for recycling technologies (see Table 4). Both the "identity preserved model" and the "book and claim model" were ruled out because the former is not applicable to the case of waste recycling (multiple sources), while the latter is considered not suitable for (plastic) recycling and not transparent enough. It seems clear that the development of a "mass balance" approach is only really needed for thermal depolymerisation technologies such as pyrolysis and gasification because the remaining physical (mechanical and dissolution) and chemical (depolymerisation) recycling technologies can apply chain of custody models with higher physical presence, therefore with higher credibility and transparency. For example, both

depolymerisation and dissolution can apply a controlled blending model similarly to mechanical recycling of PET bottles. With this in mind, section 4.2 will focus on the definition and implementation of calculation rules specifically for a mass balance model to be applied primarily to the case of thermal depolymerisation technologies. It is important to notice that if the specific characteristics of the system are known (e.g., the stoichiometry), then one should implement one of the methods with higher physical presence; the mass balance approach should be used if these are not known.

Table 4: Application of Chain of Custody models for recycling technologies (from Eunomia Research & Consulting Ltd, 2022).

Model ¹	Physical Recycling		Chemical Recycling	
	Mechanical	Dissolution	Chemical depolymerisation	Thermal depolymerisation ²
Segregation	Partly	Partly	Partly	No
Controlled blending	Yes	Yes	Yes	No
Mass balance	Yes	Yes	Yes	Yes

¹ "Identity preserved" and "book and claim" models were considered not relevant in the context of recycling.

² Thermal depolymerisation (i.e., pyrolysis and gasification).

4.1.3 Unit of measurement

To achieve a correct mass balance, the units of measurement of input and output have to be consistent with each other. For the purpose of calculating and verifying the attainment of the targets set in the Commission Implementing Decision 2019/665 (European Commission, 2019b), it is stated that the weight of recycled packaging waste, as well as the input and output materials, shall be measured applying a natural humidity rate for the packaging waste comparable to the humidity rate of the equivalent packaging put on the market. Whenever the two differ, the amount of packaging waste at the calculation point shall be corrected to reflect the humidity of equivalent packaging placed on the market.

Following this approach, we propose that for any material, product or substance recovered in the recycling process, the quantity at the measurement point within the mass balance is corrected using a natural humidity rate comparable to that of the equivalent virgin material, product or substance placed on the market. This applies to both input (e.g., packaging waste entering the recycling process) and output (e.g., chemicals and materials) of the process.

4.1.4 Input/Output

In the mass balance, the inputs should be distinguished between waste feedstock, virgin feedstock, and co-materials for each process. With respect to the outputs, these should be distinguished between intermediates, energy recovered, and recycled materials obtained.

We define as inputs for each process:

- Waste feedstock (WF): Amount of waste that is used in the process to produce secondary materials. The waste feedstock contributes to the recycling yield.
- Virgin feedstock (VF): Amount of virgin (primary) feedstock that is used in the process (either fossil or bio-based feedstock). The virgin feedstock does not contribute to the recycling yield.
- Co-materials (CoM): Amount of ancillary materials aiding the process and usually recovered at the end of the recycling process (e.g., water, enzymes). Co-materials do not contribute to the recycling yield and are not included in the calculations.

We define as outputs for each process:

- Intermediates: Substances that are manufactured for subsequent processes or sub-processes further down the recycling process chain and consumed in or used for chemical processing to be transformed into other substances.

- Energy recovered: Amount of mass that is converted into either energy products or useful energy needed during the recycling process.
- Output materials: Amount of mass converted into valuable materials, substances, or products.
- Losses: Amount of mass that not recovered as output materials and energy recovered.

It is important to highlight that the amount of recycled outputs must not be higher than the amount of input-waste.

4.2 Calculation rules for the mass balance

4.2.1 Methodological approach

In order to establish a protocol to define and measure the recycling, energy recovery, and loss yields in multiple-output recycling processes (producing a mix of energy, fuels, materials, etc.), a chain of custody method is applied. Among the different approaches of the chain of custody, the mass balance has some characteristics that make it useful for this application. The mass balance encloses the physical mixing of materials and intermediates coming from WF, VF and CoM, and also the chemical reactions of the materials and substances. It should be noted that the system boundaries in the procedure herein presented for calculating the recycling yield refer to the recycling process, which can be composed by one or several sub-processes, that occur after the sorting of the material (Figure 5b). Further, as explained in section 4.1, the system boundaries would ideally account for the whole recycling process of chemical recycling, i.e., from refining/purification processes, up to the production of chemicals and/or materials that do not need further treatment before their subsequent use for production/manufacturing.

For this reason, we herein use the term “recycling yield” (RY) that is associated with the recycling process itself, as opposed to “end-of-life recycling rate” (EoL-RR) that refers to the efficiency of the entire recycling chain (Figure 5a). By using these system boundaries we can consider the RY herein calculated as the fraction of the total waste feedstock that eventually is converted into any of the output materials of the entire recycling process. The EoL-RR can be then obtained by multiplying the RY with the sorting and collection/segregation rate.

Finally, it should be noted that RY differs from recycled content. While they are both a quotient of mass and in some cases they may share the same numerator⁹ (i.e., the amount of recycled material), in the recycling yield the denominator is the total amount of input waste feedstock (to a recycling facility), whereas in the recycled content the denominator is the amount of output material of a given sub-process (intended as the total mass coming from both virgin and waste feedstock, which end up in a final output).

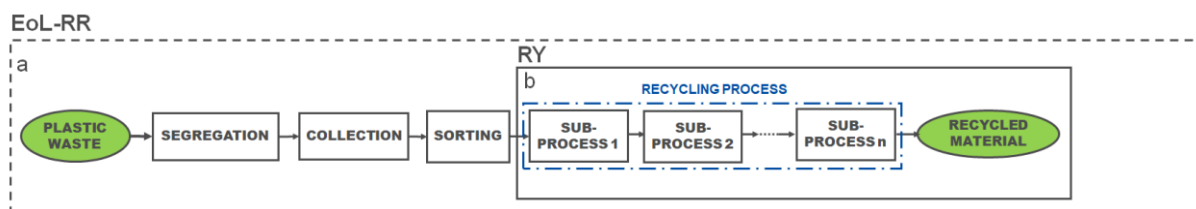


Figure 5: The approach used in the framework for calculating the recycling yield (RY) in terms of system boundaries. The RY here calculated refers to the recycling process up to the final recycled material (b) and not to the entire end-of-life recycling rate (EoL-RR) including segregation, collection, sorting, and final recycling (a). Notice that the recycling process (blue dashed line) can be made up by n sub-processes.

⁹ The numerator will be the same under the following conditions: the recycling process only produces polymers and the same allocation method is used.

The scope of the procedure is based on traceability (not within the process itself but between actors) and chain of custody. The main principle is to give the possibility to track the amount of waste that is recycled along the whole recycling process, from the WF until the final recycled output. It also includes energy recovery, albeit this does not count as recycling. Indeed, in the method herein proposed energy and losses are accounted for; this could be further expanded to investigate if, for example, mass recovered as energy is either used as fuel or energy or internally in the process. However, this is out of scope of the current study. Yet, to close the mass balance, we have to account for energy and losses.

The starting point of the mathematical framework presented herein is the standard ISO22095 (ISO, 2020). According to ISO 22095 the amount of either WF or VF would be considered identical in terms of mass and, therefore, identical within the calculation rules of the mass balance. However, for our purposes, the estimation of the RY should only be based on the amount of WF and thus excluding the VF, even if it is part of the input. Therefore, an adjustment of the method proposed by the ISO 22095 is herein presented in order to take into account the proportion between WF and VF as input to the recycling process and the corresponding calculation of recycling, energy recovery and loss yields.

4.2.2 Mathematical framework

By properly defining all inputs and outputs, the mathematical framework herein proposed calculates:

- The recycling yield (RY) of the recycling process defined within the system boundaries.
- The energy recovery yield (ERY) of the recycling process defined within the system boundaries.
- The loss yield (LY) of the recycling process defined within the system boundaries.

Figure 6 illustrates a generic recycling process composed of two sub-processes resulting in intermediates, output material, energy recovery and losses.

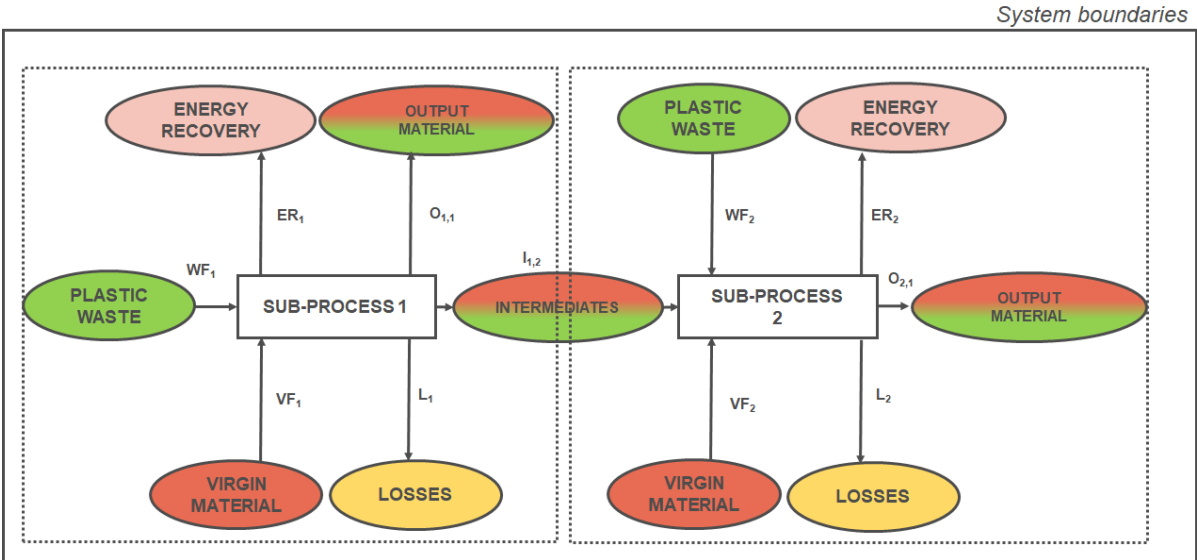


Figure 6: Illustration of a generic recycling process composed of two sub-processes. After undergoing sorting and pre-treatment, plastic waste (input ; WF₁ and WF₂) enters into sub-process 1 and sub-process 2. Sub-process 1 produces four outputs: output material 1 (O_{1,1}), mass recovered as energy (ER₁), losses (L₁) and intermediates (I_{1,2}). The intermediates flow into sub-process 2 contributing to the sub-process as input together with an additional input of plastic waste (WF₂). Sub-process 2 produces three outputs: output material 2 (O_{2,1}), mass recovered as energy (ER₂) and losses (L₂).

Notice that in Figure 6 the following inputs/outputs are defined as follows:

$O_{p,m}$: Output material m (with m=1...k) in sub-proces p (with p=1...n) [t]

ER_p : Mass transformed into energy from sub-process p (with $p=1\dots n$) [t]

L_p : Material loss from sub-process p (with $p=1\dots n$) [t]

WF_p : Waste input to sub-process p (with $p=1\dots n$) [t]

VF_p : Virgin input to sub-process p (with $p=1\dots n$) [t]

$I_{p,s}$: Intermediate material flowing from sub-process p (with $p=1\dots n$) to sub-process s (with $s=1\dots n$, under the condition that $p \neq s$) [t]

The CoM are not considered in the calculation as they do not affect the RY.

The approach herein proposed provides a mathematical way to calculate the RY (expressed as %) of the total waste feedstock (WF) fed to a recycling process. Indeed, recycling only refers to mass coming from waste inputs. The framework also allows for calculating the yield of mass recovered as energy and the yield of mass lost in the recycling process. All yields are referred to mass coming from waste inputs only.

As the yields are all expressed with respect to waste feedstock only, it is necessary to quantify how much of material output, mass recovered as energy and losses are obtained from it. This is done by calculating three different allocation factors, using mass as allocation key, for material outputs, for mass recovered as energy, and for mass lost. The allocation factors are calculated for each sub-process that contribute to the recycling process chain and denote the fraction of each input converted into output material, energy or material loss. It is expressed as a percentage (%) and, in a mass balance method, it is independent from the type of input (waste or virgin). The allocation usually starts with the sub-process that exhibits no production of intermediates and, therefore, one needs to proceed backwards in the calculation up to the first sub-process of the recycling process chain.

Calculation of recycled output material and recycling yield

The framework herein presented allows for calculating the RY of a recycling system defined by appropriate boundaries. As it was already mentioned, the RY is defined as the efficiency of converting WF into output material. To be able to calculate the RY, the amount of output material generated from waste only needs to be defined. This is done by means of a material allocation factor (MAF_p) that is defined for each sub-process p of the recycling process chain and is multiplied by the corresponding input waste feedstock. As it was already mentioned, one needs to start calculating the mass allocation factors from the sub-process that does not produce intermediates and then proceed backwards in the calculation up to the first sub-process of the recycling process chain. By proceeding backwards the allocation factor takes into account the partitioning of mass in the sub-processes where intermediates are used. By allocating the masses of the recycled output products to the waste inputs, it is then possible to calculate the RY of the recycling process.

Hereafter, the MAF_p and RY are calculated starting from the most simple recycling process chain (i.e., one that is composed by only one sub-process) and gradually we generalize the corresponding equations for more complex non-linear processes. Figure 7 displays the simplest recycling process, i.e., one that is only composed by one sub-process.

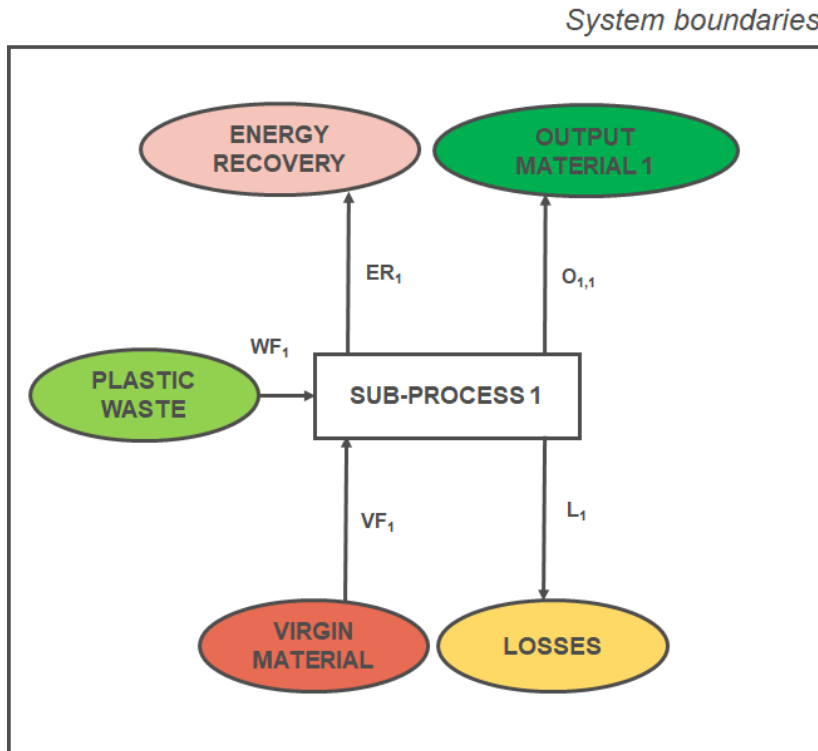


Figure 7: Illustration of a recycling process composed by only one sub-process (sub-process 1) transforming waste (WF_1) and virgin feedstock (VF_1) into an output material ($O_{1,1}$), mass recovered as energy (ER_1) and material losses (L_1).

In this particular case illustrated in Figure 7, the mass allocation factor (MAF_1) corresponds to the RY, which can be calculated as the ratio between the total mass of output material ($O_{1,1}$) and the total output of the process (including mass recovered as energy and losses) as reported in Equation 1.

$$RY = MAF_1 = \frac{O_{1,1}}{ER_1 + L_1 + O_{1,1}}$$

Equation 1

Note that yields are usually defined as the ratio between outputs and inputs. However, since the systems considered herein are assumed not to have any mass accumulation, the sum of the total outputs equals the sum of the total inputs (see Equation 2; VF is the virgin feedstock in input). Therefore, the denominator can more simply be defined as the sum of the outputs of the recycling process.

$$WF_1 + VF_1 = ER_1 + L_1 + O_{1,1}$$

Equation 2

The amount of waste feedstock (input) that is recycled into the output material (RM_p), is given by the waste feedstock entering the sub-process (WF_p) multiplied by the allocation factor of the sub-process (MAF_p), as shown in Equation 3.

$$RM_p = MAF_p \cdot WF_p$$

Equation 3

In the example provided in Figure 7, Equation 3 would be written as shown in Equation 4.

$$RM_1 = MAF_1 \cdot WF_1$$

Equation 4

Note that recycled material (RM_p) only refers to the fraction of the output material that originates from waste feedstock. When a generic process requires virgin feedstock, part of the output material will originate from this non-waste feedstock. In the mass balance approach, the fraction of recycled material originated from non-waste feedstock is not considered as recycled material. In general, the amount of recycled material will be lower than the output material. Notice that when a process requires only waste feedstock the amount of output material is equal to the amount of recycled material.

Let us now focus on a more complex recycling process chain composed by two sub-processes, as displayed in Figure 6. In this case the waste inputs of each sub-process need to be allocated to the output materials. This is done by performing a mass allocation and the procedure entails starting from the sub-process that does not produce any intermediate and proceeding backwards. Sub-process 2 is the last process of the recycling process chain and it does not produce any intermediate that would contribute as input in subsequent sub-processes. Therefore, for sub-process 2, the input waste is entirely allocated to the output material obtained. The allocation factor of sub-process 2 (MAF_2) is calculated as shown in Equation 5.

$$MAF_2 = \frac{O_{2,1}}{ER_2 + L_2 + O_{2,1}}$$

Equation 5

When calculating the allocation factor of sub-process 1 (MAF_1), we have to consider the influence sub-process 2 has on the intermediate resulting from sub-process 1, as shown in Equation 6.

$$MAF_1 = \frac{O_{1,1} + MAF_2 \cdot I_{1,2}}{ER_1 + L_1 + O_{1,1} + I_{1,2}}$$

Equation 6

Where $I_{1,2}$ is the intermediate produced in sub-process 1 that is used as input into sub-process 2. Focusing on the numerator of Equation 6, the first term coincides with the amount of input waste feedstock resulting as output material of sub-process 1. The second term represents the share of input waste feedstock of sub-process 1 flowing to sub-process 2 as an intermediate and contributing to the production of output material. Notice that the allocation factor relative to sub-process 2 (MAF_2) would take into account the influence of sub-process 3/4/.../n if there were more sub-processes.

Hence, Equation 5 and Equation 6 can be generalized as follows for calculating the allocation factor of a sub-process p.

$$MAF_p = \frac{\sum_{m=1}^k O_{p,m} + \sum_{s=1}^n (I_{p,s} \cdot MAF_s)}{ER_p + L_p + \sum_{m=1}^k O_{p,m} + \sum_{s=1}^n I_{p,s}} \quad \forall p; s. t. \quad p \neq s$$

Equation 7

$O_{p,m}$: Output material m (with $m=1...k$) in sub-process p (with $p=1...n$) [t]

$I_{p,s}$: Intermediate material flowing from sub-process p (with $p=1...n$) to sub-process s (with $s=1...n$, under the condition that $p \neq s$) [t]

MAF_s : Mass allocation factor of sub-process s (with $s=1...n$, under the condition that $p \neq s$) [t]

ER_p : Mass converted into energy from sub-process p (with $p=1...n$) [t]

L_p : Material loss from sub-process p (with $p=1...n$) [t]

In Equation 7, the first term of the numerator coincides with the amount of input waste feedstock that is converted directly into an output material, while the second term represents the fraction of each intermediate that is converted into an output material in subsequent sub-processes adjusted by the corresponding allocation factors. Notice that

intermediates cannot be negative and only material flows towards a subsequent sub-process are considered (i.e., loops are not allowed).

Thus, to calculate the amount of waste feedstock that eventually flows into the output materials, the allocation factors obtained by means of Equation 5 and Equation 6 need to be multiplied by the corresponding amount of waste entering as input to the specific sub-process (Equation 8 and Equation 9).

$$RM_2 = MAF_2 \cdot WF_2$$

Equation 8

$$RM_1 = MAF_1 \cdot WF_1 = \frac{O_{1,1} + MAF_2 \cdot I_{1,2}}{ER_1 + L_1 + O_{1,1} + I_{1,2}} \cdot WF_1$$

Equation 9

The total mass of recycled waste (RM) in the entire recycling process is calculated as the sum of the masses obtained through Equation 8 and Equation 9, as shown in Equation 10.

$$RM = RM_1 + RM_2$$

Equation 10

Further, Equation 10 can be generalized in Equation 11 as follows:

$$RM = \sum_{p=1}^n RM_p$$

Equation 11

Then, it is possible to calculate the recycling yield of the recycling process by dividing the total mass of recycled waste by the total input waste (see Equation 12).

$$RY = \frac{RM_1 + RM_2}{WF_1 + WF_2}$$

Equation 12

Finally, Equation 12 can be generalised in Equation 13 as follows:

$$RY = \frac{RM}{\sum_{p=1}^n WF_p} = \frac{\sum_{p=1}^n RM_p}{\sum_{p=1}^n WF_p}$$

Equation 13

Notice that within the current framework, we propose a mass-based allocation of inputs (that is proportional between WF and VF), which we believe is more appropriate than energy- or price-based or other allocation options in the specific context of a mass balance. The mass-based allocation should be applied across all recycled materials produced in the recycling process, regardless of the type (e.g., polymers or non-polymers) as long as they can be considered as recycled products¹⁰. It should be noted that CoM are not considered here as they do not affect the recycling yield as well as the amount of recycled material. All products used for energy (internal consumption or as fuels) are not considered as recycled material following the definition of recycling.

It is important to highlight that the applied mass-based allocation framework herein proposed determines the total quantity of any waste-based output, but it does not allocate the input waste feedstock to a specific output material obtained from the recycling process, as this is closely related to determining the recycled content of an output stream (that is

¹⁰ Other options for allocation exist, e.g. allocating only across polymer-products (thus excluding all materials that are not used to produce new plastics), including fuel-products in the allocable products, or using a so-called free allocation option (i.e. up to the company reporting how to allocate the input-waste across the output-products).

beyond the scope of this report). The calculation method to determine the recycled content of the output materials based on a mass balance approach can be developed by using the same mathematical reasoning as presented in this chapter, but, again, this is not the scope of this report as mentioned before. In case an operator wants to allocate the total quantity of waste-based output (obtained from Equation 11) among output streams (when multiple output exists), it should be done according to the methods proposed in literature (Eunomia Research & Consulting Ltd, 2022). As an example of those methods, Figure 8 illustrates the case of an allocation in a one-step recycling process with three different outputs. Thus, the recycled quantity obtained can be (i) allocated to the different outputs by using a proportional allocation method or (ii) be freely allocated (i.e., non-proportional) to one or more output streams of the total process as long as the total allocation does not exceed the amount of recycled material.



Figure 8: Illustration of the proportional and non-proportional allocation method in a one-step recycling process. In the proportional method, the allocation is applied to each output to guarantee an actual physical and chemical relationship of the recycled material or product with the waste feedstock. In the non-proportional, the allocation is applied entirely to one single output. The choice of what kind of allocation is used, is out of the scope of this report and is up to the operator. What matters for the calculation of the recycling yield is the total amount of recycled material (sum of individual recycled materials).

Calculation of mass recovered as energy and energy recovery yield

The calculation of allocation factors and ERYs follows the same reasoning as explained for the output material and RY. If mass is converted into energy from a sub-process p , Equation 14 can be used to calculate the corresponding allocation factor ($ERAF_p$), which is defined similarly to the allocation factor calculated for the output material.

$$ERAF_p = \frac{ER_p + \sum_{s=1}^n (I_{p,s} \cdot ERAF_s)}{ER_p + L_p + \sum_{m=1}^k O_{p,m} + \sum_{s=1}^n I_{p,s}} \quad \forall p; s, t, p \neq s$$

Equation 14

Where:

$ERAF_p$ = Energy recovery allocation factor for the mass converted into energy in sub-process p [%]

$ERAF_s$ = Energy recovery allocation factor for the mass converted into energy in sub-process s [%]

As for the allocation factors calculated for output materials, the allocation factor for mass converted into energy of sub-process p is affected by the allocation factors of other sub-processes if the former results in the production of intermediates. In Equation 14, the first term of the numerator coincides with the mass of input waste feedstock converted into energy, while the second term represents the fraction of input waste feedstock contributing to the production of energy in other sub-processes as the former results in the production of intermediates. On the other hand, the denominator is simply the sum over the total outputs of sub-process p .

The amount of input waste feedstock converted into energy in sub-process p can be calculated by implementing Equation 15.

$$ER_p = ERAF_p \cdot WF_p$$

Equation 15

The total quantity of input waste feedstock recovered as energy throughout the recycling process is calculated by applying Equation 16.

$$ER = \sum_{p=1}^n ER_p$$

Equation 16

Finally, the ERY can be calculated as the ratio between the total quantity of input waste feedstock recovered as energy (ER) and the total input waste feedstock to the recycling process (see Equation 17).

$$ERY = \frac{ER}{\sum_{p=1}^n WF_p} = \frac{\sum_{p=1}^n ER_p}{\sum_{p=1}^n WF_p}$$

Equation 17

The ERY accounts for the part of the waste feedstock that is used to produce energy or fuels, including self-consumption. Notice that a distinction between self-consumption and other fuel products can easily be made, but it is not an objective of our exercise. Finally, it is important to note that the presence of CoM, usually recovered at the end of the recycling process, does not affect the amount of energy recovered and, consequently, the ERY.

Calculation of mass loss and loss yield

The calculation of the losses, as mass, and LY is derived analogously to that of recycled output mass and energy recovery.

First, the allocation factor of mass lost for a sub-process p (LAF_p) is defined in Equation 18. Notice that the same reasoning as for the output mass and mass recovered as energy applies.

$$LAF_p = \frac{L_p + \sum_{s=1}^n (I_{p,s} \cdot LAF_s)}{ER_p + L_p + \sum_{m=1}^k O_{p,m} + \sum_{s=1}^n I_{p,s}} \quad \forall p; s. t. p \neq s$$

Equation 18

Where:

LAF_p = Loss allocation factor of the losses generated in sub-process p [%]

LAF_s = Loss allocation factor of the losses generated in sub-process s [%]

The quantity of input waste feedstock lost in sub-process p is calculated as described in Equation 19.

$$L_p = LAF_p \cdot WF_p$$

Equation 19

While the total input waste feedstock lost over the recycling process is obtained by summing all losses occurring at each sub-process p (Equation 20).

$$L = \sum_{p=1}^n L_p$$

Equation 20

The LY is calculated as the ratio between the total input waste feedstock lost over the recycling process and the total input waste feedstock needed in the recycling process (Equation 21).

$$LY = \frac{L}{\sum_{p=1}^n WF_p} = \frac{\sum_{p=1}^n L_p}{\sum_{p=1}^n WF_p}$$

Equation 21

It is important to note that the condition reported in Equation 22 needs to be satisfied to ensure a correct mass balance, namely the total input waste feedstock needs to equal the total amount of recycled output mass, mass recovered as energy and input waste feedstock lost over the recycling process.

$$\sum_{p=1}^n WF_p = \sum_{p=1}^n RM_p + \sum_{p=1}^n ER_p + \sum_{p=1}^n L_p$$

Equation 22

Finally, Equation 22 corresponds to the sum of the recycling yield, energy recovery yield, and loss yield that needs to be equal to 100% (Equation 23).

$$RY + ERY + LY = 100\%$$

Equation 23

4.2.3 Examples

In this section we apply the derived equations on some specific cases. In Table 5, four different cases are shown to illustrate the effect of modifying the input and output of a recycling process composed by one sub-process on the RY, ERY and LY. In Box A we provide an example of a more complex recycling process.

Table 5: Four different cases of single-step recycling processes and their corresponding RY, ERY and LY. Since the recycling process contains only one sub-process, the allocation factor coincides with the recycling yield. Values are given as tonne or percentage and rounded.

