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Quantifying recycling quality in Europe

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List of Abbreviations

AD	Anaerobic Digestion
DRS	Deposit Return Scheme
EBC	Economic Boundary Conditions
ECN	European Compost Network
EEA	European Environment Agency
EI	Environmental Impact
EOL-RR	End-Of-Life Recycling Rate
EU	European Union
F2F	Fibre-to-fibre
GWP	Global Warming Potential
HORECA	Hotels, Restaurants and Catering
IV	Intrinsic viscosity
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LTUO	Long-Term in-Use Occupation
MBT	Mechanical and Biological Treatment
MSW	Municipal Solid Waste
PET	Polyethylene terephthalate
TSP	Total Substitution Potential
TSS	Technical Suitability for Substitution
VCM	Value Chain Model
Wm	Market Weight

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1 INTRODUCTION

The purpose of the project “Quantifying recycling quality in Europe”, launched by the European Environment Agency (EEA), is to develop a suitable and readily operational model to assess the quality of recycling for a given material in the present and recent past, and to test its application on a set of representative waste material fractions in the European Union (EU).

Several scientific and technical publications have emphasised the need for high-quality recycling. Tonini et al. (2022) highlighted that the definition of recycling quality remains unclear, and a framework to implement such quality considerations in the recycling process is lacking.

In 2020, the JRC defined the ‘quality of recycling’ for sorting and recycling plants for mixed household packaging waste in the EU to understand which factors impact the quality and quantity of recycling outputs (Grant et al., 2020): “*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 to maximise their potential to be re-used in the circular economy*”.

Recycling waste is an essential part of transitioning to a circular Europe, where waste is re-introduced to the economy, thereby reducing the dependency on virgin resources. Currently, most recycling fails to keep the materials at their highest value. “High-quality recycling” is not only a reflection of the technical aspects of the recyclate – the output material of the recycling process – but also the environmental impact associated with transforming waste into recyclates and the new secondary raw material’s capacity to replace its virgin counterpart.

The project's objective has thus been to develop a practical model for quantifying the recycling quality, tested on three waste material fractions in the EU. This model will then potentially be used to quantify the recycling quality for other waste material fractions and for the years to come, ultimately allowing the EEA to monitor the development of the recycling systems in the EU. The model can help identify hotspots in recycling chains, i.e., when or what in the process negatively impacts the quality of the processes and the outputs. The model can also help decision-makers at the EU and Member-state levels to tailor more effective measures to improve circularity for each material.

2 METHODOLOGY

A methodology with four activities was defined to arrive at the operational metric (Figure 1):

1. Definition of assessment scope;
2. Identification and assessment of existing frameworks for assessing the quality of recycling;
3. Development and testing of quality assessment framework;
4. Validation and discussion of the proposed framework with external experts.

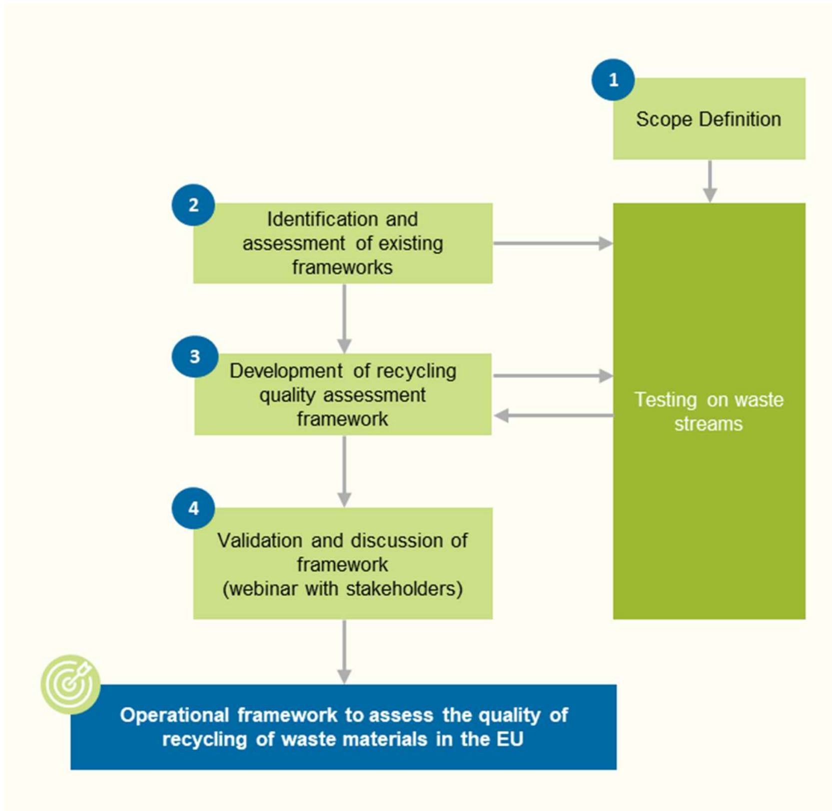

Figure 1. Methodology for development of a metric for quantifying recycling quality in Europe

2.1 Definition of the assessment scope

The assessment is focused on post-consumer waste recycling systems (pathways) implemented in the EU, with a supplementary aim of providing a historical assessment of the evolution of the quality of recycling since 2010.

The methodology is designed to be applied to waste material fractions instead of waste streams (e.g. mixed plastics). In the case of complex waste streams, the different fractions are sorted and subsequently sent to recycling processes, thus disaggregating into multiple recycling pathways. For example, in the case of construction and demolition waste (CDW), the model should consider the stages of collection and sorting that are common for all material fractions included in the stream. Then, it should look into the specific recycling processes associated with a particular material fraction. Potential further research should be undertaken to ensure the development of the model to allow the assessment of complex waste streams.

2.2 Identification and assessment of existing frameworks for assessing the quality of recycling

In 2023, a framework proposal for exploring how to revise and extend the definition of quality of recycling and making it quantifiable was developed by the JRC (Caro et al., 2023). Figure 2 shows a schematic overview of JRC's proposed framework for the quality of recycling with three main dimensions, namely the Total Substitution Potential (TSP), the Long-Term in-Use Occupation (LTUO) and the Environmental Impact (EI), all of which contribute to the definition of quality of recycling. According to the framework, the higher the TSP and LTUO and the lower the EI, the higher the recycling quality.

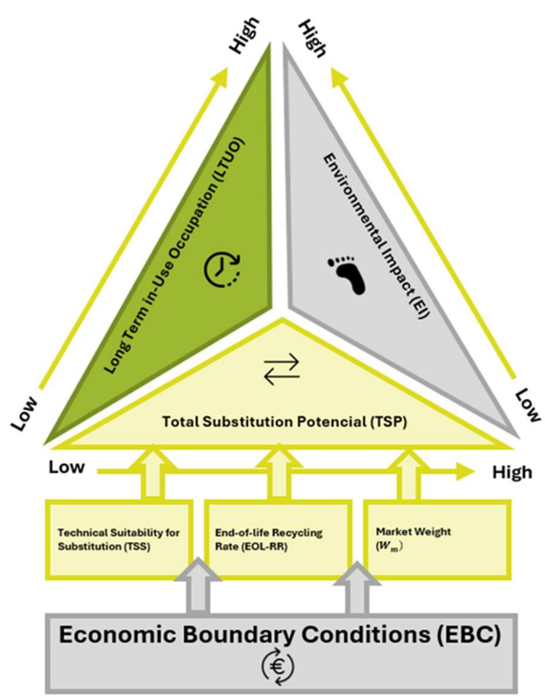

Figure 2. Schematic representation of the proposed quality framework for recycling

The present project carried out a deep critical assessment of the operability and adequacy of the JRC framework to calculate the quality of recycling of materials in the EU over a time period of 10 to 15 years. This analysis involved the application of the framework in one of the waste streams assessed in this study – PET packaging – to assess data availability.

Key aspects of the analysis include:

- The three dimensions of the JRC framework are relevant in assessing recycling quality.
- The framework was developed from a research perspective, with a high level of complexity detail and a need for comprehensive data, resulting in a lower level of operability.
- The framework was developed as a one-shot assessment of recycling pathways, with the primary goal of directly comparing different technologies' scores for the same material. It does not aim to do a temporal assessment of the evolution of the quality of assessment.