		Case 1	Case 2	Case 3	Case 4
Inputs	Waste feedstock	6000	4000	4000	4000
	Virgin feedstock	0	2000	2000	2000
	Co-materials	2000	2000	2000	2000
Outputs	Output material (WF+VF)	3000	3000	1000	1000
	Co-materials	2000	2000	2000	2000
	Mass recovered as energy (WF+VF)	2000	2000	4000	3000
	Material loss (WF+VF)	1000	1000	1000	2000
Yields	RY	50%	50%	17%	17%
	ERY	33%	33%	67%	50%
	LY	17%	17%	17%	33%

Total input waste feedstock converted into output material, energy, losses	Recycled output material	3000	2000	667	667
	Mass recovered as energy	2000	1333	2667	2000
	Losses	1000	667	667	1333

The inputs and outputs reported in Table 5 are the actual physical material flows that can be measured at the measurement points (Figure 3). In general, the RY, ERY and LY can be calculated using Equation 13, Equation 17, Equation 21, respectively, but since the recycling processes in Table 5 are described by only one sub-process, the RY, ERY and LY coincide with the corresponding mass, energy, and loss allocation factors, calculated using Equation 7, Equation 14, Equation 18, respectively. Thus, considering case 2 of Table 5, RY, ERY, and LY equal 50%, 33%, and 17%, respectively.

$$RY = MAF_1 = \frac{O_{1,1}}{O_{1,1} + ER_1 + L_1} = \frac{3000}{3000 + 2000 + 1000} = 50\%$$

$$ERY = ERAF_1 = \frac{ER_1}{O_{1,1} + ER_1 + L_1} = \frac{2000}{3000 + 2000 + 1000} = 33\%$$

$$LY = LAF_1 = \frac{L_1}{O_{1,1} + ER_1 + L_1} = \frac{1000}{3000 + 2000 + 1000} = 17\%$$

The total waste that is converted into recycled material, energy recovery and loss, is calculated using Equation 3, Equation 15 and Equation 19, by simply multiplying the input waste feedstock with the corresponding allocation factor.

$$RM = RM_1 = MAF_1 \cdot WF_1 = 50\% \cdot 4000 = 2000$$

$$ER = ER_1 = ERAF_1 \cdot WF_1 = 33\% \cdot 4000 = 1333$$

$$L = L_1 = LAF_1 \cdot WF_1 = 17\% \cdot 4000 = 667$$

Note that the equations for calculating the yields only depend on the output of the process, and not on the inputs. Therefore, in case the total input quantity is maintained but the proportion among WF, VF and CoM inputs changes (Table 5), the yields are not affected. This can be clearly observed by comparing case 1 with case 2. Although the distribution of the inputs among WF, VF, and CoM differs, the yields remain unchanged. Yet, when calculating the recycled output material, energy recovered and losses, the amount of input waste feedstock affects the results (see Table 5). The presence of CoM in the recycling process does not affect the calculation of the RY, ERY, and LY, as it is shown by the example reported in Table 5. Indeed, CoM does not appear in any of the equations.

The mass balance approach presented constitutes an aligned framework for calculating yields in systems with multiple input/output such as chemical recycling operations. Conforming to the calculation rules proposed, a calculation framework developed in Microsoft Excel has been made available to support researchers and stakeholders (Annex 4).

Box A: Recycling yields calculation – complex recycling process example

For the sake of clarity, we hereby illustrate an example of a recycling process that includes three sub-processes in which two output materials are produced and energy is recovered from both waste and virgin feedstocks (see Figure A1). The process is non-linear in the sense that some sub-processes produce multiple intermediates that are used by the subsequent sub-process, but also by other sub-processes further down the recycling process chain. Specifically:

- Sub-processes 1 produces two intermediates used as input for sub-process 2 and sub-process 3;

- Sub-process 2 receives an intermediate (as input) from sub-process 1 and produces an intermediate used as input for the subsequent sub-process 3;
- Sub-process 3 receives intermediates as inputs from both sub-process 1 and sub-process 2.

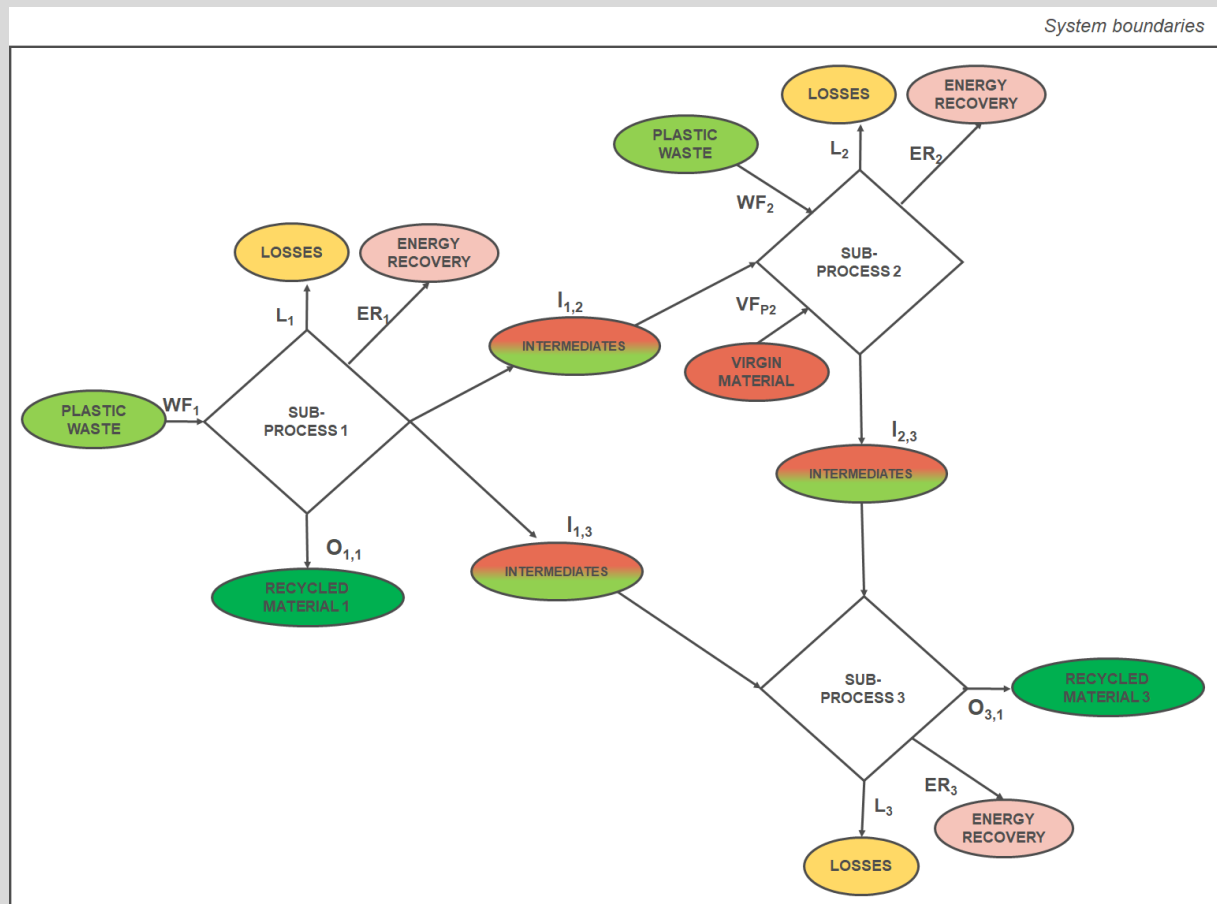


Figure A1: Illustration of non-linear recycling process involving 3 sub-processes. In sub-process 1 and 3 two output materials are produced (green circles), in all three sub-process energy is recovered (pink circles) and losses occur in all three sub-processes (yellow circles). Sub-process 1 generates intermediates that flow into sub-process 2 and sub-process 3, while sub-process 2 generates intermediates for sub-process 3 (red/green circles). The recycling process is not linear since the three sub-processes do not form a straight consecutive line in the mass flow.

The recycling process under study starts when 10000 t of waste feedstock enters sub-process 1 to produce 1000 t of output material 1, 2000 t of energy, and 4000 t and 2000 t of intermediates that flow into sub-process 2 and sub-process 3, respectively. Sub-process 2 receives 4000 t of intermediates from sub-process 1 together with 2000t of waste feedstock and 12000 t of virgin feedstock, and converts 14000 t into energy and produces 3000 t of intermediates that flow into sub-process 3. The inputs of sub-process 3 (namely, 2000 t of intermediates from sub-process 1 and 3000 t of sub-process 2) are converted into 2500 t of output material. All three sub-processes have material losses, namely 1000 t, 1000 t and 2500 t in sub-process 1, 2 and 3, respectively.

Table A1 reports all inputs and outputs, the allocation factors for the output mass, mass recovered as energy and losses of each sub-process, as well as the mass of waste feedstock that is converted into recycled material, energy recovery and losses from the waste feedstocks entering different sub-processes.

Table A1: Inputs and outputs of a non-linear recycling process entailing three sub-processes. The calculated allocation factors for the output materials (MAF), mass recovered as energy (ERAF), and losses (LAF), together with the input waste feedstock converted into recycled output material (RM), energy (ER) and loss (L) are reported.

		Sub-process 1	Sub-process 2	Sub-process 3
Input	WF	10000	2000	0
	VF	0	12000	0
	Intermediate from sub-process 1	0	4000	2000
	Intermediate from sub-process 2	0	0	3000
Output	Output material	1000	0	2500
	Intermediates to sub-process 2	4000	0	0
	Intermediates to sub-process 3	2000	3000	0
	Energy recovered	2000	14000	0
	Material loss	1000	1000	2500
Allocation Factors	MAF	0.23	0.08	0.5
	ERAF	0.51	0.78	0
	LAF	0.26	0.14	0.5
Waste conversion	RM	2333	167	0
	ER	5111	1556	0
	L	2556	278	0

First, all allocation factors (for the output mass, mass recovered as energy, and losses) need to be quantified. As sub-process 3 is the last sub-process of the recycling process chain (i.e., it does not produce any intermediate), Equation 7, Equation 14, and Equation 18 are applied starting from this sub-process and proceeding backwards.

$$MAF_3 = \frac{2500}{2500 + 2500} = 0.5$$

$$ERAF_3 = \frac{0}{2500 + 2500} = 0$$

$$LAF_3 = \frac{2500}{2500 + 2500} = 0.5$$

Having calculated the allocation factors for sub-process 3, it is now possible to quantify the allocation factors of sub-process 2 that produces intermediates flowing into sub-process 3 that, therefore, influences the allocation factors of sub-process 2.

$$MAF_2 = \frac{0 + 3000 \cdot 0.5}{3000 + 14000 + 1000} = 0.08$$

$$ERAF_2 = \frac{14000 + 3000 \cdot 0}{3000 + 14000 + 1000} = 0.78$$

$$LAF_2 = \frac{1000 + 3000 \cdot 0.5}{3000 + 14000 + 1000} = 0.14$$

Finally, the allocation factors of sub-process 1 can be quantified taking into account the allocation factors calculated for sub-process 2 and 3, as these two sub-processes receive intermediates from sub-process 1.

$$MAF_1 = \frac{1000 + 4000 \cdot 0.8 + 2000 \cdot 0.5}{1000 + 4000 + 2000 + 2000 + 1000} = 0.23$$

$$ERAF_1 = \frac{2000 + 4000 \cdot 0.78 + 2000 \cdot 0}{1000 + 4000 + 2000 + 2000 + 1000} = 0.51$$

$$LAF_1 = \frac{1000 + 4000 \cdot 0.14 + 2000 \cdot 0.5}{1000 + 4000 + 2000 + 2000 + 1000} = 0.26$$

Sub-process 1 and 2 receive a direct input of waste feedstock. Having calculated the allocation factors of each of these sub-processes, it is possible to quantify how much of each input waste feedstock is converted into recycled output mass, mass recovered as energy and how much is lost. Note that since sub-process 3 does not receive direct input of waste feedstock, RM_3 , ER_3 , and L_3 equal 0.

$$RM_1 = 0.23 \cdot 10000 = 2333 \text{ tonne} ; RM_2 = 0.08 \cdot 2000 = 167 \text{ tonne}$$

$$ER_1 = 0.51 \cdot 10000 = 5111 \text{ tonne} ; ER_2 = 0.78 \cdot 2000 = 1556 \text{ tonne}$$

$$L_1 = 0.26 \cdot 10000 = 2556 \text{ tonne} ; L_2 = 0.14 \cdot 2000 = 278 \text{ tonne}$$

The total recycled mass corresponds to 2500 tonnes, while the total mass recovered as energy 6667 tonnes and the total mass lost 2584 tonnes (Table A2).

The recycling yield, energy recovery yield and loss yield of the recycling process can be quantified by implementing Equation 13, Equation 17, and Equation 21

$$RY = \frac{\sum_{p=1}^3 RM_p}{\sum_{p=1}^3 WF_p} = \frac{2333 + 167 + 0}{10000 + 2000 + 0} = 0.210 = 21\%$$

$$ERY = \frac{\sum_{p=1}^3 ER_p}{\sum_{p=1}^3 WF_p} = \frac{5111 + 1556 + 0}{10000 + 2000 + 0} = 0.560 = 56\%$$

$$LY = \frac{\sum_{p=1}^3 L_p}{\sum_{p=1}^3 WF_p} = \frac{2556 + 278 + 0}{10000 + 2000 + 0} = 0.240 = 24\%$$

Table A2: Overview of results obtained for the recycling process described. The following acronyms are used: ER – mass recovered as energy, ERY – energy recovery yield, L – mass lost, LY – loss yield, RM – recycled output mass, RY recycling yield.

Recycling		Energy recovery		Loss		Mass balance	
RM (t)	RY (%)	ER (t)	ERY (%)	L (t)	LY (%)	Total (t)	Total (%)
2500	21%	6667	56%	2833	24%	12000	100%

4.3 Traceability

A system is needed to trace information on the material that is transferred from one actor to the other in the recycling chain. This could be the case when pyrolysis oil produced from one facility (belonging to one organisation) in one site is transferred to a second plant (belonging to a different organisation), which uses the pyrolysis oil as input to e.g., the cracking unit. The tracking would be needed up to the production of building blocks from the cracker (i.e., at the level of refiners) such as ethylene, propylene. Such traceability allows subtracting the fraction of pyrolysis oil that is directly sold as fuel (after the pyrolysis unit) or subtracting the portion of pyrolysis oil mass that is sent to cracking units but is transformed into fuels or used for internal energy consumption, or even 'lost' (not converted into valuable outputs) in the cracking owing to the process inefficiencies. A similar logic applies to the case of syngas produced in a gasification unit, which is then transferred to a chemical refinery for upgrading to building blocks to be used in the chemical industry. Having the above in mind, it is suggested that a traceability (auditing and compliance) system similar to that of renewable energy (under the Renewable Energy Directive (EU) 2018/2001) or of recycled content certification is established also for recycling declaration regarding recycling.

Economic operators shall provide third party verification and traceability and be subject to annual auditing. Certification schemes following standard EN 15343 (Plastics recycling traceability and assessment of conformity and recycled content) are presently designed for mechanical recycling and may be used to support verification and certification of recycled

plastic as well. Certification schemes that cooperate with independent third party certification bodies that conduct the audits already exist and with their auditing and certification approaches this type of traceability system can be established (e.g., International Sustainability and Carbon Certification, ISCC). All value chain actors that take legal ownership of the material must be certified by a third party and each site audited annually. This includes recyclers (including pre-treatments) until the point in which the product is placed on the market. While the point of origin of the plastic waste does not necessarily require certification or auditing, an annual self-declaration could be provided from each point of origin to the collector/recycler so to certify that the material is a waste and not a byproduct, for example. The point of origin must hold appropriate licenses and permits to act as a legal waste management company or as an entity that generates recovered material as defined in ISO 14021:2016. In addition, based on the self-declarations, additional verifications (on plausibility of volumes, types of wastes, etc.) can be done in cases of doubts and audit at the point of origin could still take place on sample basis.

However, traceability systems for carbon, sustainable energy or recycled content are typically thought to track a flow of information downstream, up to the final user of the secondary material that could be a cracker or a converter or a brand owner (Figure 9). This means that the information on the mass balance and yields would be available to the cracker/refiner using the pyrolysis or syngas, rather than to the pyrolizer or gasifier that is positioned earlier in the value chain, in case these are two distinct actors. This creates a problem about who should report recycling (amounts) because if pyrolysis oil and syngas were considered intermediate substances (i.e., not having a waste status), the cracker/refiner may not be subject to the legal obligation of reporting recycled amounts from management of waste. At the same time the pyrolizer or gasifier, which is managing the waste into intermediates such as oil or syngas, would not be able to report recycling yields corrected via a mass balance approach if it is not able to receive information on the downstream operations (conversion yields, use of the cracker products as fuel or material, etc.) because of confidentiality reasons.

Therefore, we envisage the following possible options for traceability and reporting:

- The recycled quantities are reported at the level of the final transformation, using the mass balance approach and appropriate traceability schemes.
- The recycled quantities are reported by the first waste management operator, e.g., pyrolizer or gasifier, adjusted via a mass balance approach with information on conversion and yields provided by the downstream operators via traceability schemes.
- The recycled quantities are reported by the first waste management operator, e.g., pyrolizer or gasifier, adjusted via a mass balance approach with conversion factors/yields provided by the Commission.

The last option would not need traceability schemes in place and would be based on default conversion and yield factors based on literature data. In particular, to derive these factors, the Commission could make use of existing internal studies¹¹ or launch a simple literature review study with the aim of aggregating all the data collected by existing Commission and non-Commission studies on chemical recycling technologies such as pyrolysis and gasification.

¹¹ For example the study commissioned by DG GROW and performed by JRC (Garcia-Gutierrez et al., 2023; in press).

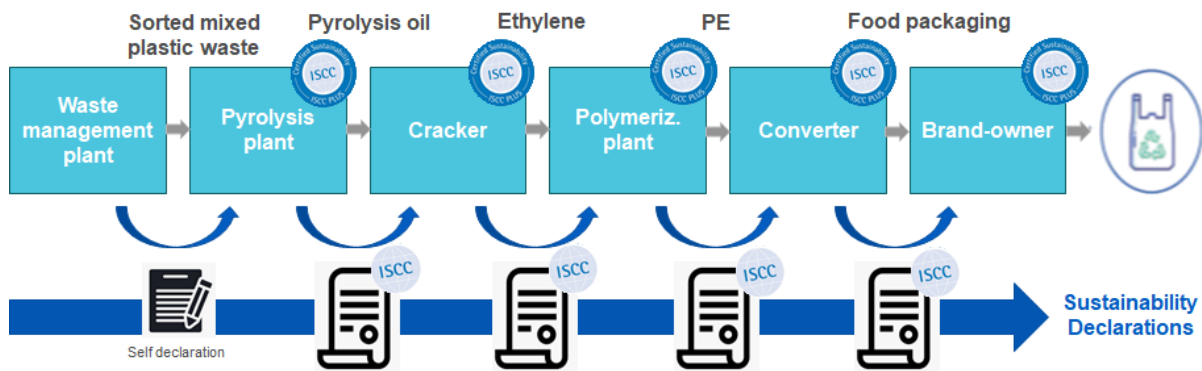


Figure 9: Flow of information in the value chain and traceability of waste and recycling. Elaborated after ISCC System GmbH 2022 website (ISCC PLUS Certification for the Circular Economy and Bioeconomy – Mass Balance Approach and Verification).

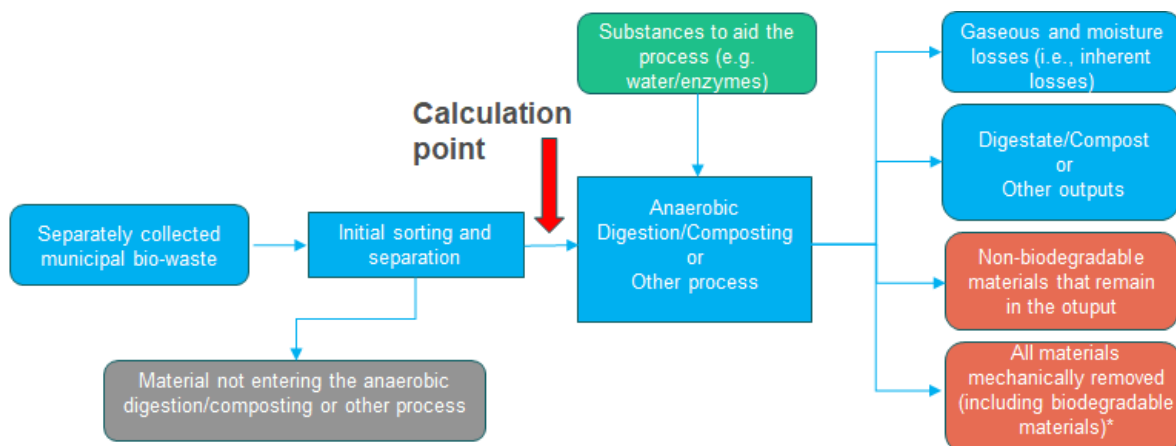
4.4 Calculation rules for biodegradable waste and compostable plastic waste

Biodegradable waste (i.e., bio-waste) differs from the other waste types since it is mostly composed of water (up to 90%) and is not inert. For this reason, the existing calculation rules for this waste stream, as well as for compostable plastic waste when collected along with it, differ from the other streams of MSW as detailed in the “Guidance for the compilation and reporting of data on municipal waste” (Eurostat, 2021) and are presented herein.

4.4.1 Existing calculation rules for bio-waste

The calculation rules for bio-waste are laid down in Commission Implementing Decision 2019/1004, and earlier in Directives 2008/98/EC as amended by Directive 2018/851, and have been summarised in Eurostat (2021) as follows:

- The calculation point is just before entering the aerobic/anaerobic process, after the initial sorting and separation activities, and subject to subtraction of either non-biodegradable materials which remain in the output, as well as all materials (including biodegradable) removed mechanically at the input or from the outputs (see Figure 10). Note that according to actual practice it is possible that the material rejected at the initial sorting and separation process (before the calculation point) might be collected (and weighed) together with the material mechanically removed from the output (after the calculation point), and thus, in order to avoid miscalculations and/or misinterpretations, it is important to keep track of both quantities separately.



* This includes only the materials sent for disposal or energy recovery. It does not include the materials that are recirculated into the next batch

Figure 10: Calculation point and calculation rules for bio-waste. As for the calculation rules: grey boxes indicate flows that are not considered in the calculation; blue boxes indicate flows that are included in the calculation; red boxes indicate flows that are subtracted from the input quantity at the calculation point; finally, green boxes indicate flows that are ignored in the calculation. Adapted from Eurostat (2021).

- Biodegradable quantities entering a process that produces compost or digestate count as recycling (in line with the previous rule), and it is not necessary to deduct evaporation or losses from biological degradation (i.e., considered inherent losses) as shown in Figure 10.
- Quantities entering other bio-waste treatment process different from aerobic/anaerobic processes that produces outputs that are not compost or digestate, only count as recycling where the quantities of outputs are similar to the input quantities and where these outputs are used as recycled product. When those output quantities are not similar to the amount of input biodegradable waste, recycled reported quantities should be scaled downwards accordingly.
- Biodegradable quantities reprocessed into materials which are to be used as fuels or other means to generate energy, which are disposed of, or which are to be used in any operation that has the same purpose as recovery of waste other than preparing for re-use and recycling, should not be counted as recycled. For processes where recycling and energy recovery of bio-waste are combined (e.g., anaerobic digestion), subject to the solid/liquid output material being used as a recycled product, the input material (net of rejects and non-biodegradable waste) is deemed to be recycled. When the output varies along the year for their different uses (i.e., compost, backfilled and thermally treated for energy production), then the amounts reportable for recycling, energy recovery and other recovery should be scaled according to the proportion of output used for each purpose.
- Where outputs (i.e., compost or digestate) are used on land, then ecological or agricultural benefits must be documented for the process to be considered recycling. This can be done either using compost standards and EoW criterion or establishing the source of the waste.

4.4.2 Existing calculation rules for compostable plastic waste

Compostable plastic can be collected together with bio-waste, if legally allowed in the Member State¹² when it presents “[...] similar biodegradability and compostability properties which complies with relevant European standards or any equivalent national standards for packaging recoverable through composting and biodegradation” as stated in the Directive 2018/851. In that case, they might enter aerobic/anaerobic processes along with bio-waste, and thus they will follow the calculation rules explained in section 4.4.1.

However, when compostable plastic waste is included in the recycled amounts, it needs to be classified/recorded under the total plastic recycling and total plastic waste generation figures. According to the Commission Implementing Decision 2019/665, this relates to the municipal compostable plastic packaging but can be extended to other compostable municipal plastic waste. To this end, the amount of compostable plastic entering a bio-waste treatment facility must be determined through waste composition analyses (acknowledging that it is very difficult to differentiate compostable materials from non-compostable plastics using visual discrimination).

¹² For example in Germany only compostable plastic bags are allowed to be collected with biowaste (no other compostable packaging).

4.4.3 Issues concerning bio-waste/compostable plastic waste calculation rules

Some of the above-mentioned rules applied to the quantification of recycled bio-waste and compostable plastic waste are not sufficiently clear. Furthermore, some calculation rules differ within the bio-waste stream depending on the technology applied (anaerobic digestion/composting versus all the remaining). This creates a discrepancy in the overall calculation framework reported in section 4.2.2 as well as a non-neutral playing field for recycling technologies. The main issues are pointed out and further discussed as follows:

- One issue concerns the inclusion of the inherent losses in the recycling yield for aerobic/anaerobic processes when producing compost or digestate. The reasoning behind this rule is that whereby organic matter is applied directly onto soils, degradation occurs spontaneously: these process losses are thus considered as 'inherent degradation of recovered organic matter' and hence should not be subtracted from the tonnages deemed to have been 'recycled' (Hogg et al., 2020). On the other hand, when the same feedstock enters a treatment different from aerobic/anaerobic processes, a downscale in the quantity deemed 'recycled' is expected based on the proportion input/output (e.g., for biochemical technologies producing starch for paper or paperboard strengthening, for pyrolysis or hydrothermal technologies producing biochar-like materials). This means that for these other technologies a mass-balance approach needs to be taken. We argue that there is no objective reason why degradation of organic matter occurring in composting/anaerobic digestion is treated differently than in other fermentation/biochemical processes if such degradation would occur anyway in a natural state. It should be noted, as an example, that the degradation rate of organic matter in composting is different than in anaerobic digestion. However, we acknowledge that another reason for this differentiation between anaerobic digestion/composting and the rest of technologies in the calculation rules is the strong need to foster technologies that help on stopping the existing degradation of EU soils, by returning significant amounts of organic matter to the soil (as stated in the EU soil strategy for 2030 (European Commission, 2021). In line with that, anaerobic digestion/composting appeared as the only technologies ready to be implemented at large scale for the biological treatment of large amounts of bio-waste feedstock (as recognized by the BREF document; Pinasseau et al., 2018) producing outputs destined to amend soil. However, we argue that emerging technologies are able to produce outputs that can be used as soil amendments, in similar quantities compared to anaerobic digestion/composting and some of them in TRLs up to 6-7 (i.e., close to the full deployment phase) (see Table 2).
- Another issue related to the inherent losses in the case of fossil-based compostable plastics is that the origin of the carbon released is not biogenic, as occurs with bio-waste and bio-based compostable plastics. The origin of the carbon will not have an impact on the recycling yields addressed herein, but it does affect the environmental impacts.
- Another element not clear from the calculation rules reported in Eurostat (2021) concerns processes where recycling and energy recovery are combined. There is a discrepancy of two rules since biogas from anaerobic digestion could be considered as an inherent loss, and thus accounted in the recycled quantities, or as material which is to be used as fuel or other means to generate energy and, thus, not considered in the recycled quantities. The general approach is to count biogas as recycling under the condition that digestate is used as a recycled product, material or substance (e.g., on soil).

- Finally, an issue appears in the need to document ecological or agricultural benefit when outputs (i.e., compost or digestate) are used on land in order for the whole process to be considered recycling. Some literature claims that there is no evidence regarding the ecological benefit of compostable plastic waste in the compost/digestate. We argue that the same could be claimed for other waste material fractions that enter aerobic/anaerobic processes along with the bio-waste stream such as paper tissues or napkins (mainly carbon-rich and containing negligible nutrients). Anyway, it is important to highlight that the criteria is the ecological or agricultural benefit of the output as a whole (as a result of treating all bio-waste fractions, not only compostable plastic waste), and not limited to the nutrient contribution (nitrogen and phosphorus). It is well documented that compost and digestate applied on land are a carbon source and a soil improver even if they do not provide nutrients. See Box B for further discussion.

Box B: Ecological and agricultural benefit on land of outputs from biological recycling processes.

As mentioned in the legislation, there is a need to document ecological or agricultural benefit of outputs (i.e., compost or digestate) on land, and it seems that the positive contribution of those as a carbon source and a soil improver, when properly produced, is beyond discussion. The problem is that compost obtained from incompatible feedstock materials and from a bad composting process, will be either immature (thus potentially phytotoxic), contaminated with foreign objects, or containing hazardous chemicals and metals. Spreading this type of compost on soil might lead to agronomic damages and, in any case, it would not bring the expected benefits to agriculture or ecological improvements. Based on that, the core of the discussion in this topic is concerning the role of compostable plastic waste in the composting or anaerobic digestion process, and the contribution to the final output (i.e, compost or digestate, respectively).