Regarding the mechanics of the JRC framework, Table 1 presents the main points identified in the assessment.

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Table 1. Summary of analysis of operability and adequacy of the JRC framework

Dimension	Parameter	Conclusions on operability and adequacy of framework
Total Substitution Potential (TSP): It represents to which extent the analysed material can replace or have the same function as the primary material. This includes not only the technical aspects but also volume/scale, i.e. if the analysed material can replace significant quantities of what is currently used as the primary material.	**Technical Suitability for Substitution (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*	A 0-1 indicator might not be the most adequate approach as this indicator has to be used in the LCA in dimension EI, for determining the avoided impacts associated with the displacement of virgin material.
	Market Weight (Wm): Represents the market share of a given application *i* (e.g., bottles) in the market of the corresponding material j (e.g., PET).	Parameter is relevant and mathematical basis is straightforward. It should be calculated annually.
	End-Of-Life Recycling Rate (EOL-RR): Focuses on how efficient recycling industries and recycling routes in the EU are - captures the amount of secondary material *j* recovered and functionally recycled at end-of-life compared to the overall waste quantities generated.	Parameter is relevant and mathematical basis is straightforward. It should be calculated annually.
	Economic Boundary Conditions (EBC): Represents 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, and that new technologies will initially have a lower economic feasibility.	The parameter should not influence 'quality of recycling' definition, as it only represents the economic viability, which itself might depend on scale, maturity, among other aspects. This parameter should be removed or analysed outside the equation.
	Overall: The TSP of a material *j* in a particular recycling pathway considers all applications in the market, i.e. it is the sum of the substitution potentials (SP). However, the other two dimensions, LTUO and EI, are specific to a particular material *j* and application *i*. Considering only the SP of a specific application would be more consistent with the methodology of the remaining dimensions.	
Long Term in-Use Occupation (LTUO): Capacity of the recycling chain to keep the intrinsic value or characteristics of the material in the economy, either by producing long-lasting products or products that can again be recycled	Lifespan of applications and recyclability	Recyclates can be used for many different applications, with very different lifespans and recyclability. According to experience and research, this data does not exist. The parameter should be included but with focus on the immediate output (and not the ultimate application) and be based on a qualitative assessment.
Environmental Impacts (EI): Represents the full life cycle impacts of a specific recycling chain.	-	This important parameter determines the environmental impacts associated with recycling pathways and should be included. It is a most complex dimension which requires a systematic and harmonised approach.

2.3 Development and testing of quality assessment metric

The new metric presented in this report builds on the JRC framework but is more oriented towards the different stages of the recycling value chain and designed to facilitate the calculation of the quality of the recycling of materials using available data.

The metric qA tested on three waste material fractions for which recycling technologies exist and are widely applied in the EU: polyethylene terephthalate (PET) packaging, municipal bio-waste, and post-consumer textiles. These materials were chosen based on three criteria: (i) scale/volume, representing the significance of these waste material fractions in the total waste production; (ii) potential increase, representing how these fractions are expected to increase in the next five to fifteen years; and (iii) environmental significance, i.e. the expected environmental impacts of current waste management practices. All three materials have two or more recycling technologies applied in the EU, which presented diverse challenges in the testing phase due to data availability and maturity of recycling pathways and processes.

Table 2. Rationale for selection of analysed waste material fractions

Analysed waste material fraction	Rationale
PET packaging	PET packaging is one of the better characterised waste material fractions. This allowed to test the framework on a well determined example (which was also used as a demonstrative example in the JRC report making it optimal for testing the metrics ability to operationalise the JRC framework). Two collection systems are analysed, separate household collection and deposit-and-return, and two recycling options, for food/ beverage packaging, and for polyester fibres for textiles, respectively. While not the most significant fraction in terms of volume, PET packaging has significant economic value and therefore differs from the other analysed materials.
Municipal bio-waste	Bio-waste is characterised by a big volume and high maturity of recycling operations. Bio-waste is one of the key waste streams in Europe, with many countries already having a dedicated collection system for food and green waste. And given the push by the Waste Framework Directive, separate collection of biowaste is expected to increase. When landfilled, biowaste generates high greenhouse gas emissions, which can be mitigated or avoided through two recycling pathways, namely composting or anaerobic digestion facilities, representing an important environmental significance.
Post-consumer textiles	Post-consumer textile waste has a significant volume but is still mostly separately collected for reuse purposes; a significant quantity ends up being landfilled or incinerated. With the establishment of mandatory post-consumer textile waste collection in all Member States from 2025, this fraction is expected to substantially increase. Today, textiles are mostly recycled mechanically with the majority resulting in non-woven applications and a small part in new textiles (Fibre-to-Fibre).

2.4 Validation and discussion of the proposed framework with external experts

An online webinar was organised on January 26th, 2024, to present and validate the developed recycling quality assessment framework. This webinar gathered external experts from the three sectors (PET, bio and textiles) and other stakeholders from policy and industry. This validation aimed to assess (i) whether the proposed framework addresses all the necessary aspects for assessing the quality of recycling, (ii) the quality and representativeness of used data in the three analysed case studies and (iii) the weighting of the three dimensions proposed (detailed in section 3).

A summary of the webinar and the stakeholders' contributions are presented in Annex I.

3 THE VALUE CHAIN MODEL – A METRIC TO ASSESS THE QUALITY OF RECYCLING

The developed metric to assess the quality of recycling, hereafter referred to as the Value Chain Model (VCM), is built on the framework proposed by the JRC.

3.1 The conceptual design

The VCM has two levels of assessment (Figure 3):

a) The assessment of recycling quality at the pathway level allows the comparison of quality between different recycling pathways for the same material fraction but does not consider the combined share and quality of the pathways in relation to the total amount of generated waste of the fraction.

b) The assessment at the system level - such as at the EU level or at country level - for a particular material fraction provides a score for the pathways combined in relation to the total amount of generated waste.

Figure 3. Conceptual design of Value Chain Model

A recycling pathway is the recycling system defined by the combination of:

- the waste collection system,
- the sorting process,
- the recycling technologies and
- the application of the recyclate.

The quality of recycling of a pathway (QRP_i) is characterised by the efficiency of the recycling system (E_{RPi}), the preservation of the material functionality in the economy ($Loop_{RPi}$) and its environmental

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performance (EI_{RPi}). These properties are determined at the recycling pathway level (RP_i). They are expected to be relatively constant and valid for 5 to 10 years if no significant technological disruptions occur.

To evaluate the quality of recycling of a material at the system level, such as at a country level or EU level, it is necessary to analyse a set of recycling pathways that are representative of the market, taking into account their market weights. As a rule of thumb, it was considered that to evaluate the recycling quality at the system level, the set of recycling pathways should cover at least 90% of the recycling market of a specific waste material fraction. Only recycling pathways that can be applied to the material fraction in the study can be included (e.g. anaerobic digestion cannot be implemented for garden waste). This market data is dynamic, and it is intended to reflect the changes in recycling in the system over time (e.g., the expected increase of the weight of PET bottle collection through Deposit Return Schemes in the upcoming years).

It is worth noting that although the VCM, in principle, enables the comparison of quality of recycling between different waste material fractions, e.g. glass and plastics, it is not intended to be used for assessing the overall preferability of different materials. Such a judgement should be based on a broader range of factors not included in the model.

Unfolded and folded approach

The VCM is characterised by its implementation flexibility in accordance with data availability (Figure 4). The complete model ('unfolded') enables the calculation to consider specific data on market weight, efficiency, and environmental performance for each value chain stage, e.g., the environmental performance of different collection systems. This makes it possible to conduct a hotspot analysis of the pathway, identifying where the aspects significantly impacting the final quality score stem from. If there is insufficient data for one or some of the value chain steps, the model can be 'folded' into a more straightforward version, which, as a minimum, must consider the following set of parameters for characterising the quality of recycling:

- Global efficiency of recycling pathway (avoided primary material in application *i* in relation to the collected waste material fraction);
- Environmental performance of recycling pathway (LCA studies typically provide total results for the entire pathway, including collection);
- Environmental benefits from use in application *i* (the benefits associated with the avoided production of virgin material, considering technical substitution ratio);
- Loop potential of application *i*, as described in section 3.2.1.