Concerning the benefit of compostable plastic waste on compost, one of the main topics is about compostable plastic waste being nutrient-free and, therefore, not bringing any benefit to the compost. Most likely, this refers to the lack of nitrogen compounds, which in fact are generally not present in these materials. However, the statement that the lack of nitrogen makes a feedstock useless for the composting process and the formation of compost seems not correct. The contribution of materials to composting can be catabolic (energetic), anabolic (structural), or both (Degli-Innocenti, 2021). It is well known that the composting process, like all biological processes, needs a balanced carbon-nitrogen ratio. For this reason, nitrogen-rich fractions (bio-waste, manure, etc.) are mixed with low-nitrogen fractions (e.g., cellulose) to avoid fermentation imbalances. Materials rich in carbon (polymers such as cellulose and biodegradable plastics) are necessary for the composting process as they bring energy and carbon. Without these components the composting process does not happen and compost is not produced. Actually, the chemical energy of feedstock evolves as heat leading to the very high temperature reached by the composting pile. The composting mass reaches temperatures of 60°C and higher, without any external heat source. High temperatures are needed to speed up the biodegradation process and to kill the pathogens present in the original waste. Thus, the carbon is oxidised to heat the composting pile and make the composting process, including pasteurisation, happen without any external energy source. Therefore, the statement that compostable plastic does not contribute to the value of the compost product, since it does not contain nutrients in its composition, is scientifically groundless (Degli-Innocenti, 2021). It is important to highlight that the amounts of compostable plastics, and their relative carbon contribution, are extremely small nowadays, in comparison to the vast volumes of bio-waste entering aerobic and anaerobic digestion.

Another topic, claimed by some stakeholders, is that compostable plastic waste contain additives whose harmlessness has not been evaluated, or that they are just as toxic as conventional plastics with regards to the chemicals they contain (Zimmermann et al., 2020). At this moment there are no rules or policies looking at avoiding the use of non-biodegradable additives, which could harm the quality of compost. Even the EN 13432 standard allows for 10% non-biodegradable additives in packages. Further research is needed in this topic.

A further discussion topic is about compostable plastic waste releasing microplastics in the final output hence promoting their dispersion in the environment (Qin et al., 2021; Wei et al., 2021). On the other hand, several studies in different EU countries provide evidence that compostable plastic fit the composting process and do not result in microplastics in the output (Edo et al., 2022; van der Zee & Molenveld, 2020). Again, further research is needed in this topic.

Apart from that, there are many other arguments not strictly related to the ecological and agricultural benefit on land that needs to be enumerated (but not considered for the scope of this study):

- Technological constraints - Some technologies are unsuitable for compostable plastic waste, causing in some cases technical problems in composting facilities where they are rejected as refusal. Besides, some stakeholders claim that biodegradation is only achieved in industrial plants (with a clear distinction between industrial composting and home composting).

- The contribution of compostable plastics, when treated in composting facilities, to the achievement of circular economy targets and their environmental performance (usually measured through LCA).

- Claimed additional co-benefits of certified compostable plastic packaging:

- Increases the separate collection of bio-waste/organic waste/food waste.

- Allows to recover bio-waste that is attached to the packaging, that would be discarded and sent to incineration or landfilling, if the packaging is sieved from the bio-waste going into composting.

- Reduces the contamination from plastics in compost.

- Reduces the moisture content and increases the bulking effect, useful when composting food waste.

4.4.4 Proposals for calculation of recycling

It is important to unify criteria and calculation rules along waste fractions and treatment technologies in order to set a common playground and avoid inconsistencies.

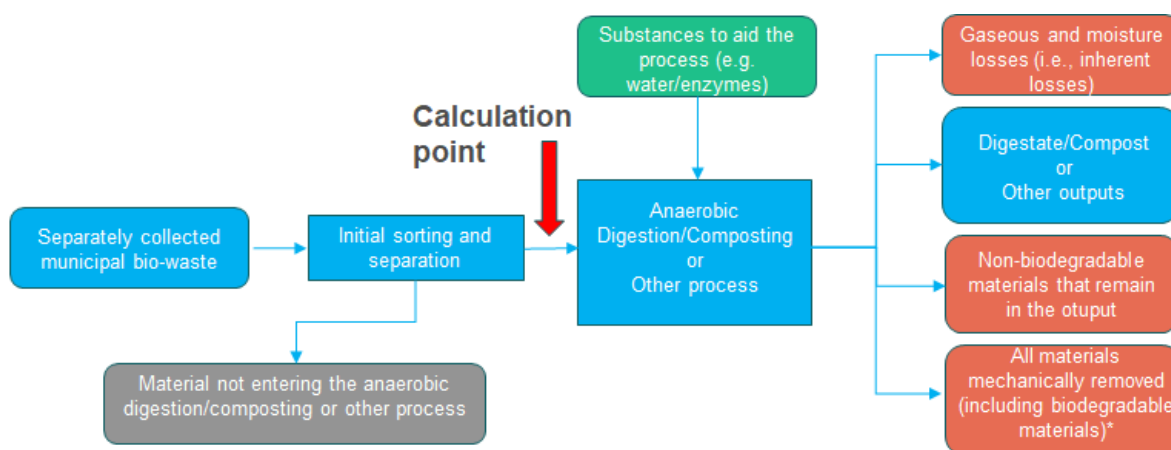
Bio-waste

When bio-waste is the only fraction entering the recycling process, in light of the issues raised in section 4.4.3, the following calculation method is proposed:

- When the process generates similar output quantities as the benchmark composting and anaerobic digestion processes, and this output is used as a recycled product, material or substance, inherent losses (i.e., evaporation and losses from biological degradation) are accounted as recycled material in the output, regardless of the process occurring, whether biological, physical, or chemical. This follows the current logic as in Eurostat (2021) (at least for aerobic/anaerobic degradation processes) (see Figure 10). The mass balance as detailed in section 4.2 is not required, and the input quantities can be claimed as recycled subject to subtraction of non-biodegradable materials which remain in the output and all materials (including biodegradable) removed mechanically at the input or from the outputs.

- When the process generates very low amount of output compared with the benchmark of composting and anaerobic digestion processes, and under the condition that this output is used as recycled product, material or substance, Inherent losses are considered as process losses instead of accounting as recycled material (see Figure 11). For simple processes the math can be simplified and input quantities can be claimed as recycled subject to subtraction of non-biodegradable materials which remain in the output and all materials (including biodegradable) removed mechanically at the input or from the outputs, and to subtraction of the inherent losses. In case of complex multi-process and/or multi-output, then the framework in section 4.2.2 for a mass balance approach can be applied.

A possible issue is the selection of the unit of measurement of the mass balance for composting/anaerobic digestion processes (i.e., dry basis or wet basis) and the quantification of the water added to the process since compost/digestate are wet products with varying water content, especially for digestate. According to the rules, water added to aid the process should be ignored in the calculation, and thus the amount of output product reported should be corrected according to the natural humidity of the product when this is placed on the market.



* This includes only the materials sent for disposal or energy recovery. It does not include the materials that are recirculated into the next batch

Figure 11: Calculation point and calculation rules for bio-waste process generates very low amount of output compared with the benchmark of composting and anaerobic digestion processes. As for the calculation rules: grey boxes indicate flows that are not considered in the calculation; blue boxes indicate flows that are included in the calculation; red boxes indicate flows that are subtracted from the input quantity at the calculation point; finally, green boxes indicate flows that are ignored in the calculation.

Compostable plastic waste

In case compostable plastic waste is collected and treated along with bio-waste, the following adaptations have to be made to the general mathematical framework presented in section 4.2.2. to be able to account and report compostable plastic recycled quantities in the total plastic recycled quantities (note that there is an obligation for packaging, acknowledging the technological difficulties earlier mentioned in section 4.43.1):

Quantification of compostable plastic waste entering the process:

The system boundaries start with the materials after the initial sorting just before entering the recycling process. If the individual material fractions (i.e., garden waste, food waste, compostable plastic waste, etc.) are only quantified before the initial sorting, then the material rejected before the treatment process must be subtracted from the input to calculate an *Input compostable plastic corrected*. If the material rejected is traceable to the specific material (e.g., compostable plastic packaging), then Equation 24 is applied (see Figure 12a)

$$\text{Input compostable plastic corrected} = \text{Input compostable plastic} - \text{Compostable plastic rejected}$$

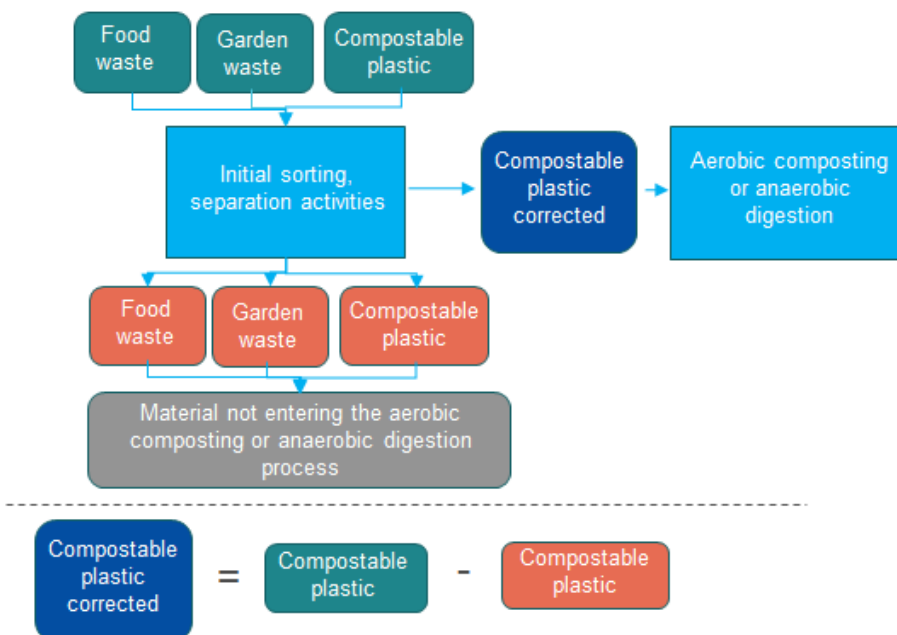
Equation 24

In case this quantity is not traceable because rejects are reported together (i.e., total rejected materials), it will be proportionated with the inputs according to Equation 25 (see Figure 12b). Note that this option is not the preferred one since it would lead to an overestimation of the *Input compostable plastic corrected* and it should only be used in case composition analyses are not possible and no further data is available.

$$\text{Input compostable plastic corrected} = \text{Input compostable plastic} - \left(\frac{\text{Input compostable plastic}}{\text{Total inputs}} \right) \cdot \text{Total rejected materials}$$

Equation 25

a)



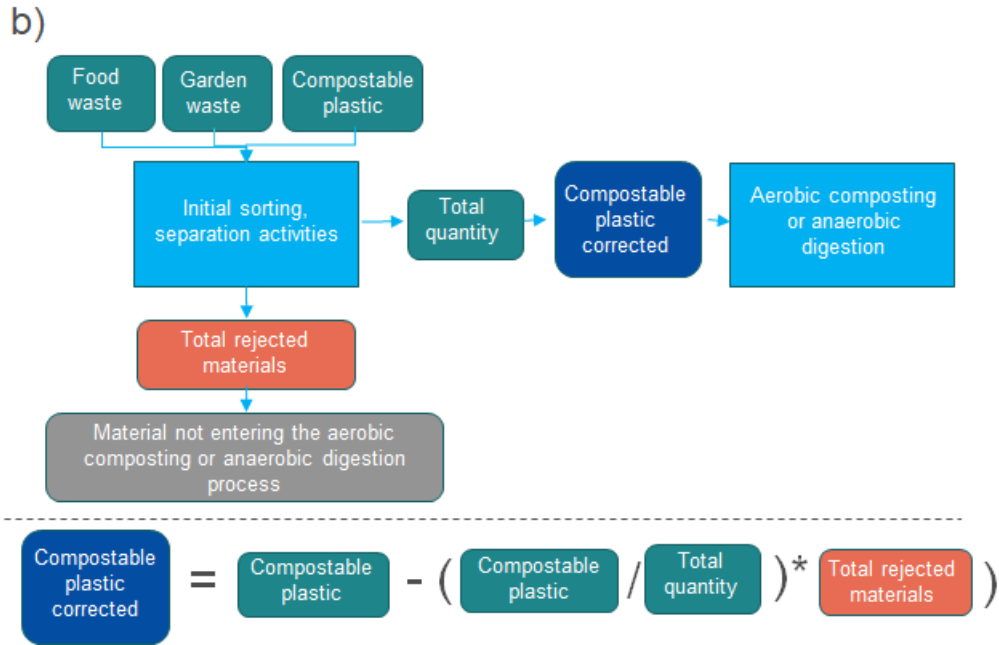


Figure 12: Quantification of compostable plastic waste entering the process. a) When the material rejected (i.e., orange boxes) is traceable to the specific material; b) when the material rejected (i.e., orange box) is not traceable because rejects are reported together.

If necessary to apply the framework as in section 4.2.2, this can be adapted by considering the compostable plastic waste as waste feedstock and the remaining bio-waste (i.e., other waste) as the virgin feedstock. Simplifying and assuming that the degradation of compostable plastic waste is the same as the remaining bio-waste¹³. Note that the mass balance approach might not be required in some cases and a simple calculation could be applied. However, similarly to what explained before for bio-waste, the following calculation method is proposed:

- When the process treating compostable plastic waste generates similar output quantity as the benchmark composting and anaerobic digestion processes, and this output is used as a recycled product, material or substance, inherent losses are accounted as recycled material in the output regardless of the process occurring, whether biological, physical, or chemical.
- On the other hand, when the process generates lower amounts of outputs compared with the benchmark of composting and anaerobic digestion processes, and under the condition that this output is used as recycled product, material or substance, inherent losses are considered as losses (instead of accounting as recycled material).

¹³ Alternatively, one should know the stoichiometry of degradation/conversion of the compostable plastic waste and the remaining bio-waste. This is hardly known, therefore a mass-based proportion seems the most reasonable approach.

5 Assessment of the impacts following changes proposed

Changing the definition of recycling and associated calculation rules is expected to have some consequences at environmental, economic and social (e.g., employment) level. While no or very few changes are expected for mechanical recycling technologies, emerging technologies are expected to increase the European recycling rates as they act as complementary solutions. This will translate into the opportunity of being able to recycle portions of waste that currently cannot be mechanically recycled, either for technical or economic reasons. These technologies that will play an important role in the future are the ones affected by the changes herein proposed. Within this section we attempt to identify possible and expected impacts (Table 6).

Table 6: Potential impacts (positive: benefits, negative: burdens) expected with the implementation of the calculation rules and the integrated definition of recycling herein presented.

DIMENSION	IDENTIFICATION	STAKEHOLDER AFFECTED	WASTE STREAM	TYPE OF WASTE	TYPE OF IMPACT	EXPECTED IMPACT	
Economic	Revenues	Chemical recyclers	Textiles	Cotton or mixed fibres	Positive	Increase in business investment due to the development of the sector and associated increase in revenue	
			WEEE	WEEE plastic component			
		Mechanical recyclers	Plastic	All plastic waste		Negative	Expected complementation with chemical recycling (residues would be chemically recycled hence replacing incineration) and associated increase in revenues
				Polyolefin, mixed plastic			
				Compostable plastics			
		Composting plants	Landfill plants	Incineration plants	Negative	Including other technologies within the calculation rules can create competition with other recycling technologies and lower revenues due to less mass entering the plant.	
		Administrative costs			European Union	Negative	Expenses expected for salaries and wages of personnel performing staff functions aimed to deal with the new legislation implemented
	Eurostat						
	Operating costs	Chemical recyclers	Textiles, WEEE, Plastic	Cotton or mixed fibres, WEEE plastic component, All plastic	Negative	Expenses expected for salaries and wages of personnel performing staff functions aimed to deal with the new legislation implemented	
							Member States
		European Union				Expenses expected for new utilities, insurances and office supplies for staff functions aimed to deal with the new legislation implemented	
		Eurostat					
	Opportunity costs	Citizens	Mechanical recyclers	Other streams	All	Null	If more recycled material is placed on the market, a potential reduction in market prices might be expected in the long-term
							Member States

Social/Economic	Working conditions	Chemical recyclers	Textiles, WEEE, Plastic	Cotton or mixed fibres, WEEE plastic component, All plastic	Uncertain	Working conditions might worsen. More pollution (or different pollutants) in the working place might be expected due to the increase in waste treated and new types of waste feedstock arriving to the plant (they might require different chemicals to be treated).		
	Employment			Cotton or mixed fibres	Positive	Increase in employment coupled with increase in business investment in the stakeholder sector.		
				WEEE Plastic component				
	Industrial research	All plastic waste		Positive	Increase in employment coupled with more investments in industrial research to make chemical recycling a competitive technology.			
		Chemical recyclers				Cotton or mixed fibres, WEEE plastic component, All plastic	Negative	Further training systems have to be set up in companies to instruct the employees about new calculation and reporting rules.
	Education and training, education and training systems			Chemical recyclers	Cotton or mixed fibres, WEEE plastic component, All plastic		Positive	More options for chemical recycling and more market opportunities might foster R&D and thus technological development.
	Technological development	Citizens				All	Negative	Living conditions of citizens living close to the plants. While release of pollutants in the short-term is considered low, the long-term potential leak is still not well understood with potential consequences for communities living nearby the plants.
	Living conditions							
	Mechanical recyclers	Other waste streams	All	Null	No social impact expected			
Environmental	Climate, energy utilization, fossil resource depletion and air quality	Chemical recyclers	Textiles, WEEE, Plastic	All plastic waste	Negative	Possible negative impacts are: increase of toxic emissions, production of GHG emissions due to high energy demand (operations) and higher release of permanent micro plastics.		
				WEEE Plastic component				
				Cotton or mixed fibres	Positive	Decrease of the environmental impact reducing GHG emissions, hazardous substances, energy consumption and fossil resource depletion due to an expected increase of recycled material.		
		Mechanical recyclers		Polyolefin, mixed plastic			Decrease of the environmental impact reducing GHG emissions, hazardous substances, energy consumption and fossil resource depletion due to avoiding incineration & landfilling with chemical recycling process.	
		Member States		Cotton or mixed fibres, WEEE plastic component, All plastic				Contribution to EU Green Deal (environmental impacts reduction, e.g., globalw warming) and Circular Economy Action Plan (more efficient use of resources).
		European Union			Mechanical recyclers	All	Null	
				Biological treatment				Plastic

In a nutshell, considering the contribution of chemical recycling within the definition of recycling and related recycling yields will likely bring to an increase in the EU recycling rate with a positive contribution towards the attainment of the EU Green Deal target of a climate neutral economy by 2050 (European Commission, 2019c). This will be accompanied by a reduction of environmental impacts, in particular GHG emissions, because of (i) avoiding incineration, i.e., diverting to recycling waste that currently is sent to incineration, and (ii) the simultaneous substitution of primary material produced from virgin source, e.g., oil. This is especially true when chemical recycling acts as complementary option to mechanical recycling offering a solution for the plastic waste that cannot be mechanically recycled and that is currently incinerated. In this context, chemical recycling may be used to process a wider scope of plastic waste, thus reducing GHG emissions, overall creating additional value from waste. Finally, the change in the definition of recycling is expected to affect costs and employment rate. Despite chemical recycling technologies being available for

some time already, it is only in the past few years that the global plastic waste challenge has been brought to the attention of many parties in an effective way. Consumers, companies, governments and NGOs, amongst others, have realized that a concerted effort, involving the entire plastic value chain, is required to address the issue. Increasing recycling yields of plastic waste has recently been put at the top of EU's and national governments' agenda. In this playing field, chemical recycling and other advanced/emerging technologies for waste recycling are expected to build recycling capacity and thus affect (waste management sector) employment and costs.

5.1 Environmental and economic implications

Quantifying environmental and economic burdens and benefits is not straightforward since the changes related to the new calculation rules are expected to mainly affect chemical recycling, which is a technology under development and at its infancy in the market penetration. Therefore, it is complicated to understand the contribution of chemical recycling on the change of the burdens and benefits. With this in mind, in the next sections we strive to provide an estimation of potential burdens and benefits associated with the changes proposed and the implementation of new calculation rules that are expected to support the development of emerging technologies.

5.1.1 Administrative burdens

Complying with the provisions of regulations leads to administrative burdens for businesses. Among these administrative burdens, there are certain costs associated with administrative activities performed to comply with administrative obligations included in legal rules (European Commission, 2006), such as reporting, registration and assessment needed to provide information. Thus, following the change in the definition of recycling and the new calculation rules, administrative burdens are expected to comply with obligations stemming from government regulations.

The EU Standard Cost Model (European Commission, 2021) is a model designed to quantify and present the administrative burdens arising from regulations per country over a certain period of time. The standard cost model has been designed to fit the structure of the regulations. To comply with an information obligation (e.g., an obligation to obtain a permit), businesses have to procure the required pieces of data that constitute the information obligation. Collected messages/data have to be delivered by a certain amount of companies and a certain amount of times per year. In order to be able to deliver the data/messages, businesses have to perform certain administrative activities. Each activity is carried out in a certain time resulting in costs for companies that need to pay the wages to the employees performing the abovementioned tasks.

Following the equations presented in European Commission (2021), the calculation of the administrative costs is displayed in Equation 26. Administrative net costs are calculated by multiplying the average cost of the new required administrative activity N (P_N) with the total number of times that activity N is performed throughout the year (Q_N) subtracted by the cost of administrative activities removed R due to the implementation of the new obligations (at EU/national level). The second term of Equation 26 avoids double-counting (old) administrative costs that are replaced by new ones.

$$Administrative\ cost = \sum_{N=0}^k (P_N \cdot Q_N) - \sum_{R=0}^j (P_R \cdot Q_R)$$

Equation 26

Where:

P_N = Price of a new administrative action N , with $N=0\dots k$

Q_N = Quantity of new administrative action N , with $N=0\dots k$

P_R = Price of a removed administrative action R , with $R=0\dots j$

Q_R = Quantity of a removed administrative action R, with R=0...j

The price of a new administrative action N (P_N) is determined according to Equation 27:

$$P_N = \text{Tariff} \cdot \text{Time}$$

Equation 27

Where *Tariff*, measured in € per hour, includes both the internal tariff (i.e., the hourly rate of the person in the company who deals with the new administrative action N) and the external tariff (i.e., the hourly rate of the person outside the company who deals with the information received). *Time* represents the time, in hours per year, that takes a business to perform a certain action.

The quantity of a new administrative action (Q_N) is determined according to Equation 28:

$$Q_N = \text{Number of businesses} \cdot \text{Frequency}$$

Equation 28

Where the *Number of businesses* refers to the number of businesses to which the regulations applies. Therefore, the target group of the regulation and the number of businesses involved will need to be ascertained. The *Frequency* is the number of times that a business delivers data/messages per year.

Applying Equation 26-Equation 28, it is possible to estimate the administrative costs required for the implementation of the new calculation rules herein presented. Due to the new calculation rules, company X will be obliged to draw up and publish an annual statement in accordance with the EU country's regulation. Notice that in our study we assume that no obligations are removed due to the implementation of the new ones (i.e., $\sum_{R=0}^j (P_R \cdot Q_R) = 0$), as chemical recycling technologies are not included in previous obligations. Based on direct information from SHs, it is expected that company X makes use of the services of an accountant, hired via an accountants' office, charging € 300 per hour for a total of 80 hours to draw up the annual statement. The company will be obliged to correlate and pass on the information that the accountant needs, which is assumed to take 2 hours at a fee of 70 € per hour (including the time of the company to upload the annual statement or the communication with the EU body). Based on these assumptions, the price of the new administrative actions for company X would be 24,140 € per year (Equation 29).

$$P = (70 \text{ €/hour} \cdot 2 \text{ hour/year}) + (300 \text{ €/hour} \cdot 80 \text{ hour/year}) = 24,140 \text{ €/year}$$

Equation 29

The obligation to draw up an annual statement does not only apply to company X but to all legal entities in the EU. Assuming that in 2030 a total chemical recycling capacity of 3.1 Mt/year (corresponding roughly to an input of 5Mt/year; SISTEMIQ, 2022) and an average 50,000 t/year per plant (SISTEMIQ, 2022), we can estimate that there will be a total of 100 legal entities operating in the EU and dealing with the implementation of the calculation rules. We can assume that they are all obliged to draw up an annual statement once per year, resulting in a total of 100 new administrative actions per year (Equation 30).

$$Q = 100 \cdot 1 \text{ year} = 100 \text{ per year}$$

Equation 30

Therefore, the (net) administrative costs (i.e., the total costs for complying with the new calculation rules) equal 2.14 M€ per year (Equation 31).

$$\text{Administrative costs} = 24,140 \frac{\text{€}}{\text{year}} \cdot 100 = 2,140,000 \frac{\text{€}}{\text{year}} = 2.14 \frac{\text{M€}}{\text{year}}$$

Equation 31

5.1.2 Economic and environmental assessment

The cost-benefit analysis is herein presented on the basis of two future recycling scenarios for 2030. In the recycling scenario W/O (i.e., without chemical recycling), the projected amount of selected packaging plastic waste generated is recycled through mechanical recycling only. In the scenario W/ (i.e., with chemical recycling) the contribution of chemical recycling technologies in recycling the projected amount of selected packaging plastic waste generated is considered. Specifically, among the variety of chemical recycling technologies, pyrolysis, gasification, glycolysis, methanolysis and depolymerization are considered.

The cost-benefit analysis (section 5.1.4) is complemented with an environmental assessment in which only GHG emissions (section 5.1.5) are quantified to have a glimpse of the potential impacts of the emerging technologies herein considered. Finally, the results obtained with the GHG accounting are further monetised to also show their burden in terms of costs.

5.1.3 Projection of selected plastic packaging waste generation

The amount of plastic packaging waste generated is projected out to 2030 using linear regression based on the historical packaging waste generation reported in Eurostat from 2010 to 2018 (Eurostat, 2022). Specifically, among the variety of plastic packaging types, four are addressed within this study: Polyethylene Terephthalate (PET), (expanded)Polystyrene ((E)PS), Polyethylene (PE) (rigid and flexible), and Polypropylene (PP) (rigid and flexible). The selected plastic packaging waste is subjected to separate collection and sorting at material recovery facilities, which create eight separate bales to be fed into either recycling facilities or incineration for energy recovery depending on the type. The quantity of the eight sorted bales, which are the input for the different recycling scenarios of this study (i.e., W/O and W/), can be found in Table 7. Notice that the mixed polyolefins (MPO) bale is composed of mixed PO (i.e., PE and PP) rigid and flexible that are not correctly sorted into their primary bale (e.g., PE rigid that is not sorted into PE rigid bale might still be sorted into MPO bales), and that the mixed plastics bale is composed of mixed PET, (E)PS, and PO (Table 7).

Table 7: Quantity of the sorted bales in 2030.

Sorted bales	Quantity (in tonne)
PET	3,811,000
(E)PS	349,000
MPO	940,000
PE Flexible	2,727,000
PP Flexible	111,000
PE Rigid	1,856,000
PP Rigid	641,000
Mixed Plastics	962,000
Total	11,397,000

As mentioned before, two recycling scenarios are investigated in this study that are the recycling scenario W/O (Figure 13a) and W/ (Figure 13b) chemical recycling. In the scenario W/O it is assumed that PET, (E)PS, PE Flexible, PP Flexible, PE Rigid, and PP Rigid

bales are sent to a mechanical recycling facility, whilst MPO and Mixed Plastic bales are sent to incineration with energy recovery. On the other hand, in the scenario W/ it is assumed that there is only a small competition between chemical and mechanical recycling to obtain the sorted bales from MRFs as feedstock for their recycling operations. Specifically, it is assumed that 10% of mass of the sorted PET bale and (E)PS bale is forwarded to chemical recycling (i.e., depolymerization such as glycolysis, methanolysis, etc.), while the remaining 90% is still forwarded to mechanical recycling. In the scenario W/, PE Flexible bale, PP Flexible bale, PE Rigid bale, and PP Rigid bale are treated via mechanical recycling. Further, in the scenario W/, it is assumed that chemical recycling complements mechanical recycling by accepting as inputs MPO and Mixed Plastic bales and the rejects of mechanical recycling process. In this study, it is assumed that pyrolysis would receive all MPO and Mixed Plastic bales, while gasification would receive 95% of the rejects from mechanical recycling and pre-treatment of pyrolysis (i.e., the 'feedstock preparation' before the plastic wastes are fed into the pyrolysis reactor), while the rest of the reject (5%) are sent to incineration for energy recovery.

The results of the projected (aggregated) material flows in both scenarios (W/O and W/) can be found in Figure 13. In scenario W/O, 8,15million tonnes of recyclates are produced, while 3,25 million tonnes of waste are incinerated (Figure 13a). On the contrary, the amount of waste being incinerated is significantly reduced to 608,7 ktonnes in the scenario W/ (Figure 13b). Further, the total recyclates produced from mechanical recycling is 7,8 million tonnes and the total polymer production from depolymerization is 374,4 ktonnes. The output of pyrolysis, i.e., naphtha and wax, equals 958,6 ktonnes while syngas for electricity production from gasification equals 1,7 million tonnes. By taking into account the definition of 'recycling' under the Waste Framework Directive (i.e., secondary materials production, excluding the use for energy sources), the quantities of recyclates and other products (i.e., wax and naphtha) are estimated to increase from 8,15 million tonnes in scenario W/O to 9,12million tonnes in scenario W/. The increase of mass of recycled materials corresponds to an estimated increase in the recycling yield from 71% to 80%.