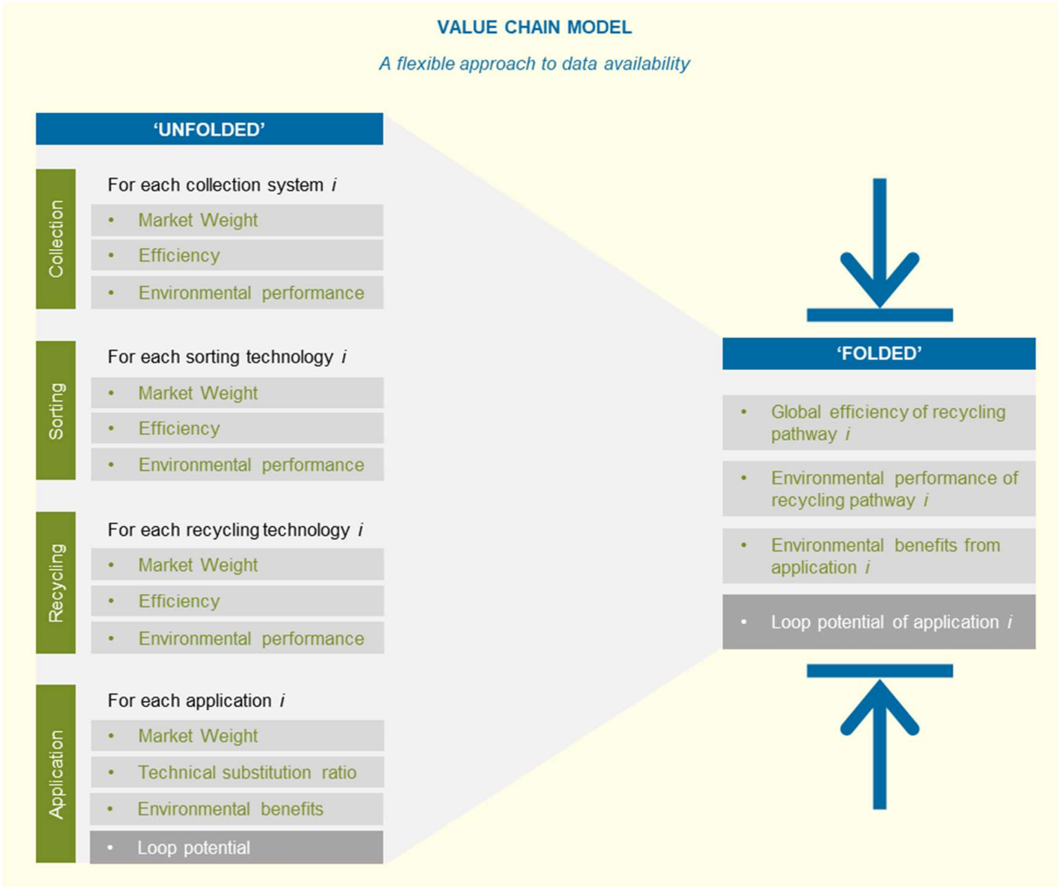

Figure 4. Representation of flexibility of implementation of the VCM

The methodology for implementing the VCM for any particular waste material fraction is represented in Figure 5. It can be applied to the two possible levels of assessment – system level and pathway level – albeit with different scopes of assessment (see 3.1.).

For the system level, the first step in implementing the model is identifying the most relevant recycling pathways in the EU market for a material fraction. A literature and market data review needs to be carried out for each analysed material waste fraction to obtain knowledge on the status of recycling in Europe regarding collection systems, sorting and recycling processes, and applications of the recyclate. A set of criteria needs to be considered:

- Scale: recycling pathways should be implemented commercially to be considered.
- Distinctiveness: there should be a sufficient degree of distinctiveness between pathways, whether in terms of efficiency or environmental performance.
- Data availability: If no data is available for modelling a particular recycling pathway, then it is impossible to determine its recycling quality.

There are, however, exceptions to these criteria if the goal of the quality of recycling assessment is to analyse different pathways for policy-making purposes or to carry out a hotspot analysis. Then it might be relevant to look into pathways which are not yet commercially implemented or that do not have available data. This analysis allows to identify data gaps and identify critical hotspots which might need to be taken into consideration in future policy making.

The remaining steps of the methodology for implementing the VCM are detailed in section 3.2, which describes the in-depth mechanics of the model, namely the parameters and calculation methods.

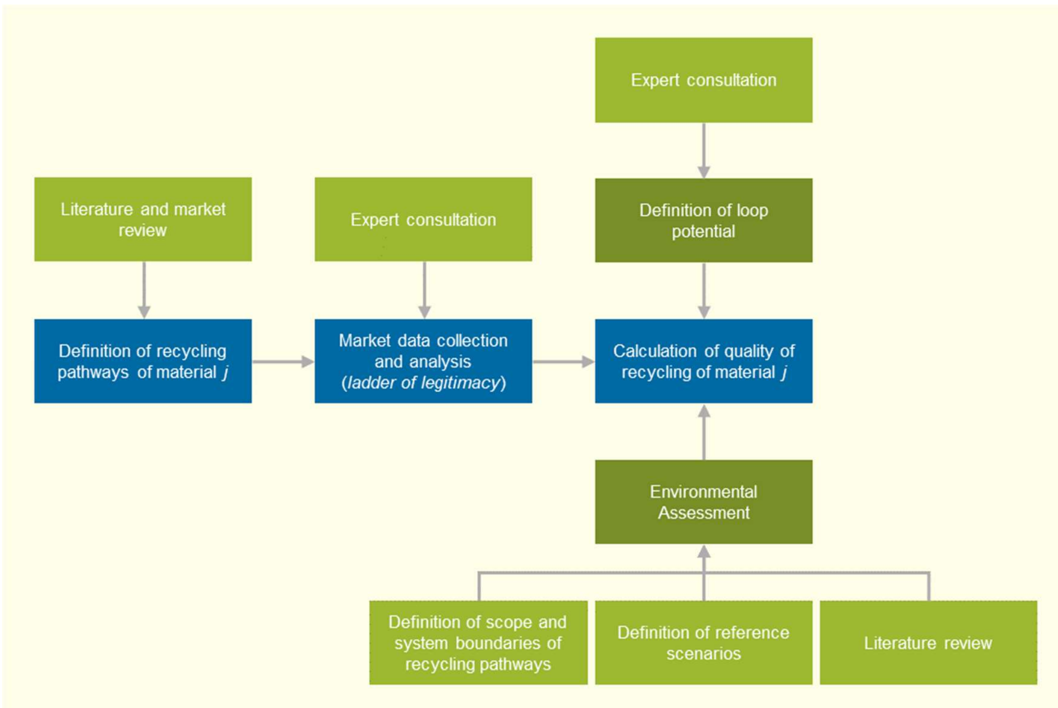

Figure 5. Methodology of implementation of the Value Chain Model

3.2 In-depth mechanics of the model

3.2.1 Quality of recycling pathway *i*

The quality of recycling of a particular recycling pathway is a function of three dimensions, as described in Equation 1.

$$QR_{RPi} = f(E_{RPi}, Loop_{RPi}, EI_{RPi}) \qquad \text{Equation 1}$$

Each of these dimensions are described in the following sub-sections:

3.2.1.1 Efficiency

This parameter characterises the efficiency of a given recycling pathway, i.e., the recycling system which includes the waste collection system, sorting and recycling technologies and the application of the recycled material. Efficiency refers to the pathway's ability to capture and move as much of the waste material as possible through the pathway. The efficiency of a recycling pathway i is given by Equation 2.

$$E_{RPi} = Ecoll_i \times Esort_i \times Erec_i \times TSR_i \quad \text{Equation 2}$$

$$E_{RPi} \in [0;1]$$

Where, for each recycling pathway i,

- The efficiency of a collection system ($Ecoll_i$) is given by Equation 3.

$$Ecoll_i = \frac{Quantity\ of\ collected\ waste\ sent\ to\ sorting}{Quantity\ of\ waste\ generated} \quad \text{Equation 3}$$

- The efficiency of a sorting technology ($Esort_i$) is given by Equation 4.

$$Esort_i = \frac{Quantity\ of\ sorted\ material\ (output)}{Quantity\ of\ collected\ waste\ sent\ to\ sorting} \quad \text{Equation 4}$$

- The efficiency of a recycling technology ($Erec_i$) is given by Equation 5.

$$Erec_i = \frac{Quantity\ of\ produced\ secondary\ material}{Quantity\ of\ sorted\ material\ (output)} \quad \text{Equation 5}$$

- The technical substitution ratio of an application (TSR_i) is given by Equation 6.

$$TSR_i = \frac{Quantity\ of\ substituted\ primary\ material}{Quantity\ of\ produced\ secondary\ material} \quad \text{Equation 6}$$

The efficiency of the recycling pathway contributes to the quality of recycling metric as it accounts for the losses in the system in all stages of the material processing, from collection to application.

The interpretation of the collection efficiency can be more challenging. In the cases where there is only one collection system for the waste material fraction, the collection efficiency can be considered as the percentage of generated waste which is separately collected for recycling, thus reflecting the

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capture rate for the material and accounting for the losses in the system (e.g. to incineration, landfill). This indicator becomes more intricate when there are multiple collection systems for the same waste material fraction (there can be one single collection system but two recycling pathways for a particular material fraction, as is the case in the textiles and bio-waste case studies). In this case, the collected and generated waste refer to the limited scope of the pathways. This means that the waste generated in recycling pathway i should be interpreted as the waste generated within the scope of the collection system of recycling pathway i that can potentially be collected. In the case of PET packaging, it is fairly intuitive that the potential for collection within the system of a Deposit Return Scheme will be the quantity of PET bottles put on the market (see 4.3.1).

As a rule, in the case of collection systems that co-occur in the same geography, it will be necessary to analyse individual efficiencies based on data from countries or regions where one model is predominant.

It is important to reinforce that the efficiencies are related to the specific recycling pathways. In the case of a particular waste material fraction having only one recycling pathway, then the efficiency represents the global efficiency of the material at the system level.

The capacity of the secondary material to replace primary material is also included in the efficiency dimension through the TSR. This indicator quantifies how much primary material is substituted by the secondary material to achieve the same function and therefore represents the standard market approach at a given time. Applications with high recycled content are expected to need more secondary material for the same technical properties, such as resistance. For example, plastic bags with high recycled content are usually thicker to compensate the polymer chain breakage from heat cycles. However, TSR does not consider explicitly the suitability of a recycling chain for a specific application or the technical limitations (e.g., rPET can replace up to 75% of virgin PET in plastic bottles), but generally, low TSR is usually associated with a low suitability. In the next section, the loop potential dimension is proposed to capture this suitability aspect, among other aspects.

The data for calculating the efficiency parameters should be collected from the industry or peer-reviewed literature.

3.2.1.2 Loop potential

This dimension aims to evaluate how the recycling pathway maintains the material fraction's properties and functionality for subsequent recycling. A straightforward approach is proposed where a qualitative scale with six discrete values (levels) is attributed to a particular application (Equation 7). The scale is set between zero, where the material value is lost after its application, and one, where the material is maintained in the economy without decreasing functionality (Table 3). The scale also includes several examples from different sectors better to clarify the meaning of each loop potential level. This parameter should be validated through expert consultation. It is important to note that this scale was defined according to the current state-of-the-art recycling technologies. The presented scores can improve significantly through policy measures and technology development and, therefore, should not be considered constant in prospective analysis. This dimension also enables the analysis of potential policy measures and how these impact the quality of recycling.

$$Loop_{RPi} \in [0; 0.2; 0.4; 0.6; 0.8; 1] \qquad \text{Equation 7}$$

The loop potential dimension does not consider the lifespan of applications to be a relevant factor when assessing the quality of recycling. When considering two practical examples of PET applications, for example, bottles (which have shorter lifespans but can be recycled multiple times), and textiles (which have longer lifespans (not considering fast fashion) but cannot be recycled after their use), it is not obvious that one application is better than the other.

Table 3. Loop potential dimension – Maintenance of material properties and function

Categories	Description	Examples
0	The material value is lost after its application	- *(this is not applied as any recycling pathway should involve some degree of recycling)*
0.2	The material value cannot be recovered after one cycle	- *textile fibres in cement* - *recycling MDF into chipboard*
0.4	Material can loop but with significant losses in functionality after a small number of cycles	- *fibres in non-woven applications* - *PET as polyester fibres* - *'recycling tyres for rubber flooring/playgrounds* - *recycling bio-waste into digestate*
0.6	Several cycles but with decreasing functionality or properties (i.e. not replacing primary material)	- *fibre-to-fibre mechanical recycling* - *recycling wood into chipboard* - *closed-loop paper recycling* - *recycling bio-waste into compost*
0.8	There is a slight decrease in its functionality, but it is possible to continue to cycle the materials	- *closed-loop PET packaging* - *steel in long-lasting products (containers, beams)*
1	The material is maintained in the economy without any decrease in functionality	- *chemical recycling of plastics* - *closed-loop recycling of aluminium cans* - *closed-loop recycling of glass bottles*

3.2.1.3 Environmental impact

The third dimension of the framework is the environmental impact (EI) analysis, which (as pointed out by Caro et al., 2023) should follow the principles described by the ISO 14040:2006 framework for Life Cycle Assessment.

Developing an LCA study can be significantly time- and resource-consuming. Given the scope of the assessment, it is recommended to develop a literature review of the existing body of work on recycling pathways in the field of LCA. However, the results of any LCA study cannot be analysed per se; one must also consider the system boundaries, functional unit, and methodological approach. The LCA standards leave room for interpretation, resulting in differences in the methodological choices and, consequently, in the assessment results. Therefore, a critical literature analysis is required to ensure the methodological choices and scope of assessment are aligned with the objectives of the quality of recycling assessment.

One significant LCA methodological problem is the allocation of environmental burdens in recycling materials and energy. It is not always straightforward to analyse the environmental impact of multi-functional processes, which often exist in waste management systems (Rigamonti et al., 2020). According to Laurent et al. (2014), the multi-functionality issues in waste management studies are solved mainly by applying system expansion. When assessing secondary resource recovery, the substitution or avoided burden approach is regarded as a particular or simplified type of system expansion, where the environmental impact of the primary resource that the secondary resource can displace equivalently is subtracted from the impact of the multi-functional system (Rigamonti et al., 2020). Calculating these avoided impacts is a critical aspect of LCA, as it greatly influences the results.

The vast majority of waste management LCA studies have so far assumed that one unit of recycled material substitutes one unit of virgin material; however, few recyclates are of a high enough technical quality to do so, which then implies an over- or underestimation of the real benefit of the recycling activities, which, in turn, could result in misleading conclusions (Rigamonti et al., 2020).

Therefore, defining a substitution factor is a critical and necessary aspect of the EI dimension. In this framework, this indicator is given by TSR_i and is considered in the Efficiency dimension. Rigamonti (Rigamonti et al., 2020) concluded that technical substitutability coefficient calculations have many forms and use different approaches. Harmonisation is thus necessary to make the results of LCA studies more reliable.

The following recommendations are aimed at improving the analysis of the EI dimension:

- When selecting LCA studies to develop the EI analysis, preference should be given to studies with more recent data, geographical scope focused on the EU, system boundaries that include the entire recycling pathway(s) and a detailed and transparent description of assumptions and methodology.
- Data from LCA studies concerning specific value chain stages can be used. However, the practitioner must ensure the harmonisation of methodological approaches between studies.
- When resorting to LCA studies focusing on the entire value chain, i.e., at the system level, it is essential to ensure that all stages from collection until recycling are considered. If not, the missing data must be obtained from other sources.
- LCI data should be considered for 10-year periods as one can assume no significant changes in the existing recycling technologies exist. The energy consumption mix, which might be relevant in recycling processes, might vary annually, particularly the electricity mix. It could be relevant to update data on the electricity consumption within the recycling processes according to the variations of the European electricity production mix. This might be a relevant parameter as the European energy mix is expected to be cleaner in the upcoming years, which implies that the recycling options will appear increasingly more beneficial when compared with energy recovery (Garcia-Gutierrez et al., 2023). However, this effect might be annulled by the expected cleaner production processes for primary material production.
- To monitor progress in the quality of recycling in Europe, regular checking of new LCA studies that can better address the objectives of assessing the environmental performance of the recycling pathways for a given material should be carried out.