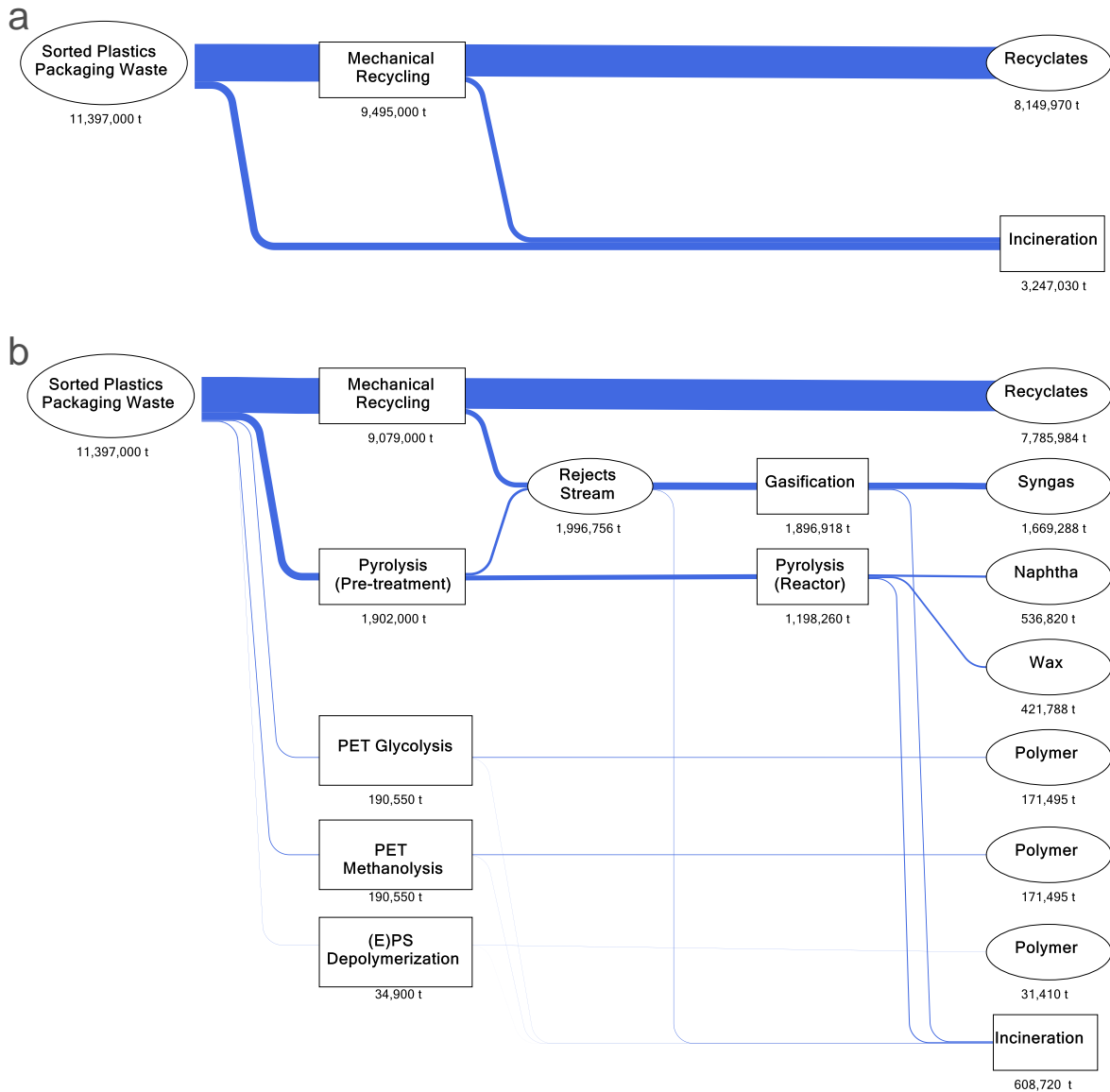


Figure 13: Aggregated plastic flow of the two scenarios: (a)W/O chemical recycling and (b)W/ chemical recycling.

5.1.4 Economic assessment

The total costs associated with the waste management technologies, expressed as EUR2020 per tonne of waste, are calculated as the sum of capital expenditures (CAPEX), operational expenditures (OPEX), pre-treatment and feedstock waste costs, to which the revenues are subtracted. The CAPEX represents the investments in assets used for production, transformation and distribution, as well as for refurbishment, upgrades, new construction and the replacement of capital assets. Also included within CAPEX are the investments made in Research & Development that are directed towards the development of new assets or production technologies.

For the case of recycling, it is assumed that the CAPEX accounts for the cost of the building, the annual interest rate (assumed at 5%), the lifetime of the buildings and of the equipment (assumed to be 20 years), and the relative project costs. The OPEX of recycling includes the annual insurance, building and equipment maintenance, and costs of energy (electricity, heat and diesel). Notice that operational hours are assumed to be 8,000 h/y and if only the total initial investment was provided OPEX was calculated as 10% of it. Overall, total costs associated with each technology are estimated and adjusted for inflation

at 2020. Table 8 summaries the assumptions and data used in the current assessment on the most important chemical recycling technologies.

Table 8: Estimation of the main costs associated with the most important chemical recycling technologies such as pyrolysis, gasification, methanolysis and depolymerization. Costs associated with mechanical recycling and incinerator are also reported. CAPEX, OPEX, pre-treatment and feedstock costs, revenues and total costs are adjusted for inflation at EUR 2020 (HICP=105.76 according to Eurostat; 2021).

	Chemical Recycling					Mechanical recycling ¹	Incineration ²
	Pyrolysis	Gasification ³	Depolymerization	Glycolysis	Methanolysis		
References	KIT, 2018	KIT, 2018	KIDV, 2018	KIDV, 2018	Carducci et al., 2020	Andreasi Bassi et al., 2020	Andreasi Bassi et al., 2020
Capacity (Mt waste/y)	0.3	0.12	0.02	0.02	0.04	0.20	0.15
CAPEX (EUR/t waste)	141	191	107	77	75	78	121
OPEX (EUR/t waste)	21	32	618	572	94	156	51
Pre-treatment and others (EUR/t waste)	21	66	0 ⁴	0 ⁴	0 ⁴	57	30
Feedstock waste cost (EUR/t waste)	0 ⁵	0 ⁵	51	102	650	227	0 ⁵
Revenues (EUR/t waste)	51	72	426	413	527	535	242
Total costs (EUR/t waste)	132	217	350	337	292	-17	-41

¹ Values presented are calculated as mean average of the values reported in Andreasi Bassi et al. (2020) for scenario A1 (door-to-door collection and plastic packaging management) considering clear, light blue and mixed coloured PET, HDPE, PP, Flexible and MPO.

² Values adjusted assuming the thermal load of plastic equal to 30 MJ/kg.

³ Values calculated as mean average of values provided by KIT (2019) for fixed bed, fluidized bed and entrained flow gasification of residual household waste, automotive shredder residues and electronic shredder residues.

⁴ Pre-treatment costs included in the CAPEX value.

⁵ Assuming a feedstock with very low quality with a negligible price.

Combining the total costs resulting from the CAPEX, OPEX, pre-treatment costs, feedstock costs, and revenues expressed as EUR per tonne of waste (Table 8) with the projected material flows in 2030 for each waste management technology in the two different scenarios (W/O and W/, Figure 13), it is possible to estimate the total costs (in MEUR) associated with the two scenarios (Figure 14).

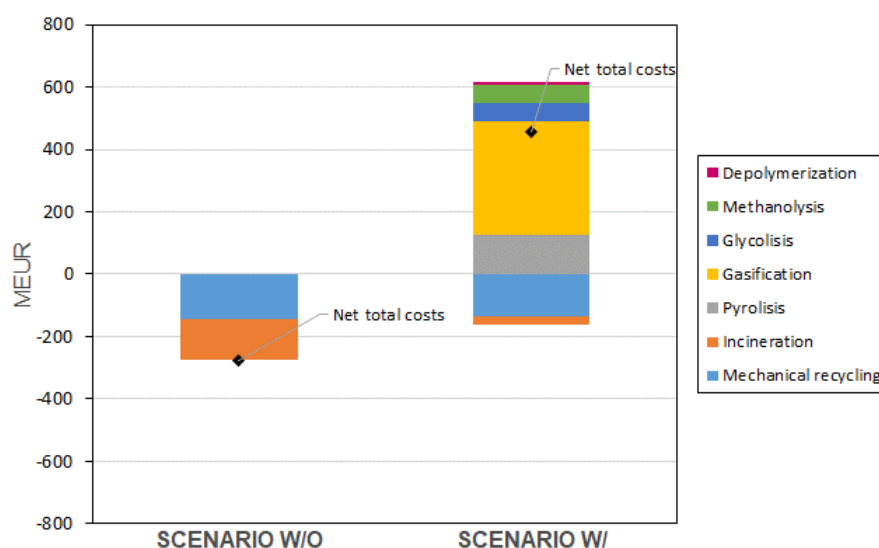


Figure 14: Total costs (MEUR) including CAPEX, OPEX, pre-treatment and feedstock costs, and revenues associated with the projected material flow (2030) in both scenarios (W/O and W/). Positive contributions mean economic costs, while negative mean revenues. A net total cost below the zero line represents a revenue.

As shown in Figure 14, we obtain negative net total costs (-276 MEUR; negative means net income) in the scenario W/O and positive net total costs (455 MEUR) in the scenario W/. This is mainly due to the investment required to develop the emerging chemical recycling technologies. It is clear that the high amount of revenues, especially for depolymerization, methanolysis and glycolysis (Table 8) is not sufficient to compensate the high amount of costs. In particular, investment costs (CAPEX) are remarkable for pyrolysis and gasification (141 and 191 EUR/t of waste, respectively) as well as operational costs (OPEX) for depolymerization and glycolysis (618 and 572 EUR/t of waste, respectively). Acknowledging data uncertainties and limitations, the key message is that the development of chemical recycling technologies in the coming years needs substantial investments that cannot be entirely balanced from the associated revenues. It should be noted that our estimation does not take into account the economy scale of technologies and it is based on different assumptions. Therefore, these results should be used and interpreted with care.

5.1.5 GHG emission savings

The estimation of GHG emissions (by means of CO₂-eq emission) of the above-introduced scenarios (i.e., W/O and W/) is done by multiplying the mass flow (in tonne) with the emission factors of each process (in kg CO₂-eq per tonne flow). The emission factor of mechanical recycling (i.e., 500 kgCO₂-eq/t for PE Films, PP Films, PE Rigid, and PP Rigid bales; 550 kgCO₂-eq/t for PET bales; 980 kgCO₂-eq/t for (E)PS bale) is obtained from the

study of Garcia-Gutierrez et al. (2023) and Civancik-Uslu et al. (2021). For depolymerization of PET and (E)PS bales, the emission factors are also taken from Garcia-Gutierrez et al. (2023). As for the pyrolysis (cracking, condensation, and distillation), the emission factors are obtained from the study of Civancik-Uslu et al. (2021) and Garcia-Gutierrez et al. (2023). For gasification, the emission factor is estimated from the study of Ardolino et al. (2018). Finally, the emission factor for incineration with energy recovery is estimated from the study of BASF (2020) and Garcia-Gutierrez et al. (2023). The summary of the emission factors per tonne input waste can be found in Table 9.

The secondary materials produced from mechanical recycling are assumed to substitute their virgin counterpart, e.g., PET recyclates substitute fossil-based PET granulates. The pyrolysis products are assumed to replace virgin production of naphtha and wax as shown in the study of Civancik-Uslu et al. (2021). The gasification products, i.e., syngas, and incineration are assumed to substitute energy (i.e., electricity and/or heat) as shown also by Cossu et al. (2017). The emission factors of the European projected energy mix in 2030 (averaged value) are assumed to be 0.2732 kg CO₂-eq/kWh for electricity and 0.1504 kg CO₂-eq/MJ for residential heating (Andreas Bassi et al., 2020). The average yield of electricity and heat from incineration plant are assumed to be 11% and 33% (Andreas Bassi et al., 2020; COWI A/S & University, 2019), respectively, with the lower heating value of 36 MJ/kg mixed plastics waste (Saveyn et al., 2016). Lastly, the estimation of the net environmental impact (in kg CO₂-eq) is calculated as the delta between environmental burdens and savings.

Table 9: Estimation of the emission factors and savings for Mechanical Recycling, Depolymerization, Pyrolysis, Gasification, and Incineration of the selected plastic packaging waste. The values are shown in kg CO₂-eq per tonne input waste.

Technology	Emission factor [kg CO ₂ -eq/tonne input waste]	Source
Mechanical recycling	500 – 980 ¹	Garcia-Gutierrez et al. (2023); Civancik-Uslu et al. (2021)
Depolymerization of PET (Glycolysis)	950	Garcia-Gutierrez et al. (2023)
Depolymerization of PET (Methanolysis)	1,400	Garcia-Gutierrez et al. (2023)
Depolymerization of (E)PS	600	Garcia-Gutierrez et al. (2023)
Pyrolysis	1,000 – 1,500	Garcia-Gutierrez et al. (2023); Civancik-Uslu et al. (2021)
Gasification	2,800	Ardolino et al. (2018)
Incineration	3,000	Garcia-Gutierrez et al. (2023); BASF (2020)
Substituted Polymer from Mechanical Recycling	1,900 – 2,700 ²	Garcia-Gutierrez et al. (2023); Civancik-Uslu et al. (2021)

Substituted Material from PET Depolymerization (Glycolysis)	2,800	Garcia-Gutierrez et al. (2023)
Substituted Material from PET Depolymerization (Methanolysis)	1,900	Garcia-Gutierrez et al. (2023)
Substituted Material from (E)PS Depolymerization	2,050	Garcia-Gutierrez et al. (2023)
Substituted Naphtha and Wax from Pyrolysis	700	Civancik-Uslu et al. (2021)
Substituted Energy from Gasification	2,125 ³	European Environment Agency (2022); Khoo (2019)
Substituted Energy from Incineration	2,000	Garcia-Gutierrez et al. (2023); BASF (2020)

¹ The value ranges from 500 kg CO₂-eq per tonne PE or PP recycling, 550 kg CO₂-eq per tonne PET recycling, and 980 kg CO₂-eq per tonne (E)PS recycling, after Garcia-Gutierrez et al. (2023) and Civancik-Uslu et al. (2021)

² The value ranges from 2,000 for substitution of LPDE and HDPE to 2,700 for substitution of PET

³ The value is estimated from the amount of substituted energy per tonne input of gasification, i.e., 27,800 MJ equivalent to 7,700 kWh, after Khoo (2019). The emission factor per kWh is estimated from European Environment Agency (2022), i.e., 0.273 kg CO₂-eq per kWh to be replaced by syngas.

The results of the GHG accounting of the two different scenarios in 2030 can be found in Figure 15, where values above the zero line represent burdens and values below the zero line are savings. The diversion of plastic from incinerated and the implementation of chemical recycling incurs more environmental savings compared to the W/ scenario. It is estimated that the environmental savings in the scenario W/ would increase by up to 5% compared to the scenario W/O, equivalent to the reduction of 0.5 Mtonnes of CO₂-eq emissions (here estimated for the packaging sector only).



Figure 15: Results of GHG emissions, expressed as Mt CO₂-eq, of the two scenarios (Scenario W/ and Scenario W/O) in 2030. Notice that values above the zero line are burdens, while values below the zero lines are savings.

As shown in Figure 15, the net environmental impact is reduced from -12.66 Mt CO₂-eq in the scenario W/O to -13.13 Mt CO₂-eq in the scenario W/. The improvement is mainly driven by the higher net environmental performance of pyrolysis and gasification compared to incineration, e.g., the C footprint of pyrolysis is 800 kg CO₂-eq /tonne input compared to 875 kg CO₂-eq /tonne input for incineration in 2030. Overall, the burdens decrease from 14.85 to 14.65 Mt CO₂-eq going from the scenario W/O to the scenario W/, while savings increase from 40.18 to 40.91 Mt CO₂-eq. The results in Figure 15 show potential environmental benefits from recycling plastic through emerging technologies, such as chemical recycling. Finally, in order to understand the potential effect of GHG emissions in economic terms, the results obtained with the GHG accounting were further monetised by applying the carbon price suggested in van Essen et al. (2019) (we use the central value for the price of CO₂ for the short-and-medium-run costs up to 2030, by using an avoidance cost approach), i.e., 104 EUR₂₀₂₀/tCO₂-eq. Although the scenario W/ incurs lower externalities of GHG emissions (1,523.6 MEUR₂₀₂₀) with respect to the scenario W/O (1,544.4 MEUR₂₀₂₀), their difference is negligible. This is due to the slight difference in terms of burdens between the two scenarios (Figure 15).

It should be noted that our estimations do not take into account the potential improvement of the chemical recycling products, such as further processing steps of naphtha and syngas into monomer and basic chemicals through steam crackers, Fischer Tropsch process or methanol to olefin process that is technologically feasible (Kusenberget al., 2022; Salaudeen et al., 2019). Further processing of pyrolysis and gasification products might incur more environmental savings as the monomer can be used as feedstock for polymer production and base chemicals can be used as feedstock in the petrochemical industry (SISTEMIQ, 2022). Hence, bearing these limitations in mind alongside the uncertainty and limitations of the (secondary) data used to perform the calculations, results should be seen as preliminary and used with care.

6 Technical proposal synthesis

6.1 Mass balance approach

The following recommendations are made with regards to the mass balance and calculation rules of recycling yields for technologies producing a mix of fuel, energy, and materials such as chemical recycling technologies:

- The mass balance procedure herein presented could be used as an EC guidance or integrated in Commission Implementing Decisions. The mass balance approach proposes rules and procedures for calculating the share of recycling across all technologies (not the recycled content). While this is generally applicable to all technologies, it is especially relevant for chemical recycling and similar multi-output systems. For systems as mechanical recycling, there is no need to apply mass balances.
- The mass balance herein presented can be applied to calculate the recycling yields of a specific system, where the system boundaries include the recycling process. Specifically for the case of chemical recycling, the system boundaries should include any chemical re-processing, including any refining/purification processes, up to the production of chemicals and/or materials that do not need further treatment prior to use for product manufacturing.
- In the mass balance, for each process, the inputs should be distinguished between waste- and virgin feedstock, as well as co-materials. Concerning the outputs, for each process these should be distinguished between the intermediates (that is the material going through a subsequent process), the energy recovered, the output materials obtained, and possible losses.
- By properly defining all inputs and outputs, the mathematical framework herein proposed calculates (i) the recycling yield of the recycling process defined within the system boundaries; (ii) the energy recovery yield of the recycling process defined within the system boundaries; (iii) the loss yield of the recycling process defined within the system boundaries.
- The scope of the mathematical framework is based on traceability and chain of custody. The starting point of the methodology herein presented follows the standard ISO 22095 (ISO, 2020). According to ISO 22095 the amount of either waste or virgin feedstock would be considered identical in terms of mass and, therefore, identical within the calculation rules of the mass balance. However, for our purposes, the estimation of the yields should only be based on the amount of waste feedstock thus excluding the virgin feedstock, even if it is part of the input. Therefore, an adjustment of the method proposed by the ISO 22095 is herein presented in order to take into account the proportion between waste and virgin feedstock as input to the recycling process and the corresponding calculation of recycling, energy recovery and loss yields. As a general rule, we propose that if the mathematical relationship between input (waste and virgin feedstock) and output (i.e., the stoichiometry) is known, then a chain of custody method with high physical presence is used. On the other hand, if this relationship is unknown, the proposed mass balance approach can be applied.
- The main limitation of the proposed mass balance is the traceability, as subsequent processes (e.g. pyrolysis is followed by refining) and associated operators may be involved in the recycling supply chain. We envisage the following possible hierarchy of options for traceability and reporting:
 - o The recycled quantities are reported at the level of the final transformation, using the mass balance approach and appropriate traceability schemes.

- The recycled quantities are reported by the first waste management operator, e.g., pyrolyzer or gasifier, adjusted via a mass balance approach with information on conversion and yields provided by the downstream operators via traceability schemes.
 - The recycled quantities are reported by the first waste management operator, e.g., pyrolyzer or gasifier, adjusted via a mass balance approach with conversion factors/yields provided by the Commission (if traceability downstream is not possible, i.e. the options above). This last option would not need traceability schemes in place and would be based on default conversion and yield factors based on literature data. To derive these factors, the Commission could make use of existing internal studies¹⁴ or launch a simple literature review study with the aim of aggregating all the data collected by existing Commission and non-Commission studies on chemical recycling technologies such as pyrolysis and gasification.
- Conforming with the calculation rules proposed, a calculation framework developed in Microsoft Excel has been made available to support researchers and SHs (Annex 4).

6.2 Biodegradable and compostable plastic waste

The following recommendations are made with regards to the calculation rules of recycling yields for biodegradable waste and compostable plastic waste:

- Rules on the calculation of the attainment of the (recycling) targets should explicitly include other possible technologies treating biodegradable waste apart from aerobic/anaerobic treatments, provided that their output amount is similar to the benchmark of the composting/anaerobic digestion processes, and is used as a recycled product, material or substance. This benchmark has been calculated as the lowest conversion efficiency value among all possible processes (i.e., composting, anaerobic digestion, and anaerobic digestion + composting), and the value reached is 15% (wet mass %)¹⁵. Besides, the concept of “recycled content” for biodegradable waste treatments needs to be clarified and/or rephrased. The recycled content is defined dividing the total mass of recycled material in a product by the total mass of the product. It is clear from this definition that the recycled content concept applies to products that incorporate recycled material, while the calculation rules herein discussed are meant to calculate the recycling yields of processes that result in a recycled material. Thus, according to this, the following adaptations of different articles within the legislation are proposed:
 - For legislation referring to municipal waste, Article 1.13 of the Directive (EU) 2018/851 that inserts article 11a(4) could read (proposed inserted text in **bold**, proposed eliminated text ~~striketrough~~) *“For the purpose of calculating whether the targets laid down in points (c), (d) and (e) of Article 11(2) and in Article 11(3) have been attained, the amount of municipal biodegradable waste that enters **a recycling operation (e.g., aerobic or anaerobic treatment, or any other technology)** may be counted as recycled where that ~~treatment~~ **operation** generates compost, digestate, or*

¹⁴ For example, the study commissioned by DG GROW and performed by JRC (Garcia-Gutierrez et al., 2023; in press).

¹⁵ Official data published on the ISPRA report (Italian National Institute for Environmental Protection and Research) and processed by the CIC (Italian Composting and Biogas association) (CIC, personal communication, July 25, 2022) showed that the average compost production (% wet mass produced over input biowaste treated) for the integrated AD+C process, which is the typical Italian approach, on food waste and other bio-waste fractions is 15% (average for the time period 2018 – 2020). This value is around 22% for compost production from mainly food waste and around 35% for compost production from mainly green waste. Around the EU there are also plants performing just AD generating fresh digestate used directly as fertilizer (e.g. Germany) that present values much higher than 35%.

*other output with a similar quantity of recycled content in relation to input **with a similar output quantity**, which is to be used as a recycled product, material or substance. Where the output is used on land, Member States may count it as recycled only if this use results in benefits to agriculture or ecological improvement.” Similarly, Recital (48) of the Directive (EU) 2018/851 could read: “Where the calculation of the recycling rate is applied to **any recycling operation** ~~aerobic or anaerobic treatment~~ of biodegradable waste, the amount of waste that enters **that operation** ~~aerobic or anaerobic treatment~~ can be counted as recycled provided that such treatment generates output which is to be used as a recycled product, material or substance. While the output of such treatment is most commonly compost or digestate, other output could also be taken into account provided that it contains comparable quantities of recycled content in relation to the amount of the treated biodegradable waste **that it generates similar output quantities taking as benchmark the composting and anaerobic digestion process**. In other cases, in line with the definition of recycling, the reprocessing of biodegradable waste into materials which are to be used as fuels or other means to generate energy, which are disposed of, or which are to be used in any operation that has the same purpose as recovery of waste other than preparing for re-use and recycling, should not be counted towards the attainment of the recycling targets.”*

- In the same line, for legislation referring to packaging waste, Article 1.6 of the Directive (EU) 2018/852 (European Commission, 2018b) that inserts article 6a(4) (that is the same as Article 47.8 of the proposal EC 2022/0396 (European Commission, 2022)) could read (proposed inserted text in **bold**, proposed eliminated text ~~strikethrough~~) “... the amount of biodegradable packaging waste that enters **a recycling operation (e.g., aerobic or anaerobic treatment, or any other technology)** may be counted as recycled where that ~~treatment~~ **operation** generates compost, digestate, or other output with a similar quantity of recycled content in relation to input **with a similar output quantity**, which is to be used as a recycled product, material or substance. Where the output is used on land, Member States may count it as recycled only if this use results in benefits to agriculture or ecological improvement.” Similarly, Recital (17) of the Directive (EU) 2018/852 (that is the same as Recital (112) of the proposal EC 2022/0396 (European Commission, 2022)) should read: “Where the calculation of the recycling rate is applied to **any recycling operation** ~~aerobic or anaerobic treatment~~ of biodegradable packaging waste, the amount of waste that enters **that operation** ~~aerobic or anaerobic treatment~~ can be counted as recycled provided that such treatment generates output which is to be used as a recycled product, material or substance. While the output of such treatment is most commonly compost or digestate, other output could also be taken into account provided that it contains comparable quantities of recycled content in relation to the amount of the treated biodegradable packaging waste **that it generates similar output quantities taking as benchmark the composting and anaerobic digestion process**. In other cases, in line with the definition of recycling, the reprocessing of biodegradable packaging waste into materials which are to be used as fuels or other means to generate energy, which are disposed of, or which are to be used in any operation that has the same purpose as recovery of waste other than recycling, should not be counted towards the attainment of the recycling targets.”
- Criteria concerning the inclusion of inherent losses should be harmonised (or further clarified) along all processes and technologies used for recycling bio-waste and compostable plastic waste regardless of the process occurring, whether biological,

physical, or chemical. As they are now, Recital (46) of the Directive (EU) 2018/851 seems to be in line with the technology neutrality: "... Losses in weight of materials or substances due to physical or chemical transformation processes inherent in the recycling operation whereby waste materials are actually reprocessed into products, materials or substances should not be deducted from the weight of the waste reported as recycled", but it links the concept of inherent losses to the recycling operation. Inherent losses have not been defined for the different recycling operations, and interpretation leads to think that emissions occurred through biological processes are inherent but emissions occurred through chemical processes are not, e.g., CH₄ produced in an anaerobic digestion process is inherent since that bio-waste could be decomposed anaerobically into that product, meanwhile CH₄ produced in a hydrothermal carbonization reactor treating the same bio-waste is not considered inherent because this process would have never occurred naturally. A possibility is that inherent losses could be linked to the material treated as shown in the example from Eurostat report (Eurostat, 2021)(Eurostat, 2021; p. 49) (e.g., extruded filter cake or fine dust for plastic, water and CO₂ for bio-waste, slag for metals, glass fines for glass, inks and dragged fibres for paper) and applied to the technologies that fulfil the criteria imposed (i.e., provided that it generates similar output quantities taking as benchmark the composting and anaerobic digestion processes, and is used as a recycled product, material or substance). Thus, a formal definition of inherent losses for each material would need to be proposed. For bio-waste and compostable plastic waste, inherent losses could include CO₂, CH₄ and water.

- Identify compostable plastic packaging waste entering whatever recycling operation (not just aerobic/anaerobic treatment). According to this, Article 7 of the Commission Implementing Decision (EU) 2019/665 that inserts article 6c could read (proposed inserted text in **bold**, proposed eliminated text ~~striketrough~~): "*Where biodegradable packaging that is subject to ~~aerobic or anaerobic treatment~~ **recycling operations** is included in the recycled amounts for the respective packaging material, the amount of biodegradable packaging in biodegradable waste shall be determined by performing regular composition analyses of the biodegradable waste entering those operations. Biodegradable packaging waste that is removed before, during or after the recycling operation shall not be included in the recycled amounts*". It is recommended to further determine the amount of compostable plastic waste (not only packaging), acknowledging the technical difficulties.

It is recommended to use a simplified calculation method when possible (i.e., simple subtraction of non-accounted outputs from the inputs). Otherwise, when needed due to multi-input/multi-output processes, we recommend to follow the mass balance approach presented in this document.

6.3 Disclaimer

It should be stressed that, at this point, the European Commission has not undertaken any steps for the revision of the definition of recycling. Therefore:

- The present document contains technical proposals and does not constitute the official opinion of the European Commission regarding the revision of the definition of recycling and related calculation rules;
- The present document does not constitute any commitment by the European Commission to start work on the revision of the definition of recycling and related calculation rules.