- Multiple recycling and cascading cycles, i.e. subsequent uses of the same material after the first recycling loop, are not to be considered as they require a shift in scope and functional unit (Garcia-Gutierrez et al., 2023).

Using GWP as the chosen impact category

In LCA, the classification and characterisation of an impact into its relevant impact category are mandatory elements according to ISO 14040/14044. The LCI results are converted to standard units aggregated within the same impact category. Several LCIA methods for classifying and characterising each impact into its relevant impact category have been developed for regional and global use (Iqbal et al., 2020).

The IPCC 2013 is the second most used impact assessment method by LCA practitioners (14.44%), the first being ReCiPe, used by 18.89% of practitioners, which implies either a set of several midpoint indicators or endpoint ones (i.point systems, 2018). The same study found that aspects that make impact assessment methods popular mainly relate to calculating Global Warming Potential (GWP) impacts. This is the impact category that is present in most methods (ReCiPe, IPCC, ILCD, Impact 2002+, CML 2012, etc.), represented in the same standard unit (kg CO_2e).

In a study focused on life cycle impact assessment methods application, Koch et al. (2023) identified GWP as the midpoint impact category existent in most methods and the one that showed consistent interpretations for the scenario analysis from different impact assessment methods among the most relevant categories of:

- global warming potential (GWP, 12 LCIA methods totalling 48 subcategories),
- eutrophication potential (EP, 9 LCIA methods totalling 18 subcategories),
- and water assessment (WA, 10 LCIA methods totalling 26 subcategories).

In this study, it is possible to verify that different assessment methods consistently calculate GWP impact and present a constant unit (kg CO_2e), even if some methods use factors from different IPCC reports (2001, 2007, 2013). On the contrary, eutrophication, for instance, is calculated based on distinct environmental processes and resulting in different units of measure (e.g., kg Peq.; kg PO4eq., kg NO4eq., kg Neq., a mole of Neq., m^2 UES).

Moreover, GWP is the only indicator subject to the consideration of standard metrics under a United Nations Framework Convention (SBSTA 36 – the thirty-sixth session of the Subsidiary Body for Scientific and Technological Advice) (UNFCCC, 2023). For this reason, this framework proposes to consider only the results for the GWP impact category in calculating the EI dimension.

However, this can be a limitation of the proposed model, as climate change (CO_2 emissions) may not be the most relevant impact category for specific waste material fractions. For instance, this is the case for bio-waste recycling, which involves soil application of digested or composted material. Environmental regulations are put in place to limit the application of these materials in the soil to limit the level of contaminants, including heavy metals. In the future, the proposed model could be expanded to include other impact categories, accentuating data quality and comparability issues.

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To be considered in scoring the quality of the recycling pathway, the EI dimension must be normalised to result in a value between zero and one, in the same way as the Efficiency and Loop Potential dimensions. To achieve this, a minimum (r_{min}) and maximum (r_{max}) value must be established following the worst- and best-case scenarios in terms of environmental impacts, respectively. These need to be defined and calculated along with the environmental performance of the recycling pathways. For all materials, the best-case scenario should ideally be a recycling process with no environmental impacts that can replace virgin material in a ratio of 1:1. This reference scenario will allow us to frame the results from all possible applications. The worst-case scenario should vary according to the material. The environmental performances of the recycling pathways are normalised to values between zero and one in relation to both reference scenarios.

Once these reference scenarios are defined and calculated, it is possible to proceed to the normalisation using Equation 8.

$$r_{normalized} = \frac{r - r_{min}}{r_{max} - r_{min}} \qquad \text{Equation 8}$$

3.2.1.4 Scoring model

The framework's three dimensions are considered in a multi-dimensional rectangular hyperbola to determine the highest quality recycling pathway, which returns an individual score. The theoretical optimal quality of recycling is one where the Environmental impact, the Efficiency and the Loop potential dimensions would be one. If any of these dimensions tends to zero, then the quality of recycling score also tends to zero.

In the proposed framework, a scoring method is defined for calculating the final score for the quality of recycling of a unit pathway i, in accordance with Equation 9. This equation considers specific weights for each dimension (represented by *wi)*, allowing for different perspectives based on user preferences.

$$QR_{RPi} = \frac{1}{\frac{w1}{E_{RPi}} + \frac{w2}{Loop_{RPi}} + \frac{w3}{EI_{RPi}}} \qquad \text{Equation 9}$$

$$w1 + w2 + w3 = 1$$
$$w1,\ w2,\ w3 \in\ [0;1]$$
$$E_{RPi},\ Loop_{RPi},\ EI_{RPi} \in\ [0;1]$$

This formula penalises deficient performance in one parameter, thus resulting in less risk of having “compensation” between parameters, as seen in the sensitivity analysis in Table 4.

Table 4. Results of sensitivity analysis for scoring approach

$w1$	$w2$	$w3$	$\sum w_i$

0.33	0.33	0.33	1

E_{RPi}	$Loop_{RPi}$	EI_{RPi}	QR_{RPi}
0.10	0.20	0.10	0.12
0.80	0.10	0.80	0.24
0.30	0.30	0.30	0.30
1.00	1.00	0.10	0.25
0.50	0.90	0.90	0.71
0.90	0.50	0.50	0.59
1.00	1.00	0.01	0.03

Weights must be applied to each of the three dimensions ($w1, w2, w3$) to obtain a final score for the quality of recycling of a particular pathway. The weights presented in Table 5 and considered in the model were defined as the result of the stakeholder consultation carried out during the project's webinar, which was organised on January 26th, 2024 (see Annex I). This weighting approach privileges the end goal from a policy-making perspective: to increase the environmental benefits associated with recycling.

Table 5. Weights considered for each dimension in the VCM

E_{RPi}	$Loop_{RPi}$	EI_{RPi}	$\sum w_i$
$w1$	$w2$	$w3$	
0.30	0.30	0.40	1

3.2.2 Quality of recycling of material fraction *j* at system level

The quality of recycling of a material fraction j at the system level (e.g. EU level) is calculated considering the quality of the individual recycling pathways (QRP_i) and their respective market weights (W_{RPi}), in accordance with Equation 10.

$$QR_{material\ fraction} = \sum (W_{RPi} \times QR_{RPi})$$ Equation 10

When calculating recycling quality at the EU level, market data is required to determine the weight of the different pathways (W_{RPi}). This data is dynamic and should demonstrate any system's recycling market over time.

The weight of a recycling pathway i is given by Equation 11.

$$W_{RPi} = Wcoll_i \times Wsort_i \times Wrec_i \times Wapp_i \qquad \text{Equation 11}$$

Where,

- The weight of the collection system ($Wcoll_i$) is given by Equation 12.

$$Wcoll_i = \frac{Quantity\ of\ collected\ waste\ in\ system\ i}{Quantity\ of\ generated\ waste} \qquad \text{Equation 12}$$

- The weight of the sorting technology ($Wsort_i$) is given by Equation 13.

$$Wsort_i = \frac{Quantity\ of\ sorted\ material\ in\ technology\ i}{Quantity\ of\ collected\ waste} \qquad \text{Equation 13}$$

- The weight of the recycling technology ($Wrec_i$) is given by Equation 14.

$$Wrec_i = \frac{Quantity\ of\ waste\ recycled\ in\ technology\ i}{Quantity\ of\ collected\ waste\ sent\ to\ recycling} \qquad \text{Equation 14}$$

- The weight of the application ($Wapp_i$) is given by Equation 15.

$$Wapp_i = \frac{Quantity\ of\ waste\ used\ in\ application\ i}{Quantity\ of\ recycled\ material} \qquad \text{Equation 15}$$

In terms of mass balance, the recycling pathways must then obey the following condition:

$$W_{material\ fraction} = \sum W_{RPi} = 1 \qquad \text{Equation 16}$$

The simple interpretation is that the sum of the market weight of all recycling pathways should equal one.