7 Quality of recycling: state-of-the-art

7.1 Rational for defining quality of recycling

Whereas recycling certainly maintains the resources in circulation within the material economy, high-quality recycling preserves the characteristics of materials, which make them most useful. This translates into avoiding the loss of material characteristics that are relevant to its re-use in key product sectors. In general, high quality secondary raw materials are necessary for expanding the use of recycled content in broader product applications, ultimately enabling a more circular economy. Consumers of secondary raw materials frequently raise concerns about the quality of sourced material. This is especially true for plastics recycling, where the inability to source material of sufficient quality is a key limitation on the amount of secondary plastic that can be utilised. An example for this is the use of plastic packaging in the food industry, where high quality is required for food contact applications. A definition of high-quality recycling could help developing policies focused on improving the quality of recycling outputs by the entire recycling chain, ultimately ensuring a greater level of resource circularity.

7.1.1 Keywords used to refer to quality

Literature uses different keywords when referring to quality of recycling (Table 10). “Technical quality” or “technical characteristics” of recyclates (i.e., secondary materials obtained as an output of recycling) are intuitively key factors relevant to quality, but this would require testing standards and materials databases as discussed in Demets et al. (2021). The terms “function” or “functionality”, in turn synonymous with “utility”, are rather well-known and used in the field of functional recycling of metals (UNEP, 2011). Functional recycling has clear connections with high-quality recycling, as opposed to non-functional recycling, often associated to downcycling, i.e., low-quality recycling. In the LCA field, the concept of quality is often related to the effective “substitutability” of primary (virgin) material by the recyclate (Civancik-Uslu et al., 2021; Demets et al., 2021; Rigamonti et al., 2020; Rigamonti et al., 2009; Vadenbo et al., 2017), which has clear relations with quality. Other terms used in relation to quality of recycling include the “circularity potential”, “recyclability potential” or “suitability in circular economy” (CE), which reflect the ability of a recycling system to close material loops. “Open/closed-loop recycling” are expressions that have been used since long, sometimes with direct implications related to quality of recycling, however with controversial consequences. Closed-loop is in fact typically considered synonym of high-quality (notably, Haupt et al., 2017), whereas connections between open-loop and levels of quality can be less straightforward (Huysmans et al., 2017). Table 10 summarizes keywords and definition found in the literature (elaborated after Tonini et al., 2022).

Table 10. Keywords and terms used with reference to quality of recycling (after Tonini et al., 2022).

Keywords and terms ¹	Definition (implicit/explicit) ²	Used by ³
Impurity content	Content of untargeted materials and/or substances in a targeted waste stream destined to recycling/reprocessing (material-specific concept). This is often used in scientific literature to refer to quality of recycling.	Alassali et al. (2020); Eriksen, Pivnenko, et al. (2018); Faraca, Boldrin, et al. (2019); Muchova et al. (2011); Muchová & Eder (2010); Pivnenko et al. (2014);

		Rodriguez Vietez et al. (2011)
Technical quality	Example for plastic: the technical quality of plastics is a result of mostly mechanical properties, typically complemented with a property that describes the flow behaviour of the melt phase (Demets et al., 2021; material-specific concept).	Demets et al. (2021)
Technical characteristics (properties)	The technical properties that give the material the ability to fulfil the required functions. For example, for plastics the properties are generally divided into mechanical and processability characteristics (Demets et al. 2021; material-specific concept). Rigamonti et al. (2020) look at quality from the perspective of the technical substitutability of secondary materials relative to primary, for use in waste LCA studies, using a set of case studies. Sixteen technical substitutability coefficients are provided, for individual waste fractions ranging from paper and HDPE/PP/mixed plastic waste to recycled aggregates. The approach differs from that of Demets et al. (2021) in that only one main technical property is considered.	Many authors; e.g.,: Demets et al. (2021); Eriksen & Astrup (2019); Grant et al. (2020); Rigamonti et al. (2020)
Function/Functionality	A defined bunch of physical and chemical properties that made the material desirable in the first place (material-specific concept).	Many authors; e.g.,: Eriksen, Damgaard, et al., (2018); Eriksen & Astrup (2019); Hahladakis & Iacovidou (2019); Stewart & Weidema (2005); Talens Peiró et al. (2018); UNEP (2011); Vadenbo et al. (2017)
Functional recycling ⁴	Recycling in which the element in a discarded product is separated and sorted to obtain secondary material displacing same primary material. Non-functional recycling refers to recycling in which the element in a discarded product is collected and incorporated in an associated large magnitude material stream (e.g., copper incorporated in a flow of stainless steel). This represents the loss of its function as it is generally impossible to recover it from the large magnitude stream	Many authors; e.g.,: Diener & Tillman (2015); Eriksen, Damgaard, et al. (2018); Graedel et al. (2011); Guinée et al. (1999); Hahladakis & Iacovidou

	(system-wide concept; BIO by Deloitte (2015)	(2019); Reck & Graedel (2012); Stewart & Weidema (2005); Talens Peiró et al. (2018); UNEP (2011); Vadenbo et al. (2017)
Quality	<p>The extent to which, through the recycling chain, the distinct characteristics of the material (the polymer, or the glass, or the paper fibre) are preserved or recovered so as to maximise their potential to be re-used in the circular economy (Grant et al., 2020). These characteristics vary by material but may include, for example, food-contact suitability, structural characteristics (i.e., uniformity and viscosity), clarity and colour form, and odour. The same authors link their definition to the practical utility of the material in the circular economy, and on identifiable characteristics of materials within the recycling chain. Such a definition seems promising towards a quantitative approach to assessing the quality of recycling, though making it operational could be challenging. In any case, the authors also provide concrete examples of quality category for packaging waste, notably plastics (PET, PP, etc.), glass, and paper/cardboard.</p> <p>The ability of a secondary material to fulfil the functionality of the raw materials substituted (Vadenbo et al., 2017).</p> <p>The remaining functionality (i.e., described via the remaining properties and characteristics) of material, components, and products once they become secondary materials (Hahladakis & Iacovidou, 2019).</p>	Grant et al. (2020); Hahladakis & Iacovidou (2019); Vadenbo et al. (2017)
Resource dissipation/Dissipative flows	<p>Dissipative flows of abiotic resources are flows to sinks or stocks that are not accessible to future users due to different constraints. These constraints prevent humans to make use of the function(s) that the resources could have in the technosphere (system-wide concept; Beylot et al., 2020). Roithner & Rechberger (2020) propose a calculation of the recycling rate based on exergy and show how current EU recycling rates do not capture the quality of the recycled material and/or the functionality of the recycling process (i.e., the quantity of material that is "actually" useful for further</p>	Berger et al. (2020); Beylot et al. (2020); Ciacci et al. (2015); Passarini et al. (2018); Stewart & Weidema (2005); Torres de Matos et al. (2020)

	use in the economy to displace virgin production).	
Substitutability	The degree of functional equivalence between alternative resources/products for a specific end-use (Vadenbo et al., 2017). Also called substitution ratio or displacement/substitution factor (material-specific concept). For example for plastics: a measure of the functionality of the recycled plastic divided by the functionality of the substituted virgin plastic (Vadenbo et al., 2017).	Many authors; e.g.,: Civancik-Uslu et al. (2021); Rigamonti et al. (2020); Rigamonti et al. (2009); Vadenbo et al. (2017)
Circularity potential	The ability of individual recycled fractions to fulfil quality demands in a steady-state market representing a closed material loop situation (Eriksen, Damgaard, et al., 2018; system-wide concept). The circularity potential indicator (a number between 0 and 1, where 1 is the highest quality) is proposed, which measures the ability of the material to substitute high-quality virgin material, for which the circularity potential is set to 1. For recycled products, the circularity potential would equal the potential substitution of primary material ("true recycling rate") multiplied by the potential market share of the application group. For example, food-grade plastic is given a potential market share of 1 (i.e., maximum) because it can in principle be used for all (high-, medium-, and low-quality) market applications.	Eriksen, Damgaard, et al. (2018)
Downcycling vs Upcycling	Recycling process whereby the recycled material is used for a lower-quality market application than that of the previous life cycle, normally defined by a lower market value, as opposite to upcycling (system-wide concept), defined for plastics as: 'the use of plastic waste, postindustrial or postconsumer, as a feedstock for the synthesis of value-added products, being polymers, molecules, or materials' (Jehanno et al., 2022).	Eriksen, Damgaard, et al. (2018); Haupt et al. (2017); Jehanno et al. (2022); Koffler & Florin (2013); Rigamonti et al. (2020)
Closed-loop vs Open-Loop Recycling	Closed loop is a recycling process whereby the recycled material is reused for the same market application as that of its previous life cycle (system-wide concept). Open loop is a recycling process whereby the recycled material is used for a different market application than that of the previous life cycle (system-wide concept).	Many authors; e.g.,: Andreasi Bassi, Tonini, et al. (2021); Geyer et al. (2015); Graedel et al. (2011); Haupt et al.

¹ Keywords were used to retrieve studies from Scopus, Scencedirect and Google search. The studies were then further screened and only those studies containing a theoretical/operational approach to quality of recycling and/or a definition of quality of recycling were retained.² Whenever the "Definition" is taken from a specific study, the associated reference is reported. Alternatively, a reasonable definition based on the literature is given. ³ The list of sources is exhaustive. ⁴ Term coined by Guinée et al. (1999).

7.2 Possible avenues to define quality of recycling

Figure 16a shows how the different concepts of quality, as used in the studies identified in the literature, can be integrated to draft a framework for quality. Impurities are part of the broader group of technical characteristics, and a bunch of these is typically essential to fulfil the desired functions of the intended material application. We believe that preserving functionality is key to distinguish between levels of quality recycling. In this respect, a recyclate may have an adequate quality for an intended market application and be not adequate (thus not functional) for other applications (see Figure 16b). If the sum of the (functional) substitutions is high, the recycling system tends to circularity.

The extent to which the functionality of the secondary material is preserved can be described in different ways, notably the substitutability of primary material or the suitability in the circular economy. For example, Grant et al. (2020) propose quality grades (A, B, C, etc.) for recycled material, as widely used in some of the standards for material recycling (e.g., for paper EN643). The grade reflects the remaining functionality of the material after recycling. Knowing the origin of the waste, thus the application in the previous life cycle, a loss of grade (thus of functionality of the material) could be traced if the intended application of the recyclate is known. Demets et al. (2021) propose to describe functionality for plastics via the technical substitutability, which represents the extent to which the recycled material is able to meet the functions required for the intended application, relative to the virgin counterpart. Low values for this factor suggest that the recycling is not functional to the intended application, and that the distinct characteristics of the material are thus not preserved in the intended application (loss of functionality). Eriksen, Damgaard, et al. (2018) suggest instead a factor that measures the ability of the material to substitute high-quality virgin material, represented by the market share of the potential application groups. For example, for the case of plastics, food-grade recycled plastic is given a potential market share of 1 (i.e., maximum quality), because according to the authors can be used for all (high-, medium-, and low-quality) market applications. While closed/open loop considerations seem not important once the focus is on preserving functionality, one may still argue that closed-loop recycling often coincides with preserving functionality (thus quality), as *de facto* shown in the results of Demets et al. (2021) for PE, and Eriksen, Damgaard, et al. (2018) for food-contact packaging plastic.

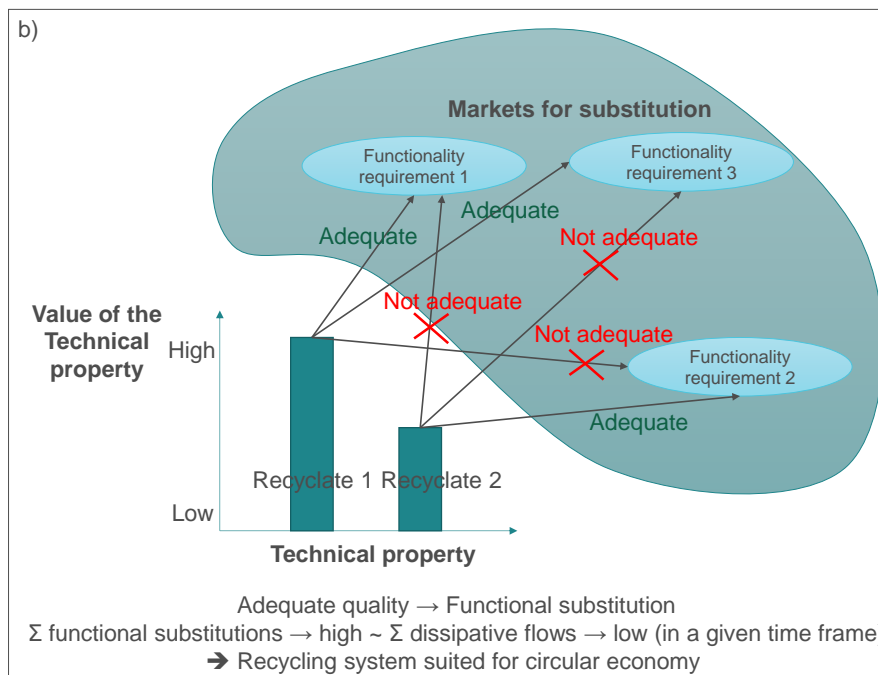
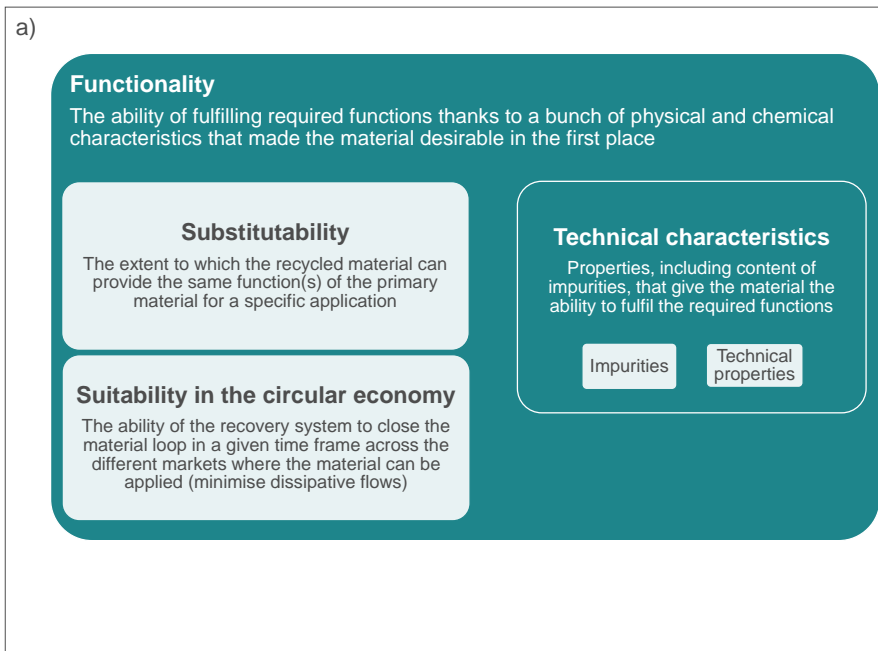


Figure 16. Possible approaches to define quality of recycling and their relationships (a) and illustration of the concept of functional recycling for substitution of primary material (b) (taken from Tonini et al. (2022)).

7.2.1 Environmental dimension

The approaches touched above do not tackle specifically the environmental impacts associated with the recycling chain. Indeed, there is the possibility that a process producing a high-quality recyclate is associated with higher environmental impacts relative to one with a lower quality output. This may be the result of energy/resource consumption for processing, but also simply material losses. Overall, it appears that the definition of quality could be complemented with an additional criterion on the environmental performance, while referring generically to avoiding adverse effects (as in the End-of-Waste criteria as in the Waste Framework Directive) seems leaving too much latitude for interpretation. This should however be carefully operationalised, as different approaches lead to different

conclusions. An example is the following: a recyclate obtained from a chemical recycling process may result to be more impacting than the virgin counterpart when applying a product-oriented LCA approach (e.g., the product environmental footprint, PEF), which neglects the alternative (or otherwise occurring) fate of the waste (Andreas Bassi, Tonini, et al., 2021; Mengarelli et al., 2017; Tonini et al., 2021). However, the same chemical recycling can still be superior to the waste management alternative otherwise occurring for that plastic waste (e.g., incineration), when applying a waste management-oriented LCA approach, which focuses on the waste material valorisation. The application of one approach or the other one would thus lead to very different conclusions on the environmental performance of chemical recycling. Another issue is that carrying out an LCA of a product generates multiple insights into its environmental performance resulting in a complete but not definitive outcome. Following the same example, the recyclate obtained from a chemical recycling process may result to be more impacting in terms of global warming but less impacting in terms of water use than the virgin counterpart. It would lead to a difficult and potential misleading interpretation of the environmental performance of quality. To define the quality of materials in a circular economy, Steinmann et al. (2019) introduced an indicator based on the energy use of recycled products versus their counterparts produced from primary material inputs only. A series of cradle-to-gate life cycle energy are used to develop the indicator. The authors implied that being energy demand a crucial aspect of circularity and associated quality of recycling, the primary energy demand of a product may be associated with the decrease of its overall environmental impacts. However, an energy-based indicator might not be considered fully informative as argued in Iacovidou et al. (2017). The same authors argue that in principle multidimensional indicators, including economic and social dimensions, should be part of the assessment. All in all, the discussion about linking quality with environmental impact and broader sustainability concepts is still far from over and further research is needed.

7.3 Quality of recycling: existing standards used by the industry

This section aims to describe the main industrial standards in use in the recycling industry (bio-waste, glass waste, metal waste, paper waste, plastic waste, and wood waste). Most of the information for plastic, paper and glass waste is taken from previous work carried out by the JRC, notably Grant et al. (2020) and Muchová & Eder (2010). A summary is presented in Table 11.

Table 11. Overview of the most relevant standards and guidelines in use at industry level for sorting/recycling of selected waste materials.

Waste material	Standards/Guidelines in use for sorted material	Standards/Guidelines in use for recycled material	Reference documents
Bio-waste & (compost digestate)	ECN-QAS 2018, national quality assurance guidelines, national bio-waste/compost/fertiliser. EU Fertilising Products Regulation	ECN-QAS 2018, national quality assurance guidelines, national bio-waste/compost/fertiliser. EU Fertilising Products Regulation	European Commission (2019f); European Compost Network, (2018); Saveyn & Eder (2014)
Glass waste	CEN guidelines; BSI specifications; WRAP protocol;	-	BSI/WRAP (2003); CEN (2008); WRAP (2008)

	GTS specifications		
Metal waste (Ferrous)	European steel scrap specification; ISRI specifications	-	EFR (2007); ISRI (2021); Muchová & Eder (2010)
Metal waste (Aluminium)	ISRI specifications; EN13920	-	CEN (2003); ISRI (2021)
Paper waste	EN643 specifications	-	CEN (2014)
Plastic waste	PRE guidelines	EN15342; EN15345; EN15348*	EN15344; EN15346; Plastics Recyclers Europe (2018); EN15342; EN15344; EN15345; EN15346; EN15348
Woodwaste	PAS 111-2012; Bundesgesetzblatt Jahrgang 2002 (German wood waste ordinance)	-	BSI (2012); Bundesgesetzblatt Jahrgang (2012)

*The standards only focus on how to characterise the material (do not distinguish quality grades).

7.3.1 Bio-waste recycling

While some small differences exist, the standards/guidelines used in the different Member States are generally aligned to the European Compost Network (2018) quality assurance scheme and to the standard laid down in the EU Fertilising Products Regulation (EU 2019/1009) (European Commission, 2019d). The European Compost Network has set up a European quality assurance scheme for compost and digestate, ECN-QAS (European Compost Network, 2018), including positive list of suitable input materials, process requirements for composting and anaerobic digestion and product criteria for compost and digestate. The EU Fertilising Products Regulation (European Commission, 2019d) lays down criteria for compost and digestate for the purpose of being marketed, traded and used in the common EU market. The criteria usually applied for classification are input materials used, process parameters, product quality, and final uses of the compost/digestate as summarised in previous work from the JRC (Saveyn & Eder, 2014).

Industry current practices

Lists of waste sources that can undergo composting/digestion are included in the ECN-QAS (European Compost Network, 2018), in the majority of national quality assurance guidelines, national bio-waste/compost and in the EU Fertilising Products Regulation (European Commission, 2019d) as well. A summary of the standards and guidelines used in the MS may also be found in Saveyn & Eder (2014).

Take away and further research needed

While some small differences exist, the standards/guidelines used in the different Member States are generally aligned to the ECN-QAS quality assurance scheme (European Compost Network, 2018) and to the standard laid down in the EU Fertilising Products Regulation (European Commission, 2019d). The European Compost Network has set up a European quality assurance scheme for compost and digestate. It should be kept in mind that while the quality of compost and digestate is often correlated to the type of sourcing (type of

compostable material sourced and collection schemes in place), the market value may not necessarily follow such logic and be even negative depending upon the very local conditions (e.g., in areas with excess of nutrients such as the Netherlands, the demand for compost/digestate is low and prices are between 2 and -5 euro per tonne; Huygens et al., 2019; Tonini et al.; 2019)

Glass waste recycling

Technical specifications and standards are widely used in the glass industry, typically referring to one or more of the following properties:

- Physico-chemical composition.
- Content of impurities.
- Physical size and shape.
- Homogeneity, i.e., the variation within the given specification.

Industry reference standards for sorting plant outputs

Various specifications have been produced by industry groups around Europe, notably the CEN guidelines (packaging waste glass; CEN/TR 13688:2008; CEN, 2008), the BSI PAS 101 (BSI PAS 101 - Specification for collected container glass cullet, BSI/WRAP, 2003) the BSI PAS 102 (specifications for processed glass for selected secondary end markets, BSI, 2004), and the WRAP protocol for flat glass (Quality Protocol for Flat Glass, WRAP, 2008) developed on behalf of the UK Environmental Agency.

In a nutshell, CEN 13688 guidelines include a set of contamination limits for packaging waste glass to be consistent with the requirements of the European Packaging and Packaging Waste Directive 94/62/EC. BSI PAS 101 similarly includes a set of contamination limits for glass waste, but at the same time classifies glass waste according to four quality grades (A, B, C, and D). BSI PAS 102 (specifications for processed glass for selected secondary end markets) only provides test methods for the determination of particle size distribution, etc. The WRAP Quality Protocol for Flat Glass sets out criteria for when the material is no longer classified as waste and is suitable as input material in the manufacture of new flat glass products. In general, impurity quality standards for cullet to be used in the manufacture of flat glass are around 20 times stricter than those for the manufacture of container glass. Notice that these standards are reviewed in detail in the technical proposals for the EoW criteria for glass cullet (Rodriguez Vietez et al., 2011).

Notice also that in the EoW criteria proposed by Rodriguez Vietez et al. (2011), FERVER (EU Federation of glass recyclers) contributed to the EoW formulation process by suggesting a set of technical specifications to characterize cullet of sufficient quality that could eventually be declared as end-of-waste. These were then elaborated in the final EoW criteria proposed by Rodriguez Vietez et al. (2011), which specify the following limits on non-glass components (based on a review of the abovementioned industry standards and specific to re-melt applications): Ferrous metals: 50 ppm; Non-ferrous metals: 60 ppm; Non-metal non-glass inorganics: 100 ppm for cullet size > 1mm; 1500 ppm for cullet size ≤ 1 mm; Organics: 2000 ppm.

Industry current practices

The technical proposals for EoW Criteria for glass summarises the situation as follows: *"There are a number of technical specifications developed by industrial or recyclers organizations (FERVER, BSI/WRAP), or independent consultant groups, and which are applied in certain Member States and in individual market transactions on a case-by-case basis. Additionally, Member States in some cases have developed technical standards for glass cullet. Feedback from the TWG pointed out that these standards may vary significantly from country to country. These national standards are usually strictly linked to the quality of the collected cullet, to the technical structures of local glass industries and to the national commercial situation"* (Rodriguez Vietez et al., 2011).

Take away and further research needed

A number of standards are applied in the Member State and in individual market transactions on a case-by-case basis. The standards are very different from country to country, and in some cases some Member States have developed their own. The latter are often specifically related to the quality of the locally collected glass waste and the industry available for reprocessing it.

7.3.2 Metal waste recycling

The metal industry has a long and established tradition with defining standards and categories of scraps and residues. For ferrous metals, specifications and standards exist at international, European, national levels as well as between individual parties. It is clear that for the reason of marketing and trading, standards and specifications are needed not only to set the price but also to be used as reference for classification and quality control. In many cases based on the production need, iron and steel scrap are processed according to the bilateral specifications agreed upon between the scrap processor and smelters. Scrap metal is basically classified according to several properties, most notably:

- Chemical composition of metals.
- Level of impurity elements.
- Physical size and shape.
- Homogeneity, i.e., the variation within the given specification.

Industry reference standards for sorting plant outputs and current practices

For ferrous metal waste, the main standards in use are the European steel scrap specification (EFR, 2007) and the ISRI specification (ISRI, 2021). As illustrated in Muchová & Eder (2010) national standards also exist (UK, Spain, Belgium, France, and Germany), as well as bilateral agreements between smelters (end users) and scrap recovery plants. For aluminium metal waste, the main standards in use are the European standard EN13920 on aluminium and aluminium alloy scrap and the ISRI specifications (ISRI, 2021). Likewise ferrous scrap, national standards also exist as well as bilateral agreements or contracts between scrap processors and end users (smelters).

Take away and further research needed

The metal industry has a long and established tradition with defining standards and categories of scraps and residues. The existing EU and international standards represent an appropriate basis to derive quality categories for metal scraps.

7.3.3 Paper waste recycling

The defining of recycling quality for paper and cardboard has, to a great extent, already been carried out by the paper processing industry, and is embodied in the specifications included in the standard EN643 (CEN, 2014).

EN643 is this European list of standard grades of paper and board for recycling, last updated in 2013. EN643 defines the grades of paper for recycling and quality requirements (including setting limits on tolerance levels of non-paper components). There are a wide range of grades within EN643 (see Grant et al., 2020), providing much more variety than simply distinguishing low/high-grade paper, cardboard, newspaper and magazines, etc. The types of paper/board can, very broadly, be characterised as:

- Mixed papers (waste and scrap paper and cardboard).
- Newspapers and magazines (paper or paperboard mainly manufactured from mechanical pulping processes and with printed material).
- High grades (mostly manufactured from bleached mechanical pulping) .
- Corrugated and kraft (unbleached paper/board).

EN643 also distinguishes grades based on whether the paper waste has undergone separate collection and specifically exclude paper that has been extracted from mixed MSW. The EN643 standards secure 'comparable' requirements for paper for recycling across Europe, assisting the trading of paper across EU.

Industry reference standards for sorting plant outputs

Benchmark standards for the quality of recycling of paper and board in relation to sorting plant inputs and outputs (which are then the inputs to the subsequent paper mills) are generally well defined and agreed upon within the European paper industry due to the development and adoption of the EN643 standard by the paper processing industry throughout Europe. However, some studies indicate that within the main EN643 grades, tolerances for undesired material are in practice deviated from depending on the requirements of individual paper mills. In practice, while the tolerances set within the EN643 grades are common reference points for the paper industry, they are in reality adapted to the specific paper mills requirements on the basis of mutual arrangements with sorting plant suppliers.

Industry current practices

The paper processing industry around EU widely makes use of the EN643 standard to define grades of quality for paper waste. As mentioned, flexibility exists around the tolerances set within EN643 depending upon the specific requirements of the local paper processing mills. This translates into specific arrangements between the final recyclers (the mills) and the sorting plant supplying the sorted paper waste to be used as input to the mills.

Take away and further research needed

All in all, the EN643 grades can form the basis of an operational assessment of high quality recycling for paper and board: outputs are higher quality recycling if they conform to, or are closely guided by, the EN643 grades which are likely to be remanufactured into paper/board products that can again be recycled into similar grades (de-inking and corrugated cardboard grades). By contrast, mixed paper grades are less likely to be recycled into similar grades, and some grades of mixed papers of lower fibre quality, fibre quality degraded through collection, storage and transport, and/or higher levels of non-paper material and other impurities, are more likely to end up as low-fibre-strength, single use material (tissues, etc.). A higher quality recycling chain is likely to maximise captures into deinking and corrugated cardboard grades, whilst fully utilising remaining mixed papers grades. If a plant is able to reduce the proportion of outputs going to non-circular paper recycling, and concurrently able to increase the proportion that adheres (either exactly, or pragmatically) to an EN643 grade which can readily be recycled again thereafter, that would indicate a tangible and easily understandable transition from lower to higher quality recycling. An initial proposal of quality categories based on material specifications in EN643 was done by Grant et al. (2020) and is presented in Table 12.