3.3 Data availability and quality

The viability of the proposed value chain model, as presented in section 3, has been tested by applying it to selected waste material fractions, namely PET packaging, municipal biowaste, and post-consumer textiles.

The first step of this process was to identify and collect data on each value chain step for each of the selected waste material fractions to ensure a focused and comprehensive examination of the data foundation for each test case. The methods utilised to collect and identify relevant data have included desk research and expert consultations with relevant associations and stakeholders. Each data identification process was structured considering the following elements:

a) **Value Chain Model Alignment:** Data identification was structured according to critical steps in the value chain model, namely Collection, Sorting, Recycling, and Application, to systematically capture data that could be utilised to inform the entire recycling process.

b) **Time aspect:** Data identification was structured to consider a time aspect when searching for relevant data to assess the advancement in the quality of recycling in the EU over a relevant period.

c) **Ladder of legitimacy:** The identification and selection of relevant data has been guided by a ladder of legitimacy, establishing a clear hierarchy for prioritising data depending on the source type. Going from most favoured to least, sources have been prioritised based on their origin from 1) Official statistics from the European Statistical System or National Statistical Offices, or other relevant data produced by EU institutions, national or local authorities; 2) Peer-reviewed research articles, and reports; 3) Reports and surveys from the private sector in the form of e.g. trade associations; 4) other data from e.g. news reports and articles.

d) **Data Type Prioritization:** Data were filtered based on data type, with a preference for dynamic data (i.e. data that is subject to changes and updates over time) over static data (i.e. fixed data that remains constant over time) to achieve a real-time and dynamic data foundation, which can be utilised continuously in the value chain model.

e) **Governance level prioritisation:** When identifying data, EU-level data has been prioritised in cases where this was available. In cases where EU-level data was unavailable, relevant state-level and local-level data were researched as an alternative to supplement the data foundation for the model.

A data table was created and utilised throughout the process to document and further structure the collection and identification of data. The first step was to meticulously record all search activity, including search place and search string, and findings, including the source type, keywords, geographical coverage, time relevance, whether dynamic or static, and whether the finding was relevant. The second step was to sort out the relevant data for each step. Furthermore, the table has also included continuous notes on the results of the following inquiries and actions (i.e., expert consultations and inquiries into available data from relevant stakeholders and associations), where no data was available through online sources. In the chapters referring to the model test on the different case studies, the search result is described, summarising the findings from the desk research or expert consultation.

As the second step in testing the viability of the proposed value chain model, the data identified and collected through the abovementioned methods and principles have been analysed to assess the robustness and adequacy of the data foundation and to what extent this data could be applied to the value chain model for each of the respective test waste material fractions. As such, in cases where available data has failed to inform the full extent of the parameters of the value chain model, the value chain model has been utilised in its folded form. Conversely, if the robustness and availability of relevant data analysed were deemed to be sufficient, the full parameters of the value chain model have been utilised on the respective waste material fraction.

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4 Test of Model on PET packaging

4.1 Introduction to the waste stream

Polyethylene terephthalate (PET) waste can have different origins and destinations, including bottles, trays, films and textiles as their main markets (Caro et al., 2023). PET waste can be recycled mechanically (based on extrusion), the most used technology, or chemically.

- Mechanical recycling aims to recover plastic waste via a process line that includes possible dismantling/disassembling, grinding, washing, separating, drying, re-granulating and compounding. The recyclates can be converted into new plastic products, often substituting virgin plastics.
- In the chemical recycling of plastic waste, the polymer chains are converted into oligomers, monomers or other basic chemicals (such as carbon monoxide, carbon dioxide, methane, and hydrogen) before further reprocessing into monomers/polymers. The process can be subdivided into depolymerisation, pyrolysis and gasification (Garcia-Gutierrez et al., 2023).

Two PET grades currently dominate the global market, i.e. fibre-grade PET and bottle-grade PET, differing mainly in molecular weight or intrinsic viscosity (IV), optical appearance and production recipes. Textile fibre-grade PET has an intrinsic viscosity between 0.55 and 0.67 dL/g, whereas bottle-grade PET appears ‘glass-clear’ in the amorphous state and has an intrinsic viscosity between 0.75 and 1.00 dL/g. Other PET grades are manufactured for packaging films, often standard grades with an intrinsic viscosity of 0.64 dL/g (Al-Sabagh et al., 2016).

The IV values for recycled PET (rPET) are substantially lower than for virgin PET material due to the chain degradation of the PET polymer. In bottle-to-bottle recycling, polycondensation steps are usually implemented to increase the IV, thus compensating for the degradation (Demets et al., 2021).

Fibre-grade PET generally has a lower degree of polymerisation than bottle-grade PET, and textiles are generally more complex than bottles. For PET textiles, the current physical recycling method can only lead to downcycling due to the gradually decreasing degree of polymerisation; also, the current chemical recycling methods are unable to achieve upcycling due to the difficulty of separating and purifying the depolymerised products (Peng et al., 2023).

4.2 Scope of assessment

Extensive research was carried out to define the representative recycling pathways of the current market for PET packaging in the EU:

1. **Collection:** Several collection systems are implemented in the EU with varying performances. PET packaging is mainly collected from households via door-to-door collection or bring-sites, mostly mixed with other plastics. In several Member States, Deposit Return Schemes (DRS) are implemented to collect beverage PET bottles. These collection systems are implemented in 10 European countries, nine of which have achieved sorted-for-recycling rates of 83% or higher (Eunomia, 2022). Since DRS is gaining scale at the European level and results in greater efficiency for PET bottle recovery, as it ensures a lower level of contamination, it should be considered in the scope assessment, along with the separate collection (SC) of mixed plastics.

2. **Sorting:** The sorting technologies used for PET packaging vary significantly across the EU and even within Member States and are dependent on the type of waste collection put in place. There is no available systematic data regarding sorting efficiencies for PET packaging in the EU and the representativeness of different sorting technologies. For this reason, sorting is not differentiated in the scope of assessment.

3. **Recycling:** There are two leading technologies for PET recycling: mechanical recycling and chemical recycling, the former being the most widely used (Chairat & Gheewala, 2023). Chemical recycling is beginning to be commercially implemented; however, there is no available data for its market representativeness. For this reason, only mechanical recycling was considered in the assessment. It is important to note that chemical recycling is expected to gain scale in the upcoming years and must be integrated into future assessments of recycling quality.

4. **Application:** The recycled high-viscosity PET can be used in high and low-viscosity applications, namely packaging and fibres, respectively (Al-Sabagh et al., 2016; Eunomia, 2022).

The identified processes were combined to define the most representative recycling pathways for high-viscosity PET (packaging). The below points describe the different steps of the recycling pathways (with abbreviation noted in parenthesis):

- Separate mixed plastics collection – Mechanical recycling – Packaging (SC-P2P);
- Separate mixed plastics collection – Mechanical recycling – Fibres (SC-P2F);
- DRS – Mechanical recycling – Packaging (DRS-P2P);
- DRS – Mechanical recycling – Fibres (DRS-P2F).

A simplification was made in implementing the proposed model in the PET case study, where bottles, trays, flexible packaging, and others are aggregated into a single packaging category. Actual recycling processes will split these materials into separate categories (e.g., trays usually have more contaminants, such as additives and barriers to reduce oxygen transfer). Still, there is a lack of data at this level. This demonstrates the trade-off between granularity and data availability that permeates this type of analysis.

This case study does not consider the production of low-viscosity PET used for fibre production, as the focus is on packaging. Only the use of rPET from packaging used in low-viscosity applications is considered.

Even though the project aimed to have a temporal scope of 10-15 years of assessment, the case study development demonstrated no publicly available data for that period. This was confirmed during the stakeholder consultation carried out in the project with stakeholders in the PET value chain: PET Core, Plastic Recyclers, PET Europe and Plastics Europe.