Recording data on quantities of bales sold into mills broadly corresponding to different EN643 grades (and on sorted quantities that do not meet any EN643 grade standard), could provide a sufficient and practical level of detail on which to base an assessment of quality of recycling for paper waste. As some EN643 grades are subject to further sorting (e.g., within sorting stages at paper mills), the measurement should ideally be taken at the point at which no further sorting is done and the sorted grade is input into the final paper waste recycling process.

EN643 grades primarily classify sorting plant outputs. In order to map EN643 grades to products, a clearer mapping is needed between some EN643 grades produced from household recycling streams and inputs to particular paper product manufacturing processes, in particular for some mixed papers outputs. The correspondence between EN643 grade and end use is instead clearer for higher grade EN643 products (de-inking and OCC grades).

Table 12. Possible quality categories for paper waste based on material specifications from EN643 as proposed in Grant et al. (2020).

Quality category	Quality dimension	Specifications (EN643)	Rationale
A	Maintain fibre characteristics, homogeneity of grade	De-inking grade (1.11) Old Corrugated Cardboard grade (1.05)	Suitable for recycling to the same grade of product Suitable for corrugated cardboard manufacture
B	Mixed fibre characteristics, some variation in grade	Mixed papers (1.02)	Suitable for manufacture of other grades of product (components of corrugated cardboard, tissues)
C	Mixed fibre characteristics, lower grade fibres	Not meeting a specified EN643 grade	May be suitable for products with less structural fibre requirements

7.3.4 Plastic waste recycling

Industry reference standards for recycling plant outputs

Standards for secondary raw materials referenced within EUCertPlast certification are EN standards for the characterisation of plastic secondary raw materials (see 'required characteristics' in the relevant EN Standard). These EN standards are:

- EN15342 for polystyrene secondary raw materials.
- EN15344 for polyethylene secondary raw materials.
- EN15345 for polypropylene secondary raw materials.
- EN15346 for poly(vinyl chloride) secondary raw materials.
- EN15348 for poly(ethylene terephthalate) secondary raw materials.

These EN standards do not distinguish different qualities of secondary raw materials. In practice, plastic recyclers create outputs to the specific quality requirements of end users, i.e., plastic converters.

Industry reference standards for sorting plant outputs

Plastic Recyclers Europe (PRE) has produced guidelines for the quality of the bales produced at sorting plants to favour a transformation of the sector towards increased circularity (Plastics Recyclers Europe, 2018). The guidelines focus on key prohibited impurities and on the threshold allowed for some of them (see summary in Table 13). The latter are to be set by the subsequent recyclers depending upon their requirements.

Industry current practices

For sorters, in practice, the quality of sorting plants outputs can diverge from the industry standards detailed earlier by PRE with regard to tolerance levels for material on the 'prohibited impurities' list. Offtakers for HDPE and PP outputs are reported by some sorting plants to tolerate higher levels of impurities than those set in PRO-proscribed standards, as highlighted in the findings of Grant et al. (2020). The quality aimed at by sorters of LDPE films has increased in the last years due to lower demand for LDPE and more competition for oftakers. For sorters operating outside of arrangements with PROs (for instance in Hungary), the purity levels of the sorted plastic waste are individually agreed with the oftakers and can thus vary within certain limits. However, since they compete for

the same offtakers as sorting plants sorting to PRO set standards, their outputs tend to be comparable to international standards (notably: American Plastics Recycling, ARA, and/or DSK/DSD specifications). An overview of the quality standards applied to sorted plastics in selected plants in EU28 may be found in Grant et al. (2020). For recyclers, the quality/characteristics of the recycling plant outputs is typically determined by the specific quality required by the end users, i.e., the plastic converters (including particularly where they utilise the output themselves in product manufacture).

Take away and further research needed

For sorting plant outputs, some guidelines on the quality of the bales have been proposed by PRE but are hardly utilised. For recycling plant outputs, the EN standards in use only focus on how to characterise the secondary raw materials but do not distinguish different recycled plastic qualities. A widely recognised definition of quality grades does not exist. In practice, both plastic waste sorters and recyclers create outputs to the specific quality requirements of recyclers and end users, respectively.

In addition, there is a lack of collated information available on specific quality requirements of major groups of HDPE and PP products (requiring different grades of HDPE and PP) across packaging and other applications. There is also a lack of information on the impact of different additives, which are used to enhance certain structural characteristics of the secondary raw materials to suit specific applications, on the onward recyclability of the polymer.

Table 13. Overview of the PRE guidelines for the quality of the bales produced at sorting plants, taken from Grant et al. (2020).

Bales	Prohibited Impurities	Limited Impurities	Grade variation
All:	Minerals, Rubber, Wood, Sacks, Hazardous Waste, Medical Waste, Glass, Oxo or degradable material, Food, Silicones		
PET Bottle grades	PET-G (PET with added glycol for flexibility) CPET (crystalline PET suitable for ovens)	Max 5% of PET from non-food consumer applications Metals Paper/Cardboard PVC Transparent Colours Opaque Colours Monolayer trays Other plastics	Clear: Max 5% light blue PET, no opaques Clear Blue: Max 20% of blue PET, no opaques Light Blue: >20% light blue PET, no opaques Coloured >80% transparent mixed colours, max 5% opaque colours
HDPE Bottles, Mixed Colour	Foams Polyurethane (PUR)	Max 5% of HDPE from non-food consumer applications Metals	n/a

		Paper/Cardboard PP Other plastics	
PP Films	Expanded Polystyrene (EPS) & PUR	Metals Paper/Cardboard PVC, LDPE, HDPE, LLDPE Other plastics Other impurities	Variations in minimum content for: PP
PE Films	EPS & PUR	Metals Paper/Cardboard PVC PP Other plastics Other impurities	Variations in minimum content for: LDPE LLDPE HDPE

7.3.5 Wood waste recycling

Wood waste is typically separately collected at the so-called municipal recycling centres (or municipal collection points) where it is sorted in different containers according to the quality. No harmonised definitions of wood waste as collected at recycling centres exist (Vis et al., 2016) and individual countries and sectors implement their own standards as also stressed in BSI (2012). However, two standards are quite often referred to in the wood industry, i.e., the British (BSI, 2012) and the German (Bundesgesetzblatt Jahrgang, 2012) one.

Industry reference standards for recycling plants outputs

The PAS 111-2012 (BSI, 2012) provides a specification for individuals and organizations¹⁶ recovering and processing post-industrial and post-consumer wood waste into wood products so that customers are assured about the verified and consistent quality of the material. In annex A of the same, a classification of wood waste, as collected from consumers and industry (e.g., at recycling centres) into four quality grades is provided: A is assigned to clean recycled wood, B to industrial feedstock, C to wood fuel, D to hazardous waste (impregnated wood to be incinerated). The grades are also associated with a specific collection scheme or source. For example grade A is associated with wood collected at retailers and secondary manufacture, and consists of packaging wood, solid softwood and hardwood, scrap pellets, packing cases, cable drums and offcuts from manufacture of untreated wood products. The categorisation also provides a list of the typical non-wood materials present in the wood waste collected prior to reprocessing.

Similarly, the German wood ordinance (Bundesgesetzblatt Jahrgang, 2012) provides a classification of wood waste received by reprocessors. Wood waste is similarly classified in four groups: AI includes wood waste left in its natural state or merely mechanically processed, which, when used, is not more than insignificantly contaminated with foreign matter. Type AII includes wood waste that is glued, painted, varnished or otherwise treated waste wood without halogen-organic compounds in the coating (e.g., PVC) and without wood preservatives. Type AIII

¹⁶ In PAS 111-2012 they are generically called 'reprocessors', i.e., organizations that recover wood from the waste stream and convert it into a form suitable for use in a new product or other application.

wood waste with organ halogen compounds in the coating and without wood preservatives and type AVI wood waste treated with wood preservatives, such as railway sleepers, pylons, hop poles, and other waste wood which cannot be assigned to waste wood categories AI, AII or AIII due to its pollutant load, with the exception of PCB waste wood. A special case is the PCB (polychlorinated biphenyls) wood waste that is subject to the PCB/PCT Regulation, mainly coming from construction and demolition wood waste. However, the use of such chemicals in construction was banned worldwide by the Stockholm Convention in 2001.

Industry current practices

No harmonised definitions of wood waste exist (Vis et al., 2016), and individual countries and sectors implement their own standards as stressed in BSI (2012). However, two standards are quite often referred to in the industry, i.e., the British (BSI, 2012) and the German (Bundesgesetzblatt Jahrgang, 2012).

Take away and further research needed

A harmonised classification of wood waste, and the implementation of quality criteria across all sectors, may be desirable to ensure clean material flows for wood recycling and at the same time to offer a basis for recycling facilities to reject unwanted wood fractions prior to reprocessing.

8 Quality of recycling: definition and framework proposal

8.1 The proposed framework for quality of recycling

The starting point of the framework is the definition of Grant et al. (2020) and the EoW criteria from the European Union Waste Framework Directive (European Commission, 2019a). The latter stipulates that a given waste ceases to be waste when it has undergone a recovery operation (including recycling) and complies with specified criteria related to the existence of a market or demand (which are often also technical criteria), when it fulfils the applicable sector legislation and standards, and the environmental impact of the recovery operation does not lead to overall adverse impacts.

A schematic overview of the proposed quality framework for recycling is shown in Figure 17, showing the three main 'dimensions' that are included in the framework, namely the Total Substitution Potential (TSP; expressed in %), the Long-Term in-Use Occupation (LTUO; expressed in tonne*years), and the Environmental Impact (EI; which has typical LCA-based units). Each of these three dimensions provides additional information relevant to the quality of recycling. In general, the higher the TSP and LTUO and the lower the EI of a certain recycling pathway, the higher the quality of recycling.

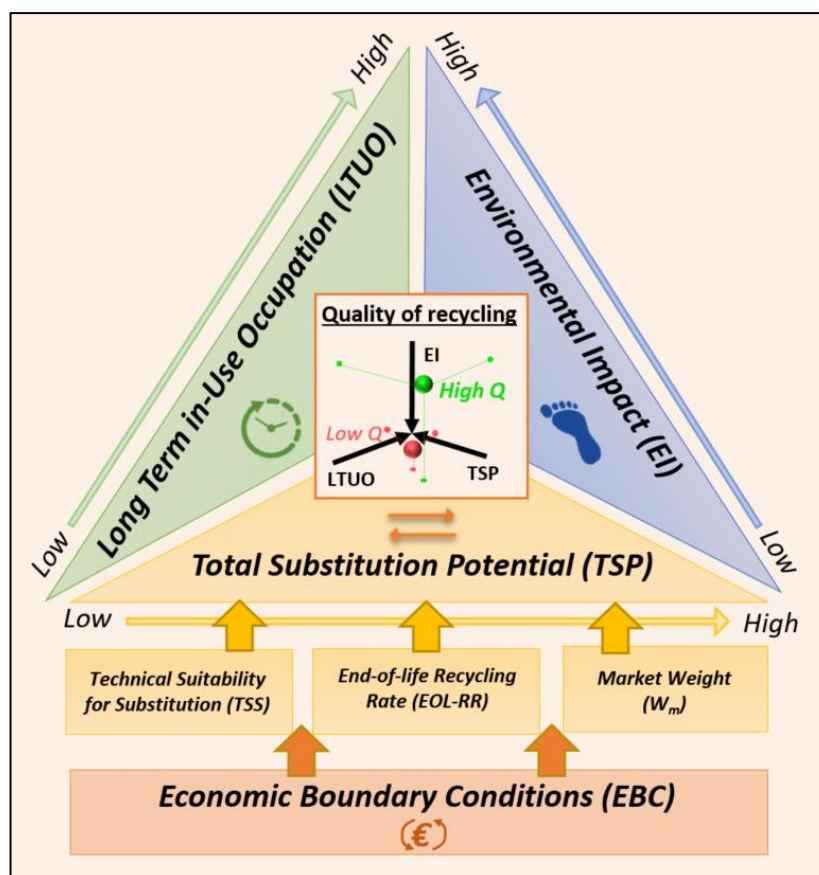


Figure 17: Schematic representation of the proposed quality framework for recycling. Based on the three dimensions, namely the Long Term in-Use Occupation (LTUO), Environmental Impact (EI), and Total Substitution Potential (TSP), the quality of recycling can quantitatively be evaluated.

Below, each dimension will be introduced in a general way. In sections 8.2-8.4, a more detailed explanation is given, together with examples on how to calculate the dimensions and indicators.

The first dimension of the framework, the TSP, indicates to which extent a secondary material can provide the same function(s) as the primary material. This indicator builds on the work of Vadenbo et al. (2017) who proposed a similar indicator, i.e., the substitution

potential (γ), which is based on four main determining factors, namely the technical quality of secondary materials ($\alpha^{rec/disp}$), the overall recycling efficiency (η^{rec}), the amount of material in a waste stream that can be used as secondary material (U^{rec}), and the impact on the market in which the secondary materials are used (π^{disp}), as can be seen in Equation 32.

$$\gamma = U^{rec} \cdot \eta^{rec} \cdot \alpha^{rec/disp} \cdot \pi^{disp}$$

Equation 32

In the current framework, the impact on the market in which the secondary materials are used is omitted for simplicity reasons. Hence, our proposed TSP depends on different factors (or in this work referred to as 'indicators'), being the Technical Suitability for Substitution (TSS; which corresponds to $\alpha^{rec/disp}$ in the equation of Vadenbo et al., 2017), but which we have further elaborated in a way that it can be more profoundly calculated), the End-Of-Life Recycling Rate (EOL-RR), the Market Weight (Wm ; which is based on the market share of a given application in a certain market), and the Economic Boundary Conditions (EBC). The TSS of a secondary material (i.e., the first indicator) specifies whether or not the secondary material meets the (technical) quality requirements for substitution of primary materials. The TSS is a technical quality indicator and specifies for which applications a substitution is possible. If the recycled material is of low quality, the multiple end-use markets in which it can be used are restricted, which in turn would inevitably result in market saturation at a certain point in time. Yet, the TSS does not implicitly indicate 'how much' substitution is possible. The larger the market that is not covered by a secondary material, the worse. For this purpose, the Market Weight (Wm) indicator is incorporated. Furthermore, the EOL-RR is included, which is a measure for the resource dissipation that occurs during material recycling processes (e.g., collection, sorting, pre-treatments, and processing) (Graedel & Reck, 2014).

Often, the technical quality and the EOL-RR behave like communicating vessels. If the technical quality (in the current framework defined as TSS) is high (and thus in general more processing steps are needed to achieve this quality), then the EOL-RR might be negatively affected. These two indicators are conceptually similar to U^{rec} and η^{rec} , but we apply a slightly different strategy of including them in the equation, and can make a more direct link to recycling rate definitions, such as those proposed by the European Commission (Talens Peiró et al., 2018), which in turn builds upon the status report of the United Nations Environment Programme (UNEP) on recycling rates of metals (UNEP, 2011)

The EBC implicitly incorporates an economic dimension into the TSP. Grant et al. (2020) indicated that the suitability of an input for the production of quality secondary raw materials is dependent on the plant's economic balance, as well as the material's characteristics. Measures proposed to increase quality may indeed impact processing costs, revenues for outputs and costs for disposal that occur at a recycling plant. This in turn affects the industrial feasibility of the recycling pathway (Grant et al., 2020).

The second dimension of the framework is the LTUO, which indicates how much of a certain material is still 'functional' in society over a certain time horizon, as the TSP ignores the long-term aspects of circular economy and resource dissipation. As a society, we have to keep our resources in the technosphere as long as possible, minimizing dissipation to a level in which the materials are irreversibly lost (although, of course, this irreversibility includes an economic dimension as well and is thus difficult to absolutely quantify – in theory one could even recover metals from diluted sea water) (Beylot et al., 2020).

The third dimension of the framework is the EI and is determined by environmental impact calculations. Circular economy should focus on maximizing material recovery and minimizing the environmental impact associated with achieving circularity. However, in different contexts, improvements in the technical quality of secondary materials are likely to incur additional processes, and may not necessarily lead to significant environmental benefits (Lonca et al., 2020). Hence, for some materials, the environmental performance

of recycling is in some cases unclear or contested, which is conflicting the global goal of mitigating impacts on the environment (van Ewijk et al., 2021)

In the next sections, each of the three 'quality of recycling dimensions' is further elaborated and discussed in more detail, including clarifications regarding the objectives, definitions and equations of the different dimensions and indicators.

8.2 Total Substitution Potential (TSP)

The first dimension of the proposed quality framework is the TSP. The TSP combines the TSS, the Wm , the EOL-RR, and the EBC indicators by means of following Equation 33.

$$TSP(i,j) = \sum (TSS_{i,j} \cdot Wm_{i,j} \cdot EOLRR_{i,j} \cdot EBC_{i,j})$$

Equation 33

With $TSP(i,j)$ is the sum of the substitution potentials (SPs) of a secondary material j (e.g. regranulates coming from PET bottles), calculated by multiplying the TSS, Wm , EOL-RR, and the EBC, to substitute primary materials for various applications i (e.g., bottles, trays, fibers).

8.2.1 Technical Suitability for Substitution (TSS)

The TSS reflects the extent to which the technical properties of a secondary material j are suited for substitution of primary materials in a given application i (Zink et al., 2016), which is based on a set of properties. We link this indicator to the equation proposed by Vadenbo et al. (2017), i.e., the ratio of the end-use specific functionality of a secondary material ϕ^{sec} over the functionality of the potentially displaced alternative products ϕ^{disp} (see Equation 34).

$$TSS = \frac{\phi^{sec}}{\phi^{disp}}$$

Equation 34

Yet, our frame is really at (sub)application level, for example beverage bottles, piping for the building sector, etc. The proposed TSS indicates solely whether the secondary resources have 'adequate' technical characteristics for being used in those applications. A schematic overview of the technical characteristics to determine the TSS can be found in Figure 18. A TSS of zero indicates that the technical properties do not allow any substitution of virgin material in that application and a TSS of one indicates that virgin material can be fully substituted by a secondary material in that application. If blending the secondary material with virgin material would yield a blend of sufficient quality, a score between 0 and 1 could be given, based on the ratio of the amount of a secondary material that is used to the total amount of material that is needed to manufacture a given application in a certain market (so the sum of primary and secondary materials).

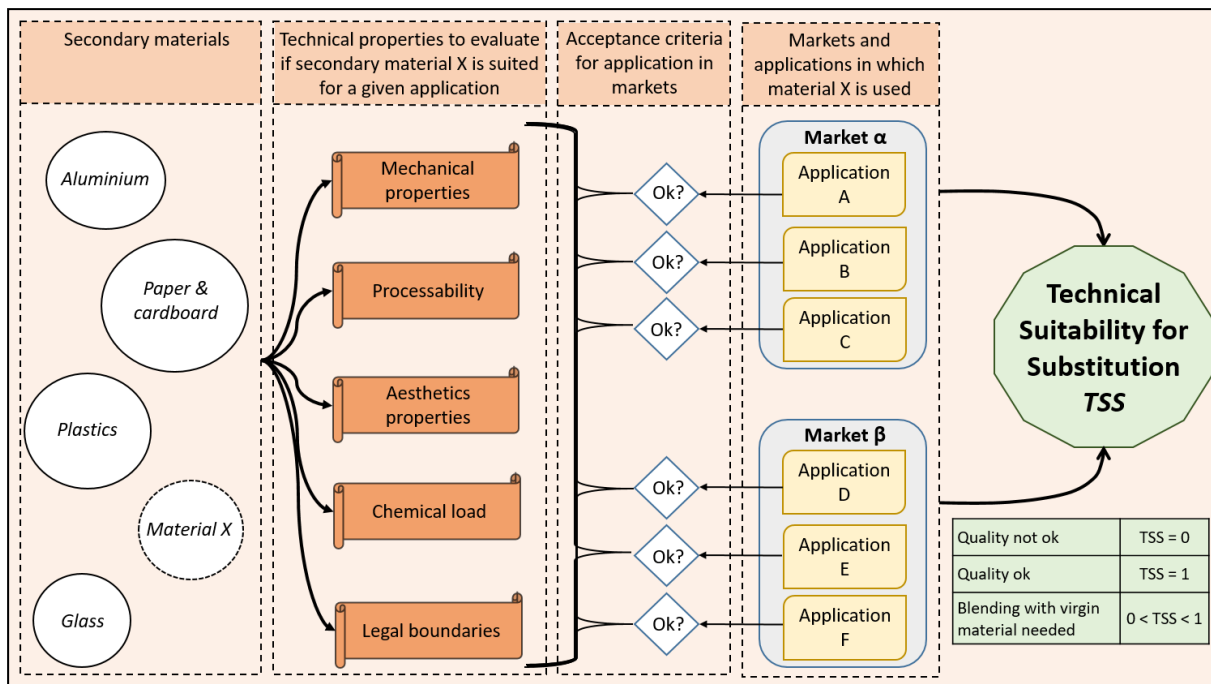


Figure 18: Theoretical framework for the determination of the technical suitability for substitution (TSS).

Important is that the TSS is assessed at a sufficiently granular level. Therefore, our proposal goes beyond the work of, for example, the 'market price-based substitution method', as proposed by Schrijvers et al. (2016) and the more simplistic ratio-based approach by Vadenbo et al. (2017). In this context, we follow the approach that is applied by industry, meaning that technical properties determine whether a resource is suited to feed in a certain production chain or not (Demets et al., 2021). This is the way that reprocessors assess the quality of the sourced material, deciding if they will use the material in a truck arriving at their plant. As shown in Figure 18, it is incomplete to arbitrarily indicate that a given secondary material X can be used in market α (e.g., automotive) but not in market β (e.g., packaging). A more in-depth analysis at application level allows to describe the TSP with more detail. It is needed to indicate, for instance, that the secondary material X can be used in application A (e.g., flower pots), but not in application B (e.g., cleaning products) of market β . The determination of the TSS should be considered as a first step to measure the TSP (and thus also the quality of recycling). Logically, the higher the TSS, the higher the quality of recycling could ultimately be.

There is a wide range of properties that can be assessed and the key properties are to some extent material dependent. However, with regard to the quality framework presented in this work, five general aspects are distinguished in the assessment of the technical material quality that should always be evaluated for a given secondary material, as listed below.

Mechanical properties

A given material must have the necessary mechanical properties to meet the specifications of the final application. Mechanical properties is a general term which covers a vast number of material characteristics such as stiffness, ductility, toughness, strength and hardness. The assessment of these characteristics of a material determine its application range and establish the service life that can be expected (Demets et al., 2021).

Processability

The processability signifies the ease of processing of any material, starting from raw or secondary materials to the final product. It is rather a broad aspect of the technical quality that includes, for instance, the chemical stability during processing, flow behavior, and

potential disruptors that can damage processing equipment (e.g., furnaces, pulping machines, etc.) (Fortman et al., 2018; Hyvärinen et al., 2020).

Aesthetical properties

The quality of materials are, among other things, determined by a considerable number of variables related to aesthetics, such as odor, color, texture, durability, etc. (Kol et al., 2021).

Chemical load

After the production and use phase, materials often contain certain chemical compounds, for example, colorants, stabilizers, additives which are added for physico-mechanical functionality of the material, or non-intentionally added substances (NIAS) such as degradation products. Depending on the type and concentration of given compounds (e.g., substances of very high concern (SVHC)), the presence of such compounds can deteriorate the quality of the material and limit the application potential (Hahladakis et al., 2018).

Legal boundaries

Materials need to comply to certain legislation or standards that are in place, such as food contact requirements set by the European Food Safety Authority (EFSA) and Registration, Evaluation, and Authorization of Chemicals (REACH) legislation. Certain legal aspects influence whether or not the produced recyclates are accepted in a certain market segments (De Tandt et al., 2021).

These requirements, together with their definitions and some illustrative aspects that influence these requirement for plastics, glass, paper and cardboard, and aluminum are shown in Table 14. Slight adaptations or additions might be needed in consultation with stakeholders with regard to certain material groups, but the intention is that this frame can generally be followed across material groups. Table 14: Main requirements that must be included in the assessment of the technical material quality for a given secondary material, their definitions, and aspects that influence the quality per requirement for given materials. (POPs = persistent organic pollutants, WEEE = waste from electrical and electronic equipment, RoHS = restriction of hazardous substances, FCM = food-contact material).

Material	Mechanical properties	Processability	Aesthetical properties	Chemical load	Legal boundaries
<i>Definition</i>	<i>Set of properties that a material exhibits upon the application of forces</i>	<i>Behavior of a material during the various processing stages</i>	<i>Set of properties of a material that are ascribed by aesthetic judgment</i>	<i>Content of chemical compounds present in a material</i>	<i>Legislation and standards that are applicable on the use of secondary materials</i>
Plastics	Strength (e.g., tensile strength, ISO 527), stiffness (e.g., tensile modulus, ISO 527), and impact strength (e.g., Charpy impact strength, ISO 179) are the predominant mechanical characteristics (Van Belle et al., 2020). In general, the presence of foreign polymers changes certain mechanical properties due to potential compatibility issues between polymer blends (Huysman et al., 2017).	Determined by the flow behavior e.g., via Melt Flow Index (MFI) measured according to ISO 1133. The preferred MFI depends on the used polymer processing technique (Demets et al., 2021).	Color and odor are limiting factors for some product uses (e.g., packaging applications) (Grant et al., 2020).	Plastics contain IAS such as additives to improve physicochemical properties or other substances such as monomers, solvents or processing aids (Ügdüler, Van Geem, Roosen, et al., 2020) and NIAS, such as impurities, oligomers, or degradation products (Groh et al., 2019).	The production, use and recycling of plastics are regulated by international regulations such as REACH, POPs Regulations, WEEE, RoHS, FCM Regulation.
Glass	The mechanical quality of glass is strongly affected by the presence of glass types different from the main glass cullet type, depending on their chemical composition (Testa et al., 2017). E.g., production of flat glass is only possible from flat glass (or from raw materials).	Contamination of non-glass materials should meet certain specifications (e.g. FERVER: ceramics <100 ppm, ferrous materials <10 ppm, non-ferrous Al <25 ppm, non-ferrous Pb <35 ppm, synthetics <200 ppm, opal <500 ppm, and organics <2000 ppm) to avoid defects in the walls and bottom of the glass furnaces and other equipment (Rodriguez Vietez et al., 2011).	Maximum accepted levels of false colors in a given colored cullet differ according to the desired product quality by the manufacturer and vary between 4-5% for grade A, 5-30% for grade B, and more than 6-30% for grade C and D glass (Rodriguez Vietez et al., 2011).	Heavy metals (e.g., Co, Cr, Ni, Sb) are a concern related to the glass recycling as they can cause processability issues and adversely affect the color properties of the mix of cullet and raw materials (Beerkens & Santen, 2008).	Glass is exempted from REACH obligations, unless: "they meet the criteria for classification as dangerous according to Directive 67/548/EEC and Directive 1999/45/EC" (Rodriguez Vietez et al., 2011).
Paper and cardboard	Recycling paper and carton board decreases the quality of the fibers due to the chemical and mechanical treatments that occur, causing irreversible changes in the fiber structure and properties (Cabalova et al., 2011; Hubbe et al., 2007). One of the main reason is hornification (i.e., loss of ability of the fibers to swell with water) of cellulose fibers which reduces the bonding	Contamination of non-paper materials (e.g., plastics, adhesives) are undesired for technical reasons (waste contaminants are difficult to remove as equipment is not prepared to receive it). The UK's Confederation of Paper Industries allows a maximum of 2 wt% of non-paper components. Within this, a maximum of 0.5% for metal cans, plastic	Odor is an important aspect in the perceiving properties of paper and cardboard packaging and limits the application range (Czerny, 2017).	A wide range of chemicals occurs in paper and paper products with paper conversion (printing, gluing, etc.) being the main source (Pivnenko et al., 2014). Additionally, chemicals may be unintentionally added through impurities and as chemical by-products, or introduced through	The production, use and recycling of paper are regulated by REACH and by national regulations (e.g., BfR Recommendations for paper and board in Germany)

	potential of recycled fibers with the number of cycles and, consequently, their strength (Balea Martin et al., 2016; Hubbe et al., 2007; Nazhad, 2005; Sanchez-Salvador et al., 2020; Wan et al., 2011).	packaging, glass containers, and 0.1% for others (e.g., textiles, wood, non-container glass) (Villanueva & Eder, 2011).		contamination during use (Muncke, 2009).	
Aluminum	Oxides lead to the formation of microcracks and voids. Hence, Al scrap cannot contain excessive oxide in any form.	The EN 13920 standard on Al and Al alloy scrap covers all types of Al scrap and provides limits on the content of foreign materials (e.g., EN 13920-13 indicates that following tolerances apply to the delivered mass: 0.5 m% of magnetic Fe, and 5 m% of moisture and oil, 3 m% of fines, after drying).		In the case of Al, many problematic impurities can occur, including Si, Mg, Ni, Zn, Pb, Cr, Fe, Cu, V, and Mn. Compared to many metals, Al presents a high degree of difficulty in the removal of tramp elements, due to thermodynamic barriers (Gaustad et al., 2012).	The production, use and recycling of Al are regulated by international regulations such as REACH and by the technical guide for metals and alloys by the Council of Europe

8.2.2 Market weight (W_m)

The W_m represents the market share of a given application i (e.g., bottles) in the market of the corresponding material j (e.g., PET), as indicated by Equation 35. Hence, the W_m weighs the TSS towards how important that substitution is in the total market on a mass basis.

$$W_{m,i,j} = \frac{\text{Total production of a specific application } i}{\text{Total production of a material } j} \cdot 100\%$$

Equation 35

Ideally, the sum of the markets in which secondary materials can be used covers the size of the total market for that material. If this would not be the case, and thus the end-use markets in which secondary materials could be used are smaller, virgin materials remain needed for the other markets (Andreasi Bassi, Tonini, et al., 2021).

8.2.3 End of life recycling rate (EOL-RR)

A second indicator to consider in the assessment of the quality of recycling is the EOL-RR (i.e., the combination of U^{rec} and recycling efficiency ' η^{rec} ' as applied by Vadenbo et al. (2017)). The EOL-RR focuses on how efficient recycling industries and recycling routes in the EU are. EOL-RR captures the amount of secondary material j recovered and functionally recycled at end-of-life (EOL) compared to the overall waste quantities generated (Talens Peiró et al., 2018) (see Equation 36) and, hence, accounts for the overall losses in the recovery processes until the point of substitution.

$$EOL - RR = \frac{m_{\text{secondary material } j \text{ functionally recycled}}}{m_{\text{generated waste of material } j}} \cdot 100\%$$

Equation 36

If the technical quality (in the current framework defined as TSS) is high, as is, for instance, often the case in closed-loop recycling, the recycling process is generally considered high quality. However, creating high-quality recycling outputs can in some cases negatively impact the EOL-RR and thus also the dissipative flows. This is in waste management known as the quality-quantity trade-off (Brouwer et al., 2019; Hahladakis & Iacovidou, 2019). Hence, more compensatory virgin material is needed for replacing the physical losses that occur during the recycling chain. It is thus important that not only the technical properties of the secondary materials are considered in the evaluation of the substitution potential, but also the EOL-RR.

8.2.4 Economic boundaries conditions (EBC)

An important condition towards industrial application is the economic feasibility of a recycling pathway. In general, it is likely that producing higher quality recyclates will incur additional costs to a recycling plant. Therefore, the demand and value of the high-quality materials needs to be sufficiently high to cover these costs. Hence, recycling pathways need to meet the economic condition that the price of the recyclates (which is influenced by their quality) should cover the costs of the recycling pathway. Yet, there can also be a 'willingness to pay' (WTP) for the circular economy of a certain material, which comes back for example in EPR fees paid by producers for the EOL stage of their products brought on the market. Previous research suggests that a WTP exists for recycled products, if the material is environmentally friendly and of sufficient high quality (Zwicker et al., 2021). If the revenue of the recyclates and the WTP cannot cover the cost of the recycling pathway, the pathway cannot be performed in an economic context of making profit.

In the current framework, the EBC assesses the economic feasibility of the recycling pathway by attributing a score of zero to the EBC in case the recycling pathway would incur too high costs to make it profitable, resulting in a TSP value of zero. Thus, if the $revenue\ recycles - cost + WTP_j > 0$, then the EBC = 1 and if the $revenue\ recycles - cost + WTP_j < 0$, then the EBC = 0.

8.2.5 Example calculation of the TSP

For clarity's sake, the TSP of a fictitious example is illustratively calculated. Assuming the recycling of a given application *i* (e.g., PET bottles). The recycling pathway has a EOL-RR of 80%. The resulting secondary materials have a TSS of 0.5. The market share W_m in which the secondary materials can be used corresponds to 50%. The EBC is assumed to be 1. Then the TSP is calculated as follows: $TSP = (0.5 \cdot 50\% \cdot 80\% \cdot 1) = 20\%$.

8.3 Long-term-in-use occupation (LTUO)

Quality of recycling can also be seen as the path that ensures the longest durability of the material in the economy (Moraga et al., 2020). In a circular economy, waste disposal is replaced with strategies that aim to maintain and recover resources in production and consumption for as long as possible (Kirchherr et al., 2017). The inaccessibility of material resources is caused by anthropogenic compromising actions related to exploration, environmental dissipation, hibernation, and in-use occupation (van Oers et al., 2019). Moraga et al. (2021) indicated that the concept of in-use occupation is of particular interest as the purpose of any extracted resource is to remain in a useful state. It was stated by the authors that the time dimension is a key parameter for the circular economy, however, this aspect is often disregarded in many circularity indicators. To take this into account, an indicator is adopted from the work of Moraga et al. (2021), namely the long-term in-use occupation. The LTUO is defined as the mass of raw material kept in the material loop over time. Mathematically, this can be described by following Equation 37.

$$LTUO = \int_{T_0}^{T_1} M(t) dt$$

Equation 37

Where $M(t)$ is the function that describes the relationship between time and the mass of a given material and T_0 and T_1 denotes the upper and lower bound constant of integration (T_0 equals to 0 years and T_1 is the number of years it takes to have a residual mass of the material of less than 1 %). By plotting the mass of material as a function of time, the LTUO is visually represented by the area under the graph. Ideally, the LTUO is as high as possible. This can be achieved by increasing the EOL-RR on the one hand and extending the useful lifespan of the secondary material on the other.

The LTUO can visually be presented via a two-dimensional area chart for the occupation of materials in a cascade of products (see Figure 19). The chart exemplifies the different types of occupation (tonne*years), where the y- and x-axes represent mass (tonne) and time (years), respectively. Each in-use occupation is the result of the amount of material entering the phase for a certain amount of time, which is followed by a recycling phase where losses of the material inevitably occur.

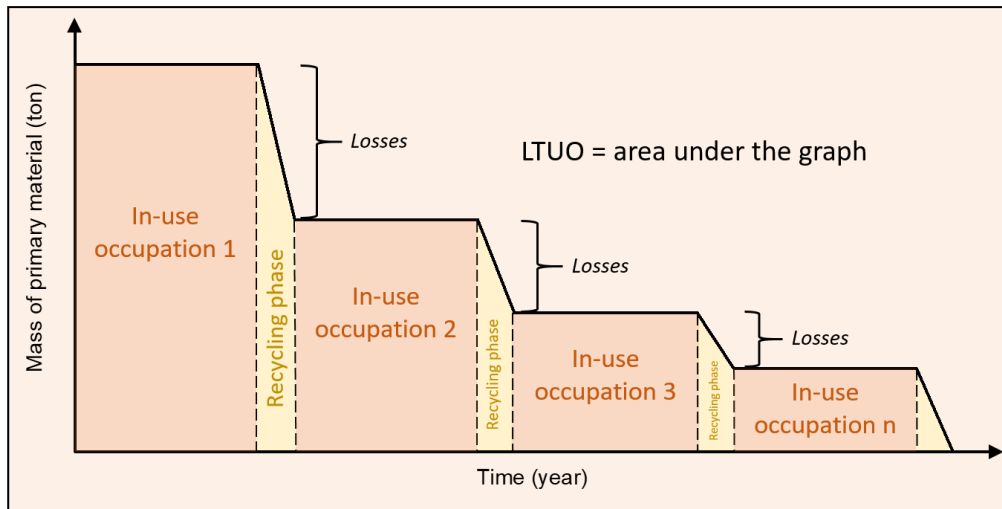


Figure 19: Area chart summarizing the occupation phases and the LTUO dimension used in the quality of recycling framework, adopted from Moraga et al. (2021).

Similar as for the TSP, also for the LTUO, a fictitious example is illustratively calculated. Assuming an initial mass of 100 ktonnes of application *i*. Secondary materials are obtained via a given recycling chain with an EOL-RR of 80%. The lifespan of the application is 1 year and the recycling pathway takes 0.5 years. The graph with time as function of mass is plotted in Figure 20. It can be seen that the LTUO equals in this case to 644.0 tonne*years, which corresponds to the area under the graph. The integration of the resulting function is performed via OriginPro 2016 software (OriginLab).

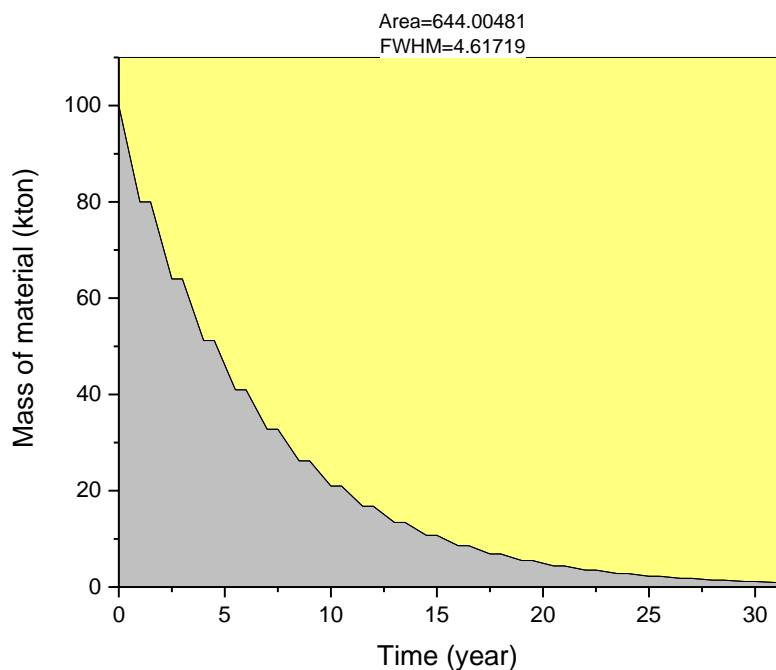


Figure 20: Illustrative example calculation of the LTUO graph generated via OriginPro 2016 software (FWHM=Full Width – Half Maximum).

8.4 Environmental Impact (EI)

The third dimension of the proposed quality framework for recycling is the environmental impact of the recycling process. Logically, the recycling chain should be designed in such a way that it has minimal negative consequences for environment and health. Therefore,

in the current framework, the environmental impact of the recycling chain should be minimized and evaluated by means of environmental footprint calculations in order to guide the waste management sector and to develop meaningful policies.

For this purpose, one can build upon recent standardization works, especially methods such as the Product Environmental Footprint (PEF) (Manfredi et al., 2012) and the Organization Environmental Footprint (OEF) (Pelletier et al., 2012) methods which are proposed by the European Commission as common ways of measuring the potential life cycle environmental impact of a product and organization, respectively. It is suggested that the analysis of the EI follows the principles described by the ISO 14040:2006 framework for LCA. As many standardization efforts have been performed and are still further ongoing, also related to circular economy 'measurements', further elaboration of this indicator is beyond scope. In the further examples in this paper, we will limit ourselves to only the Carbon Footprint impact category for the sake of simplicity.

8.5 Example of application of the quality framework to PET

To demonstrate the operability of the conceptual framework for recycling, a first example is elaborated regarding the recycling of PET. Plastic recycling is generally considered to be more complex compared to the more established recycling pathways of, for instance, steel or aluminum recycling (Walker et al., 2021). Several recycling technologies exist for PET recycling. Conventional extrusion-based mechanical recycling is most widely implemented. An alternative for the mechanical recycling of PET is chemical recycling, which can be done through different pathways, including hydrolysis, glycolysis and methanolysis to depolymerize PET to its monomers. The monomers can subsequently be purified e.g., by distillation or crystallization, and the purified monomers can be introduced again into the polymerization processes of virgin polyesters (Ügdüler, Van Geem, Roosen, et al., 2020). Furthermore, PET waste can have different origins and destinations, including bottles, trays, films and textiles as their main markets (Kawecki et al., 2018).

As a first step, the first dimension of the quality framework, the TSP, is calculated for the recycling of PET bottles, trays and fibers. Hereto, the TSSs are first determined based on the acceptance criteria that are valid for the applications in which the secondary materials will be used. In the current example, the TSSs are arbitrarily assigned as zero (no substitution is possible) or one (full substitution is possible), based on expert judgement and literature (see Table 15). Subsequently, also the Wm is determined based on previous studies (Kawecki et al., 2018; Kuczynski & Geyer, 2010). As a next step, the EOL-RRs are calculated based on literature data (Civancik-Uslu et al., 2021; Gileno & Turci, 2021; Kleinhans et al., 2021; Roosen et al., 2020, 2022; Ügdüler, Van Geem, Denolf, et al., 2020), taking into account the efficiency of the collection system (as is currently the case in Western Europe), the pre-treatment chain (including sorting, washing, float-sink separation, and drying), and the effective recycling process itself. The background data can be found in Table 16. For each of the performed steps, physical losses occur, resulting in an overall EOL-RR of 69% for PET bottles (see Table 15). Textiles, on the other hand, are currently only recycled to a limited extent (global EOL-RR is reported to be 13%), which mainly concerns applications with a lower-value, such as insulation material, wiping cloths, or mattress stuffing, all of which are currently difficult to recapture and therefore likely constitute the final use (Ellen MacArthur Foundation, 2017). The EBC has always been taken as 1, meaning that all chains are assumed to be economically feasible.

Table 15: Calculations of the TSP of different PET recycling pathways by determining the TSS, Wm, EOL-RR, and EBC. The TSP is the sum of the SP of each waste product (in the table indicated by the row 'recycling from').

Recycling from →		Bottle								Tray								Fiber							
Recycling path →		Mechanical				Hydrolysis				Mechanical				Hydrolysis				Mechanical				Hydrolysis			
Recycli ng to↓	Market weight	EOL-RR	TSS	EBC	SP (%)	EOL-RR	TSS	EBC	SP (%)	EOL-RR	TSS	EBC	SP (%)	EOL-RR	TSS	EBC	SP (%)	EOL-RR	TSS	EBC	SP (%)	EOL-RR	TSS	EBC	SP (%)
Bottle	39%	69	1	1	27	62	1	1	25	48	0	1	0	43	1	1	17	13	0	1	0	12	1	1	5
Tray	11%	69	1	1	8	62	1	1	7	48	0	1	0	43	1	1	5	13	0	1	0	12	1	1	1
Film	4%	69	1	1	3	62	1	1	3	48	0	1	0	43	1	1	2	13	0	1	0	12	1	1	0
Fiber	44%	69	1	1	31	62	1	1	28	48	1	1	22	43	1	1	19	13	1	1	6	12	1	1	5
	Sum: 98%	TSP			69	TSP			62	TSP			22	TSP			43	TSP			6	TSP			12

Table 16: Background data to determine the estimated EOL-RR of PET recycling.

	Collection rate	Recycling efficiency (after collection)	
		Mechanically	Chemically
Bottles	81%	85%	77%
Trays	57%	85%	77%
Fibers	13%		12%

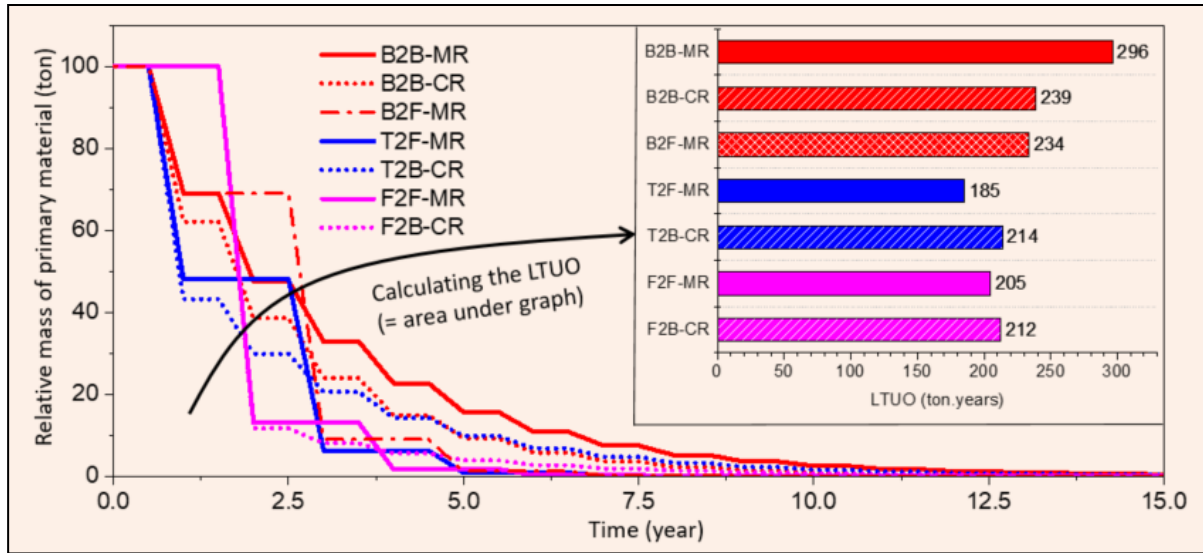


Figure 21: The LTUOs of the analyzed PET recycling pathways, visualized by plotting the mass of raw material (tonne) as a function of time (year).

It can be seen that the mechanical recycling pathway for bottles has the highest TSP of 69%. The lowest TSP of 6% is obtained for the mechanical recycling of polyester fibers due to the limited applications in which it can be mechanically recycled and the relatively low EOL-RR. The chemical recycling pathway has in general a lower EOL-RR compared to mechanical recycling due to losses that occur during alkaline hydrolysis. However, for trays and fibers, the monomers that are produced via chemical recycling have a higher TSS compared to mechanically recycled trays and fibers, resulting in a higher TSP for these applications. As a next step, the second dimension of the recycling framework, the LTUO, is determined for seven scenarios:

- Mechanical recycling of PET bottles to PET bottles (B2B-MR);
- Chemical recycling of PET bottles to PET bottles (B2B-CR);
- Mechanical recycling of PET bottles to polyester fibers (B2F-MR)
- Mechanical recycling of PET trays to polyester fibers (T2F-MR);
- Chemical recycling of PET trays to PET bottles (T2B-CR);
- Mechanical recycling of polyester fibers to polyester fibers (F2F-MR);
- Chemical recycling of polyester fiber to PET bottles (F2B-CR)).

In addition to the EOL-RR, also the lifespan of the different products is needed to calculate the LTUO. Based on previous studies, the lifespan of a PET bottle is assumed to be 0.5 years (Geyer et al., 2017) and the lifespan of textile applications 1.5 years (Henry et al., 2015). Figure 21 shows the LTUOs of the investigated PET recycling scenarios by plotting the relative mass of the material as a function of time. The background data of this graph can be found in Table 17. It can be observed that the type of recycling and the application in which the material is recycled impact the area under the graphs (i.e., the LTUO). For instance, the LTUO corresponds to 296 tonne*years for the B2B-MR pathway, whereas for the B2F-MR pathway, the LTUO is only 234 tonne*years.

Table 17: Background data of the LTUO for PET recycling.

Time (year)	B2B-MR	Time (year)	B2B-CR	Time (year)	T2F-MR	Time (year)	T2B-CR	Time (year)	F2F-MR	Time (year)	F2B-CR	Time (year)	B2F-MR
0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0
0.5	100.0	0.5	100.0	0.5	100.0	0.5	100.0	1.5	100.0	1.5	100.0	0.5	100.0
1.0	69.0	1.0	62.1	1.0	48.0	1.0	43.2	2.0	13.0	2.0	11.7	1.0	69.0
1.5	69.0	1.5	62.1	2.5	48.0	1.5	43.2	3.5	13.0	2.5	11.7	2.5	69.0
2.0	47.6	2.0	38.6	3.0	6.2	2.0	29.8	4.0	1.7	3.0	8.1	3.0	9.0
2.5	47.6	2.5	38.6	4.5	6.2	2.5	29.8	5.5	1.7	3.5	8.1	4.5	9.0
3.0	32.9	3.0	23.9	5.0	0.8	3.0	20.6	6.0	0.2	4.0	5.6	5.0	1.2
3.5	32.9	3.5	23.9	6.5	0.8	3.5	20.6	7.5	0.2	4.5	5.6	6.5	1.2
4.0	22.7	4.0	14.9	7.0	0.1	4.0	14.2	8.0	0.0	5.0	3.8	7.0	0.2
4.5	22.7	4.5	14.9	8.5	0.1	4.5	14.2			5.5	3.8	8.5	0.2
5.0	15.6	5.0	9.2	9.0	0.0	5.0	9.8			6.0	2.7	9.0	0.0
5.5	15.6	5.5	9.2			5.5	9.8			6.5	2.7		
6.0	10.8	6.0	5.7			6.0	6.8			7.0	1.8		
6.5	10.8	6.5	5.7			6.5	6.8			7.5	1.8		
7.0	7.4	7.0	3.6			7.0	4.7			8.0	1.3		
7.5	7.4	7.5	3.6			7.5	4.7			8.5	1.3		
8.0	5.1	8.0	2.2			8.0	3.2			9.0	0.9		
8.5	5.1	8.5	2.2			8.5	3.2			9.5	0.9		
9.0	3.5	9.0	1.4			9.0	2.2			10.0	0.6		
9.5	3.5	9.5	1.4			9.5	2.2			10.5	0.6		
10.0	2.4	10.0	0.9			10.0	1.5			11.0	0.4		
10.5	2.4	10.5	0.9			10.5	1.5			11.5	0.4		
11.0	1.7	11.0	0.5			11.0	1.1			12.0	0.3		
11.5	1.7	11.5	0.5			11.5	1.1			12.5	0.3		
12.0	1.2	12.0	0.3			12.0	0.7			13.0	0.2		
12.5	1.2	12.5	0.3			12.5	0.7			13.5	0.2		
13.0	0.8	13.0	0.2			13.0	0.5			14.0	0.1		
13.5	0.8	13.5	0.2			13.5	0.5			14.5	0.1		
14.0	0.6	14.0	0.1			14.0	0.3			15.0	0.1		
14.5	0.6	14.5	0.1			14.5	0.3			15.5	0.1		
15.0	0.4	15.0	0.1			15.0	0.2			16.0	0.1		

Thus, by recycling PET bottles again into new bottles, the LTUO is significantly higher compared to recycling PET bottles into textile applications. For PET trays and polyester

fiber recycling, it can be seen that chemical recycling results in a higher LTUO (214 and 212 tonne*years) compared to mechanical recycling (185 and 205 tonne*years).

The third dimension of the quality framework concerns the EI calculations. For the illustrative example in this study, the EI is based on previous literature (Gileno & Turci, 2021), where a "cradle-to-gate" LCA of two PET bottle recycling pathways were carried out, namely the production of one kg of post-consumer mechanically recycled PET by a bottle-to-bottle recycling process and the production of one kg of recycled polyester fiber by a bottle-to-fiber recycling process. For the chemical recycling of PET packaging, the carbon footprint analysis of Ügdüler, Van Geem, Denolf, et al. (2020) was adopted.

In the current framework, we give equal weight to each dimension to determine the quality of recycling. Hence, to avoid that one of the three dimensions would have a more significant impact on the distance calculation due to the difference of scale, a normalization is needed. This is done by means of Equation 38, resulting in a value between zero and one for each dimension.

$$r_{normalized} = \frac{r - r_{min}}{r_{max} - r_{min}}$$

Equation 38

Defining the minimum and maximum values impact the normalization and, hence, also the distance calculations. In the current recycling analysis, we apply the minimum and maximum values as depicted in Table 18.

Table 18: Values used to standardize the three dimensions used in the current PET recycling example.

	Lowest quality (r_{min})	Highest quality (r_{max})
TSP (%)	0	100
LTUO (tonne*years)	0	1500 (100 tonne * 15 year)
EI (CO ₂ -eq./kg waste managed)	5.7 (carbon footprint for incineration)	0

In order to determine the recycling pathway with the highest quality, the three dimensions of the framework are included in a three-dimensional (3D) scatter plot that includes each recycling pathway as an individual point. Based on the Cartesian coordinates of the points included in the 3D graph, the Euclidean distance between each point of the analyzed recycling pathways (as standardized values) and the point of the optimal quality of recycling is calculated by means of following Equation 39 with r being the point that represents a given recycling pathway and o the point of optimal quality.

$$d(r, o) = \sqrt{(r_{TSP} - o_{TSP})^2 + (r_{LTUO} - o_{LTUO})^2 + (r_{EF} - o_{EF})^2}$$

Equation 39

In Figure 22a, the three dimensions of the quality framework (i.e., the TSP, LTUO, and EI) are visualized in a three-dimensional scatter plot for the analyzed scenarios. As indicated in the graph, the optimal quality of recycling is one where the EI would be zero and the TSP and LTUO maximum (thus one since it concerns standardized values). This is of course a theoretical optimum that is technically not feasible with the current waste management processes in place. Yet, the main aim of the graph is to provide an indication of the scenarios with the highest quality (recycling pathway to be promoted) and the scenarios with the lowest quality (less interesting recycling pathway) for a certain waste product (e.g., bottles, trays, and fibers). The closer the scenario is located to the 'optimal quality point', the better the quality of recycling. The distance from each of the six data points to the optimal quality point is calculated and visualized in Figure 22b. It can be noted that the highest distance for the recycling of PET bottles (1.01), and thus the lowest quality of

recycling, is achieved by chemical recycling, while the lowest distance (0.88), and thus the highest quality of recycling, is achieved by the mechanical recycling of bottles to bottles. For trays, on the other hand, chemical recycling resulted in a lower distance (1.11) compared to mechanical recycling (1.19). Finally, for fibers, both pathways practically obtained the same value for the distance to the point of highest quality, and thus the lowest global quality within the example, being slightly lower for mechanical recycling (1.29) compared to chemical recycling (1.30).

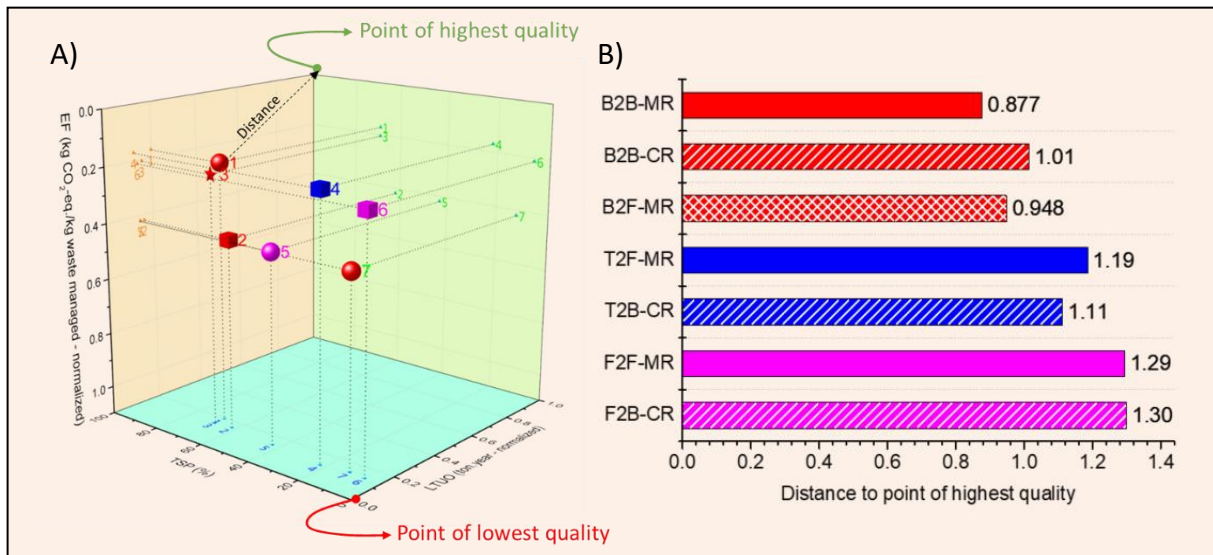


Figure 22: Overview of the quality framework applied on the seven analyzed PET recycling scenarios. a) 3D graph with indicated the analyzed scenarios; b) The distance calculated to the 'point of highest quality'.

8.6 Outlook

Recycling is an important strategy in the circular economy. However, definitions of quality in recycling are scarce and a widely supported framework to measure the quality of recycling has not yet been established in an industrial or policy-making context. In the current work, an operational framework is theoretically described and demonstrated in a case study on PET.

The framework builds upon three dimensions. The first dimension is the Total Substitution Potential (TSP), which indicates to which extent a secondary material can provide the same function(s) as the primary material. The TSP value depends on the Technical Suitability for Substitution (TSS), the Market weight (Wm) which represents the market share of a given application in the market of the corresponding material, the End-Of-Life Recycling Rate (EOL-RR), and the Economic Boundary Conditions (EBC). The second dimension is the Long-Term in-Use Occupation (LTUO), which indicates how much of a certain material is still 'in use/functional' in society over a certain time horizon. The third and last dimension is the Environmental Impact (EI), which can be measured by common LCA approaches.

Each of the three dimensions defined in the presented framework can be evaluated mathematically, which offers a quantitative way to assess the quality of recycling and, hence, compare different recycling techniques, different applications, geographical difference in waste management practices, etc. The framework is applied to the case of recycling of PET. The results indicated that closed-loop mechanical recycling of PET bottles has the highest quality of recycling, whereas the chemical recycling of PET fibers has the lowest quality of recycling.