An extensive data search for the assessment of PET was carried out. Two data types, static and dynamic, were collected, as described in section 3.1. Dynamic data, which refers to market data (Figure 6), are mainly derived from a publication promoted by the associations Natural Mineral Waters Europe, Plastics Recyclers Europe, PETCORE Europe and UNESDA Soft Drinks Europe.

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The 'PET Market in Europe: State of Play' is published every two years. The first one was published in 2020 (referring to 2018) (Eunomia, 2020), the second one in 2022 (referring to 2020) (Eunomia, 2022), and the third one is expected to be published in 2024 (referring to 2022).

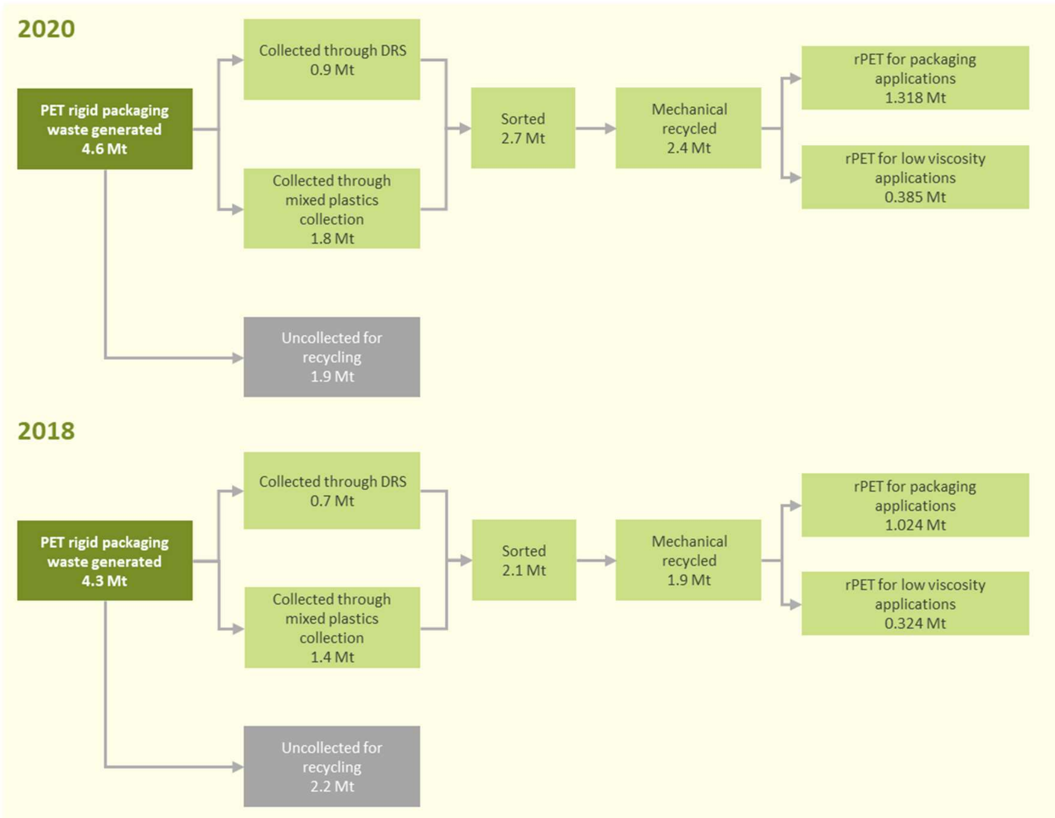

Figure 6. Market data for PET packaging used for the assessment

4.3 Application of quality of recycling metric

As already mentioned, the quality of the recycling of a given pathway is determined by the following dimensions: efficiency, loop potential and environmental impact (Equation 1). The implementation of the proposed metric is detailed in the following sections.

4.3.1 Efficiency

The efficiency of the recycling pathways (E_{RPi}) defined in the scope of assessment (section 4.2) was calculated according to Equation 2 (the sorting technology was not differentiated in the scope of assessment). Regarding the nature of the data, the recycling efficiency and technical substitution ratio are expected to remain static and valid for 5 to 10 years if no significant technological disruptions arise. Regarding collection and sorting efficiency, it is calculated based on the industry data and thus can be updated every two years. The parameters for the efficiency calculation are presented in Table 6, followed by a description of data availability and treatment.

Table 6. Parameters for the calculation of the efficiency dimension of the VCM for PET

Data nature	Parameter	Processes	2020	2018	Data source
Dynamic	Efficiency of collection and sorting	Separate collection	33%	21%	(Eunomia, 2020, 2022; Unesda, 2023; Zero Waste Europe, 2019)
		DRS	96%	90%	
Static	Recycling efficiency	Mechanical Recycling (P2P)	71%		(Gileno & Turci, 2021)
		Mechanical Recycling (P2F)	77%		
Static	Technical substitution ratio	Packaging	95%		(Roosen et al., 2023)
		Fibres	95%		

- **The efficiency of collection and sorting –** The efficiency of collection and sorting (collection and sorting system) was evaluated together. The collection rates and the European market shares of the collection systems (mixed plastics separate collection and DRS) were estimated to assess collection and sorting efficiency.

  For 2020, PET packaging collection and sorting efficiency was estimated at around 53%, resulting from 4.6 Mt of PET rigid packaging waste generated and from the 2.4 Mt of PET packaging collected and sorted (including bottles and trays) that enters recyclers. (Eunomia, 2022). It was also assumed that 35% of the total PET bottles were collected via DRS, while the remaining PET was collected through mixed plastics collection (Unesda, 2023). Based on the assumption that the average collection rate of PET bottles in European countries operating DRS is 96% (Zero Waste Europe, 2019), the efficiency of collection and sorting for mixed plastics collection was estimated at 33%.

  For 2018, a PET packaging collection and sorting efficiency of 44% was estimated, resulting from the 4.3 Mt of PET packaging waste generated and the 1.9 Mt of PET packaging collected and sorted (Eunomia, 2020). Due to insufficient data, the assumption that 35% of the total PET bottles were collected through DRS was maintained for 2018 (Unesda, 2023). Based on the assumption that the average collection rate for PET bottles in European countries with a DRS system is 90% for 2018 (Zero Waste Europe, 2019), the efficiency of collection and sorting for mixed plastic separate collection was estimated at 21%.

- **Recycling efficiency –** The efficiencies for the recycling processes were analysed for two scenarios: mechanical recycling of PET bottle-to-bottle and bottle-to-fibres. The indicator measures the efficiency of the recycling process itself, i.e. how much secondary material is produced with the input waste material fraction (Gileno & Turci, 2021).

- **Technical substitution ratio –** This parameter indicates how much virgin material can be replaced with secondary material while maintaining its technical properties. It is required in

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the calculation of environmental benefits associated with the substitution of virgin material. The technical substitution ratio for chemical recycling would be 1 (Roosen et al., 2023), as the polymer chains are converted to monomers of equal quality to the virgin material.

4.3.2 Loop potential

The application of rPET into the packaging was scored as 0.8, whereas the application of the fibres was classified as 0.4, according to the scale proposed in Table 3. It was considered that for the fibre application, there is a decrease in the potential for further recycling, as currently only a tiny share of textiles is recycled. In the case of packaging, if they are collected and maintained as packaging through closed-loop recycling, they are expected to be recycled through several loops. This dimension should be validated through expert elicitation.

4.3.3 Environmental performance

After a literature review of LCA studies, data from Gileno & Turci (2021) was considered to model the environmental impacts of two mechanical recycling processes for PET: bottle-to-bottle and bottle-to-fibre. The authors analysed the impacts of these processes, which included the transport of post-consumer PET bottle bales from concentration and compaction points to the recycling unit, as well as the production of the recycled material itself. The modelling of the waste collection stage was not considered in the study. To complement the results of Gileno & Turci (2021), data on the environmental performance of the two collection systems under analysis was required. Literature on the environmental assessment of the DRS collection system is scarce. Specific data regarding the collection of PET bottles through a DRS implemented in supermarkets and through kerbside collection from Simon et al. (2016) was used.