Making a framework for assessing the quality of recycling operational is an important step forward for industry, policy makers, and researchers to steer development in waste management processes. This section presents an operational framework, integrating many aspects of quality allowing integrating over the different dimensions. This allows to assess

the recycling pathways that are less interesting from a quality perspective or the pathways that should be included in future recycling advances. Obviously, the framework requires further elaboration together with different stakeholders and scientists, per material type, for example to set more exact technical property ranges, to develop LTUO scenarios, etc. Yet, the presented work can offer a solid frame for such discussions across different material types. It can thus be of great value in moving forward in the transition to a more circular economy for materials and resources.

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List of abbreviations and definitions

Calculation point: We provide two definitions: the former from a recycled material perspective and the latter from a waste perspective. The calculation point is the point at which the mass of material of a given stream is accounted for as being 'recycled' for the purpose of achieving the targets (Hogg et al., 2020). According to European Commission (2019a, 2019b) the calculation point is defined as the point where municipal waste and packaging waste are reprocessed into products, materials or substances that are not waste, or the point where waste materials cease to be waste as result of a preparatory operation before being reprocessed.

Chain of custody: process by which inputs and outputs and associated information are transferred, monitored and controlled as they move through each step in the relevant supply chain (ISO 22095).

Chain of custody model: approach taken to control inputs and outputs and associated information in a particular chain of custody system (ISO 22095).

Co-materials: Amount of ancillary materials aiding the process and usually recovered at the end of the recycling process not impacting the recycling rates (e.g., water, enzymes).

End-of-life recycling rate: Efficiency of the entire recycling chain.

Energy recovered: Amount of mass recovered as energy in the recycling process.

Intermediates: Amount of material that flows into a subsequent sub-process to generate output materials.

Measurement point: The point where the mass of waste materials is measured with a view to determining the amount of waste at the calculation point (European Commission, 2019a) Within this study, a measurement point does not strictly refer to the definition stated before, but it might also refer to a general measurement point for different materials (i.e., waste, non-waste, intermediates) necessary to conduct the mass balance.

Recycled material: In the context of this study, it refers generically to any product, material or substance obtained via recycling, which does not undergo any further treatment and is placed on the market whether for the original or other purposes, excluding use as fuel, conforming with European Commission (2019b).

Recycling process: A system including the sub-processes occurring in the recycling plant up to the final output material.

Recycling chain: A system including the sub-processes occurring in the entire value chain of waste management-for-recycling, i.e., segregation, collection, sorting and final recycling process.

Recycling yield: The ratio between the amount of a waste recycled in a given recycling process and the input waste feedstock to that recycling process. It should be noted that among the input's components, only the waste feedstock contributes to the recycling yield.

Virgin feedstock: Amount of virgin (primary) feedstock that is used in the process, e.g., fossil or bio-based feedstock.

Specified characteristic: Aet of product and/or production characteristics that the chain of custody is designed to maintain.

Waste feedstock: Amount of waste that is used in the recycling process to produce recycled materials.

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Annexes

Annex 1. Stakeholder consultations

Workshop I

On June 22nd, 2021, a workshop was held with stakeholders (SHs) with the purpose to illustrate the project scope and objectives and to engage SHs in the discussions. In agreement with DG ENV, 280 people were invited to the workshop. A total of 133 people finally attended the workshop, from industry and member state authorities (MS) mainly, but also from NGOs and academia. The pillars upon which the discussions were held, coincided with the three main objectives of the project:

- To identify any relevant recycling process that is excluded from the current definition of recycling and on which further assessment and guidance is necessary to define appropriate calculation rules.
- To identify appropriate calculation rules for the estimation of the recycling rate for such processes (with special attention to chemical recycling).
- To discuss and suggest potentially relevant approaches for defining quality of recycling.

The final agenda of the workshop is reported herein:

- 09.15-09.30: Virtual room opens + housekeeping rules
- 09.30-09.45: Welcome and policy framework
- 09.45-09.50: Project output and objectives
- 09.50-10.20: Q&A on policy background and project objectives
- 10.20-10.30: Objective 1: Capturing relevant aspects
- 10.30-11.00: Objective 2: Discussion panel
- 11.00-11.10: Break
- 11.10-11.40: Objective 3: Discussion panel
- 11.40-12.00: Wrap up and conclusions
- 12.00: Closure of the meeting

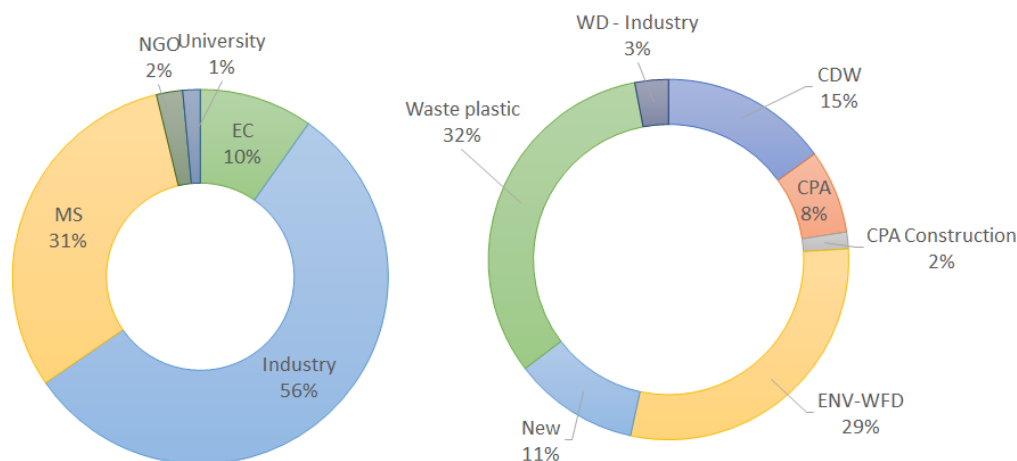


Figure 23: Breakdown of the participants to the workshop. CDW: construction and demolition waste; CPA: circular plastic alliance; EC: EU Commission; MS: member states; WFD: waste framework directive.

The most relevant inputs and feedbacks received during the workshop, subdivided per each of the 3 discussion panels (Q1-to-Q3), are summarised herein:

Discussion objective 1:

Chemical recycling and multi-output processes producing a mix of materials, chemicals, energy, and fuels from waste were largely identified as the relevant processes currently excluded from the recycling calculation rules and on which further guidance is necessary.

Discussion objective 2:

- A mass balance approach was identified as appropriate method to handle recycling rates in complex multi-output processes.
- The environmental performance of the recycling processes (environmental impacts, C-footprint, energy/water demand, etc.) was also pointed out as a relevant aspect to be taken into account.
- The quality of the recyclate and the consequent market displacement of virgin materials was identified as a relevant aspect (market aspects).
- Ensuring clarity and consistency in the calculation rules with other recycling processes (e.g., mechanical recycling). For example, it was pointed out that very theoretical approaches should be avoided. More practical examples and guidance should be provided. These guidelines should also make clarity on what data and statistics shall be used, and how these should be used in practice via specific examples. In general, calculation rules shall be made more simple and practical to be used.

Discussion objective 3:

- High quality was generally identified as that situation where the material fulfils precisely the same functions as the virgin product counterpart. The final application becomes therefore an important element to determine whether the recycling is of high or low quality.
- Product specifications, standards and environmental profiles were identified as important attributes to describe quality.
- Some SHs pointed out that the quality also depends on the input-waste streams, i.e., some streams are challenging to recycle compared with others. This difference should not be neglected when describing a process as low or high quality.
- Some SHs pinpointed the concept of a "recyclability threshold" (e.g., at least 95% of the functional unit of product shall be recyclable). It was argued that thresholds on recyclability are quite important, as they are under discussion in several areas.
- There were also some critiques against the concept of high-quality recycling; these went into the direction that the market (buyers) would decide what to buy depending on their needs and final application of the material; it was pointed out that quality has little to do with overall recycling rates and targets.

Following the workshop, a survey with some key questions was submitted to SHs involved in emerging recycling processes (see the specific questions in Annex 2). A background paper was also prepared and provided to the SHs with the purpose to illustrate the project scope and objectives and to engage SHs in the discussion. The objective of the SHs consultation is to support a possible revision of the calculation rules for recycling rates provided in Commission Implementing Decision 2019/1004/EU. In this Annex, the elaboration of the results from the SH consultation is summarised.

The need of revising the calculation rules is especially relevant for processes that are currently not included in the definition of recycling. Therefore, we initially asked SHs if any relevant additional process should be considered so as to identify any relevant recycling process currently excluded from the Waste Framework Directive. Most of SHs argued that chemical recycling is presently a process not appropriately covered by the definition of recycling (Figure 24) and they suggested for an alignment to a proper method for calculating recycling rates in operations where multiple materials enter recovery operations and result in output that includes recycled materials, fuel or backfilling materials (Figure 25).

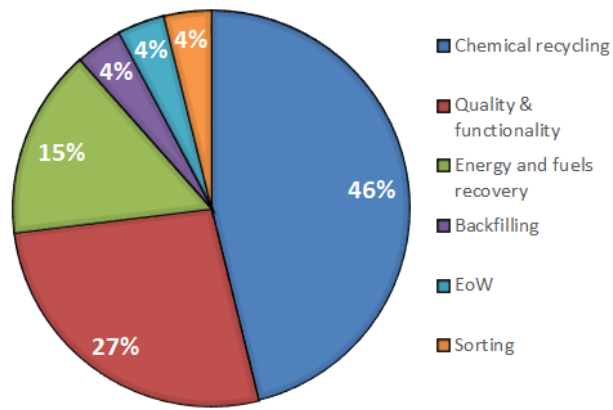


Figure 24: The relevant processes which are not currently covered by the definition of recycling, according to the SHs consultation.

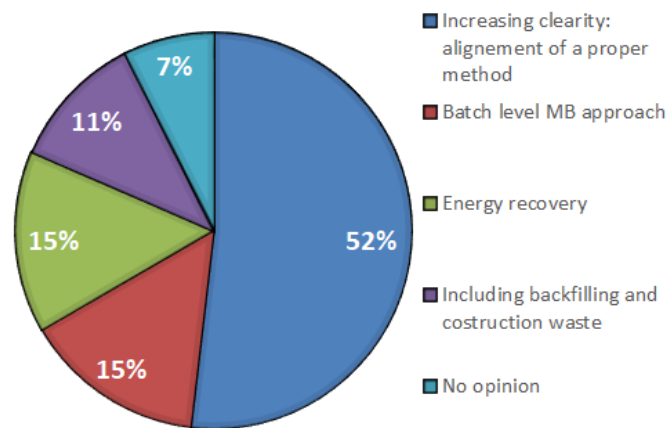


Figure 25: The improvements proposed for the current calculation rules, according to the SHs consultation. MB: mass balance.

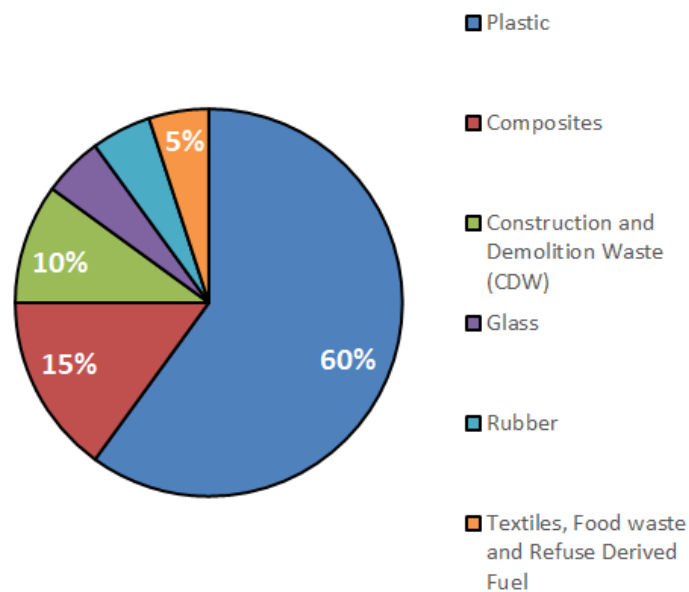


Figure 26: The relevant waste streams requiring further clarifications in the current calculation rules, according to the SHs consultation.

More in detail, according to the SH consultation, 36% of SHs identified the need for clarifications on the calculation losses and energy recovery as well as on the non-targeted materials within the calculation rules. Furthermore, focusing on the waste streams that require further clarifications in the calculation rules as in Commission Implementing Decision 2019/1004, SHs mostly highlighted that plastic is the waste stream where a higher level of detail is needed (Figure 26). In connection with this, 80% of SHs recommend calculating the share of recycling in a chemical recycling process by using a mass balance approach (Figure 27).

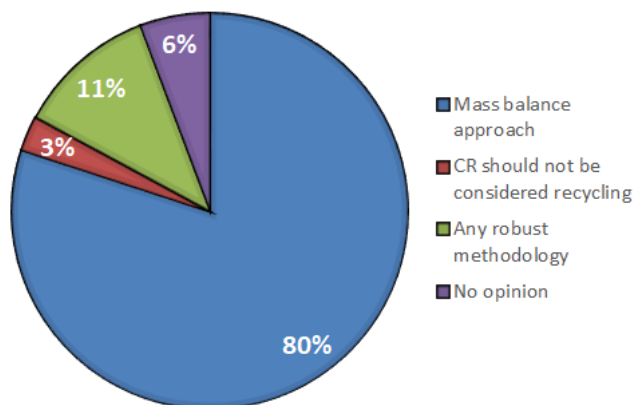


Figure 27: The recommended approach to calculate the share of recycling in a chemical recycling process, according to the SHs consultation. CR: chemical recycling.

On top of the issues previously described, another criticality is represented by the calculation point in recycling processes, namely which inputs and outputs throughout the chain should be considered to perform the calculation. Although calculation points applicable to certain waste and materials and certain operations are specified in the Commission Implementing Decision 2019/1004 (European Commission, 2019a), this is not straightforward for chemical processes. In this respect, most SHs suggested that the calculation point for chemical recycling processes shall be the same as for mechanical recycling, namely at the final recycling operation where the output of a recovery operation leads to either the polymerization of a feedstock or production of another chemical according to the current definition in the Waste Framework Directive. Only 4% of the SHs supported new or different calculation points (Figure 28).

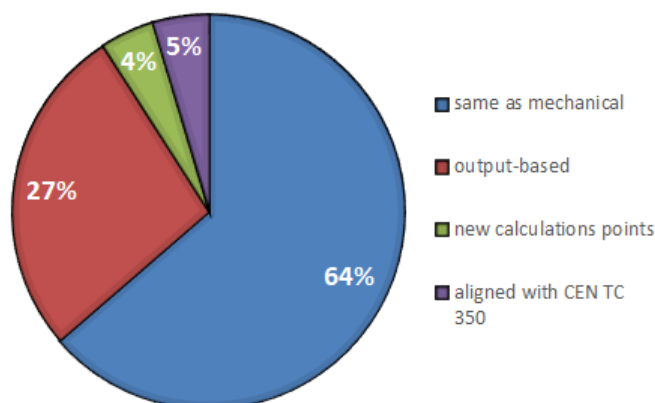


Figure 28: The recommended approach to define the calculation point in a chemical recycling process, according to the SHs consultation.

Moreover, 64% SHs recommended to apply this general rule, i.e., same calculation point as mechanical recycling, to any chemical recycling process treating waste such as plastics,

textiles, composites, organic and wood waste. Furthermore, 80% of SHs suggested not to modify the current approach used to calculate the share of recycling in composting and anaerobic digestion processes. Finally, 46% of the SHs did define themselves aware of definitions of “high” and “low” quality recycling means in practice (whereas 28% did not), identifying specifications for uses and separation at source as the main parameters determining quality (Figure 29).

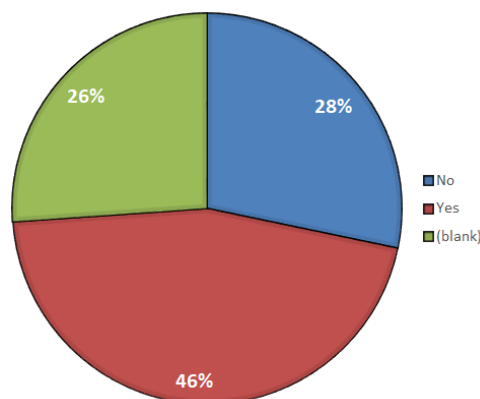


Figure 29: The answer to the question: are you aware of definitions being used in practice of high or low quality recycling?

Summarizing, these are the main results/recommendations obtained from the SHs consultation:

- Chemical recycling is a relevant recycling process but seems not adequately addressed (excluded) in the current definition of recycling of the Waste Framework Directive and related Implementing Decisions.
- Chemical recycling should be included in the current definition of recycling of the Waste Framework Directive.
- An alignment method should be introduced for calculating recycling rates in operations with multiple input/output such as chemical recycling operations.
- There is a need of clarification for losses, energy recovery and non-target materials when calculating the share of recycling.
- Plastic is the waste stream mostly affected by these shortcomings.
- The mass balance approach is highly recommended to calculate the share of recycling in chemical recycling processes.
- Calculation points for chemical recycling processes should follow the ones currently used for mechanical recycling.
- No exceptions in waste streams should occur when new rules for chemical recycling process are introduced (i.e., the method shall be generally applicable to all waste streams).
- The current approach used to calculate the share of recycling in composting and anaerobic digestion processes should not be modified.
- Although most of the SHs are aware of “high” and “low” quality recycling definitions, a significant part of them is not.
- Specifications on the recycled material end-uses and separation at source are the main parameters identified by SHs to define “high” and “low” quality recycling.

Workshop II

As a follow up, another workshop with SHs took place on April 7th, 2022. In agreement with DG ENV, 123 people were invited to the workshop. A total of 110 people finally

attended the workshop, from industry and member state authorities (MS) mainly, but also from NGOs and academia.

The final agenda of the workshop is reported herein:

- 9.15-09.30: Log in to webex
- 9.30: Opening of the workshop
- 09.30-09.45: Project background, scope, and status
- 09.45-10.45: Technical proposal for a mass balance approach
- 10.45-11.00: Break
- 11.00-11.45: Technical proposal for compostables and compostable plastic
- 11.45-12.30: Quality of recycling (exploratory work: framework and definition)
- 12.30-13.00: Stakeholder consultation & timeline for project completion
- 13.00: Closure

A second survey was submitted to SHs involved in emerging recycling processes right after workshop II (see the specific questions in Annex 3). The reviewed background paper was again provided to the SHs with the purpose to illustrate the project status and to engage them in further discussion. The objective of this SHs consultation is again to support a possible revision of the calculation rules for recycling yields provided in Commission Implementing Decision 2019/1004/EU. This second SH consultation was profitable and allowed us to improve our general analysis, improving the quality of this report based on recommendations directly provided from SHs. In this chapter, a recap of the advancements generated from the second SH consultation is provided. In particular, here are listed the main amendments carried out on the basis of the recommendations obtained from the second SH consultation:

- The term "rate" has been replaced with the term "yield" as the rate definition that was previously used in the report deviated from existing ones in EU legislation.
- Clarification on the difference between recycling content and recycling yield has been provided.
- As for emerging technologies, in particular for chemical recycling of plastic, the concept of "calculation point" has been substituted by two terms, namely "mass balance beginning point" and "mass balance ending point", which seem more appropriate.
- The issue concerning the choice between proportional and non-proportional allocation has been clarified: the choice about what kind of allocation is used is out of the scope of this report and it is up to the operator to decide what allocation rule to use. This refers to recycled content discussions rather than to recycling yield calculations.
- The previous term "material flowing" has been replaced by "intermediates" to give clarity. Intermediates represent the substances that are manufactured for subsequent sub-processes in the recycling process and consumed in or used for chemical processing in order to be transformed into other substances.
- We have added a new input, namely "co-materials". Co-materials represent the amount of ancillary materials aiding the process, which are usually recovered at the end of the recycling process (e.g., water, enzymes). Within the calculation rules, co-materials do not contribute to the recycling yield.
- A new chapter presenting chain of custody models has been added. This is to remark that the organizations conforming to ISO 22095 shall establish and implement one or more of the chain of custody models for all materials or products with specified characteristics and shall be transparent about the model chosen. The organization shall only use the same chain of custody model as its supplier or a model with lower physical presence of the specified characteristic in the output. The list of (chain of custody) models, ranked from highest to lowest physical presence of the specified characteristics has been presented.

- For the mass balance, two implementation methods are possible: the rolling average percentage method and the credit method. The rolling average percentage method is based on the use of a fluctuating proportion of input, bearing specified characteristics, entering the organization over a defined claim period, allowing a claim of an average percentage to be made for the output over the claim period. In the credit method the recorded output amount shall be equivalent to the physical input, taking into account a conversion factor. Such conversion factor shall be defined within each material or product at each site and it shall be applied to define the amount of credit to enter the credit account when using the output as the basis for calculation, or to subtract the credit when using the input as the basis for calculation. As the SHs highlighted that the rolling average method can be used only for batch type operations, the credit method has been added as an additional option.
- Waste and non-waste feedstock have been further defined. In particular the terms “waste feedstock” and “virgin feedstock” have been introduced. The former is the amount of waste that is used in the recycling process to produce recycled materials and it contributes to the recycling yield. The latter is the amount of virgin (primary) feedstock (either fossil or bio-based feedstock) that is used in the recycling process and it does not contribute to the recycling yield.
- As in a system it is needed to trace information on the material that is transferred from one plant (e.g., a pyrolysis plant) to the other in the recycling chain (e.g., a cracker), a new chapter concerning the traceability has been added. It is suggested that a traceability (auditing and compliance) system similar to that of renewable energy (under the Renewable Energy Directive) or recycled content certification is established also for recycling yields declaration.
- Concerning the possibility to apply the stoichiometry to estimate the proportion of waste feedstock and virgin feedstock generating the mass of recycled material, it has been specified that in case the stoichiometry is known any other chain of custody method with higher physical presence should be implemented.
- The potential impacts expected with the application of the calculation rules have been integrated with some suggestions from SHs.
- Concerning biodegradable waste and compostable plastic waste, the scope for the definition of recycling bio-waste has been enlarged to other technologies (besides composting and AD) that are able to produce output in similar quantity comparing to the benchmark of composting/AD (i.e., 15% of the input mass), provided that the output is used as recycled product, material or substance. In line with that, the existing concept of “similar quantity of recycled content” within the biodegradable waste recycling rule has been substituted by “similar output quantity” since the former was misleading (i.e., it applies to products that incorporate recycled material).
- Besides, inherent losses for bio-waste could include CO₂, CH₄ and water, and they should apply to all processes and technologies used for recycling bio-waste and compostable plastic waste regardless of the process occurring, provided that they fulfil the criteria imposed (i.e., generates similar output quantities taking as benchmark the composting and AD processes, and provided it is used as a recycled product, material or substance).
- A new box discussing the ecological and agricultural benefit of applying on land outputs from biological recycling processes has been added, as a result of the great debate generated during the SHs consultation. The text summarizes the opinions and arguments put forward by the different SHs.

Annex 2. Survey distributed to stakeholders in workshop I

A survey with 12 key questions was performed and provided to the SHs for a consultation. Below, the twelve questions composing the survey are listed:

1. To the aim of identifying any excluded relevant recycling process, is further assessment or guidance on the current definition of recycling of the Waste Framework Directive (Directive 2008/98/EC as last amended by Directive (EU) 2018/851)* and related calculation rules (Commission Implementing Decision 2019/1004/EU and Commission Implementing Decision 2019/665/EU) needed? If yes, which relevant processes are not covered by this definition?
2. Article 3(8) of the Commission Implementing Decision 2019/1004 states "Where municipal waste materials enter recovery operations whereby those materials are not principally used either as fuel or other means to generate energy, or for material recovery, but result in output that includes recycled materials, fuels or backfilling materials in significant proportions, the amount of recycled waste shall be determined by a mass balance approach which results in taking account only of waste materials that are subject to recycling." This rule is also similarly stated in Art. 4c(i) of Commission Implementing Decision 2019/665. Is such rule clear?
3. Is (are) there any other aspect(s) related to calculating recycled municipal waste and recycled municipal bio-waste targets in Articles (3) and (4) of the Commission Implementing Decision 2019/1004 to be clarified? If yes, please explain which aspect(s) and which improvements you would propose for it (them). Please, provide details on the suggested approach.
4. Is (are) there any other aspect(s) related to the calculation rules in Articles (3), (4), (5) and (6) of the Commission Implementing Decision 2019/665 to be clarified? If yes, please explain which aspect(s) and which improvements you would propose for it (them). Please, provide details on the suggested approach.
5. Could you point any relevant processes (and related waste streams treated) that are currently excluded from the calculation rules as in Commission Implementing Decision 2019/1004 (and similarly excluded from Decision 2019/665)? Please select the considered waste stream(s) from the drop down list.
6. Which approach would you apply to calculate the share of recycling in a chemical recycling process* (e.g., a mass balance approach)?
7. Calculation point is defined in Article 1(e) of Commission Implementing Decision 2019/1004*. Calculation points applicable to certain waste materials and certain recycling operations are specified in Annex I of Commission Implementing Decision 2019/1004. Do you have other suggestions on where to put the recycling calculation point for the waste recycling processes you suggest? Furthermore, would you suggest specifying the recycling calculation points for additional waste streams not yet considered in the Commission Implementing Decision 2019/1004. Please explain and select the considered waste stream(s) from the drop down list.
8. Do you have any other correction on input materials, like moisture content correction, or on output materials to suggest? Can you suggest other corrections to be considered (e.g., additives added during recycling or cleaning chemicals)? Please explain.
9. Would the rule above apply in general to any chemical recycling process treating waste, e.g., plastic waste or organic/wood waste (such as biorefineries transforming input waste into chemicals and other products)?
10. Considering the above, should the approach used to calculate the share of recycling in composting and anaerobic digestion processes of (municipal) organic waste* be modified? If yes, what could the approach be?

11. Considering the above, what could then be the approach to calculate the share of recycling in composting of compostable plastic (packaging) waste?
12. Are you aware of definitions being used in practice of high or low quality recycling? If yes, which are the parameters used and the threshold values used to define high and low quality recycling? Please, provide references of the standards or legislation in place. Please, explain whether quality affects market price or contracts. Please, focus particularly to exemplary waste streams such as plastic waste, wood waste or Construction and Demolition Waste (CDW)

Annex 3. Survey distributed to stakeholders in workshop II

A survey with 10 key questions was performed and provided to the SHs for a consultation. Below, the 10 questions composing the survey are listed:

1. Do you want to discuss the quality of recycling framework in a dedicated meeting with JRC?
2. Do you agree with the definition of calculation and measurement points used in the report?
3. Do you agree with the allocation used for calculating the recycling rate for input waste (waste feedstock - non-waste feedstock)?
4. Article 1.13 of the Directive (EU) 2018/851 states "For the purpose of calculating whether the targets laid down in points (c), (d) and (e) of Article 11(2) and in Article 11(3) have been attained, the amount of municipal biodegradable waste that enters aerobic or anaerobic treatment may be counted as recycled where that treatment generates compost, digestate, or other output with a similar quantity of recycled content in relation to input, which is to be used as a recycled product, material or substance." This rule is also similarly stated in Recital (48) of the same Directive. We propose that this calculation rule becomes technology neutral (i.e., not just mentioning aerobic or anaerobic treatment but including any biological, chemical or physical biowaste recycling technology). Do you agree?
5. In the same articles of Question 3: "For the purpose of calculating whether the targets laid down in points (c), (d) and (e) of Article 11(2) and in Article 11(3) have been attained, the amount of municipal biodegradable waste that enters aerobic or anaerobic treatment may be counted as recycled where that treatment generates compost, digestate, or other output with a similar quantity of recycled content in relation to input, which is to be used as a recycled product, material or substance." We find that the concept of recycled content in this context is confusing and needs to be clarified. Do you agree?
6. Concerning inherent losses, Recital (46) of the Directive (EU) 2018/851 states: "Losses in weight of materials or substances due to physical or chemical transformation processes inherent in the recycling operation whereby waste materials are actually reprocessed into products, materials or substances should not be deducted from the weight of the waste reported as recycled.". We propose that inherent losses are technology neutral and linked to the type of waste material rather than the specific recycling technology. For example, for bio-waste inherent losses could be equal to, at most, the maximum amount of material that could be degraded according to EN 13432. Do you agree?
7. Our understanding is that compostable plastic in compost is a carbon (cellulose) source that acts as a soil improver. Do you agree?
8. Could you identify any relevant recycling process affected by the changes proposed on the rules (apart from composting and anaerobic digestion)?
9. The changes proposed to the calculation rules, if implemented in the EC legislation, will likely have impacts. Do you think that the list of impacts reported is comprehensive?
10. Is there any other relevant issue in relation to the report you would like to flag?

Annex 4. Calculation framework

A calculation framework is available separately to this report. The calculator may be found at: <https://publications.jrc.ec.europa.eu/repository/handle/JRC131531>.

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