Additionally, the modelling of avoided impacts associated with substituting virgin material was not considered in Gileno & Turci (2021). Therefore, this analysis was conducted considering the values defined for the TSR of both applications and the impacts associated with virgin PET production (from the Ecoinvent database).

The literature review allowed defining the worst-case scenario as the incineration of PET without energy recovery. The best-case scenario was substituting virgin PET (1:1) in a recycling process with zero burden. These reference scenarios were modelled using data from the Ecoinvent database. As described in section 3.2.1.3, only the results for the Global Warming Potential impact category were considered and normalised to obtain the score for the EI dimension (Table 7).

Table 7. Results of environmental assessment of PET scenarios

Scenarios	Description	Value (kg CO_2eq/kg)	Normalized values	Data source
Best case scenario	Replacing 1:1 virgin PET with zero burden recycling process	-3.004	1	Ecoinvent data for PET granulate production
DRS-P2P	DRS collection system (supermarkets). B2B recycling process. Benefits with substitution of virgin material were estimated.	-1.84	0.81	(Gileno & Turci, 2021; Simon et al., 2016) Ecoinvent data for PET granulate production
SC-P2P	Kerbside separate collection. B2B recycling process. Benefits with substitution of virgin material were estimated.	-1.78	0.80	(Gileno & Turci, 2021; Simon et al., 2016) Ecoinvent data for PET granulate production
DRS-P2F	DRS collection system (supermarkets). B2F recycling process. Benefits with substitution of virgin material were estimated.	-1.61	0.77	(Gileno & Turci, 2021; Simon et al., 2016) Ecoinvent data for PET granulate production
SC-P2F	Kerbside separate collection. B2F recycling process. Benefits with substitution of virgin material were estimated.	-1.55	0.76	(Gileno & Turci, 2021; Simon et al., 2016) Ecoinvent data for PET granulate production
Worst case scenario	Incineration (without energy recovery)	2.993	0	Ecoinvent data for polyethylene incineration

4.3.4 Market weights

To estimate the quality of recycling at the European level, market data is needed to determine the weight of the different pathways (W_{RPi}). This data is dynamic and should represent changes in the different stages at the system level over time. Therefore, W_{RPi} is intended to show the representativeness of each collection system, recycling technology and application for a given period in Europe.

To develop this methodology, literature and market data were reviewed to understand PET recycling in Europe's current state in terms of collection systems, sorting, recycling processes and applications. An intensive search was carried out for publicly available information from reliable sources, with a preference for historical data. This analysis was essential for defining the most relevant PET recycling pathways that need to be modelled to determine the quality of PET recycling at the European level.

As mentioned, the calculations will be presented for 2020 and 2018. Table 8 shows the values obtained for each parameter.

Table 8. Parameters for the calculation of market weights of recycling pathways in the VCM for PET

Data nature	Parameter	Processes	2020	2018	Data source
Dynamic	Weight of collection system	Separate collection	68%	67%	(Eunomia, 2020, 2022; Unesda, 2023)
		DRS	32%	33%	
	Weight of recycling technology	Mechanical recycling	100%	100%	-
	Weight of application	Packaging	77%	76%	(Eunomia, 2020, 2022)
		Fibres	23%	24%	

- **Weight of collection system** – This estimation requires the amount of total PET packaging collected and the amount of PET packaging collected through separate mixed plastics collection and DRS.

  For 2020, Eunomia's 2022 report does not present the quantities of PET packaging collected; it only reports the quantity collected and sorted in one stage. Therefore, to determine the weights of the collection systems, a 10% sorting loss was assumed based on the 2018 sorting loss (equivalent to 10%) (Eunomia, 2020). It was therefore possible to estimate that 2.7 Mt were collected, of which 2.4 Mt were bottles and 0.2 Mt were trays (Eunomia, 2022). There is no reference to the amount of PET packaging collected by each collection system, which presents a limitation for the study. As such, it was assumed that 35% of the total PET bottles were collected via a DRS, while the remaining PET bottles and PET trays were collected through mixed plastics collection (Unesda, 2023).

  In 2018, 2.1 Mt of PET packaging (bottles and trays) were collected (in this year, the quantities collected are known) of which 2 Mt were bottles. Due to insufficient data, the assumption that 35% of the total PET bottles were collected through DRS was maintained for 2018 (Unesda, 2023).

- **Weight of recycling technology** – Chemical recycling was not included in the scope of assessment as it does not fulfil the criteria of scale, distinctiveness and data availability. As such, the only analysed recycling technology was mechanical recycling.

- **Weight of application** – The application of rPET can be either packaging or fibres. Data on the applications of the PET collected by each collection system is not available. So, the total application shares of PET were considered the same for both collection systems. For 2020, Eunomia reports 1.7 Mt of PET flake produced, of which 1.3 Mt is used in packaging applications and 0.4 Mt in low-viscosity applications – fibres (Eunomia, 2022). As for 2018, Eunomia reports 1.3 Mt of PET flake produced. Fibres represent 24% of the rPET end-market, equivalent to 0.3 Mt of rPET used in low-viscosity applications, with the remaining 1 Mt used in packaging applications (Eunomia, 2020).

4.4 Consolidation

The weights of the different pathways (W_{RPi}), which make it possible to assess the quality of PET recycling on a European scale, are shown in Table 9 for the two years under study, as well as the scores for quality of recycling for the specific pathways (QR_{RPi}). It was, therefore, possible to estimate an aggregated recycling quality score (QR_{PET}) for PET packaging in Europe through Equation 10.

Table 9. Calculation of the quality of recycling of PET in Europe.

	2020		2018	
Pathway	W_{RPi}	QR_{RPi}	W_{RPi}	QR_{RPi}
SC-P2P	53%	0.447	51%	0.337
SC-P2F	15%	0.395	16%	0.312
DRS-P2P	25%	0.746	25%	0.729
DRS-P2F	7%	0.586	8%	0.577
QR_{PET}	0.523		0.452	

It should be noted that for 2020, there is uncertainty associated with the quantities of PET packaging collected since Eunomia's 2022 report does not report this quantity, only the quantity collected and sorted. For this reason, and to estimate this amount, the percentage of sorting losses (equivalent to 10%) from 2018 was assumed. In this way, it was possible to estimate the amount of PET packaging collected. This estimate is reflected in the weight of the collection systems defined.

The uncertainty is also associated with the quantities collected by the DRS system, which are not reported in Eunomia's reports. The report for 2018 only mentions that DRS accounts for up to 35% of total PET collection. As such, this estimate affects the final recycling quality score determined for PET and is one of the points for improvement in the future.

4.5 Limitations and uncertainty of case study

Evaluating the quality of PET packaging recycling faced several challenges in developing the present framework due to the inherent complexity of this waste stream, as well as the publicly available data. Right from the start, the recycling quality score presented the uncertainty associated with the quantity of PET bottles collected through the DRS system in Europe. According to the Eunomia 2020 report, 35% of the PET bottles are collected through DRS. However, no estimate is made for 2018, so the same figure of 35% was assumed. This assumption influences the collection efficiencies presented, as well as the market weights of the collection systems.

Another challenge encountered when selecting the recycling pathways was finding quantitative (and even qualitative) data regarding the sorting of PET packaging waste, which is why sorting was not differentiated in the scope of the study but considered together with the collection. The recycling quality of PET packaging waste was assessed for 2018 and 2020. A limitation in the analysis was identified because, for 2020, no losses in the sorting process were reported. Still, for 2018, these losses were quantified. This inconsistency is reflected in the efficiency of collection and sorting.

The definition of applications also faced constraints, particularly regarding the granularity that the study could include. Currently, with the available published data, it is impossible to differentiate applications into bottles, trays, and flexible packaging, which is why a global category called “packaging” was considered.

At a more global level of the study, a general difficulty was encountered in obtaining historical data to evaluate the evolution of PET packaging recycling quality. Quantitative data before 2018 are scarce and hard to find. The study was feasible for 2018 and 2020, based on Eunomia’s reports for 2020 and 2022, respectively. Note that this source publishes reports every two years, so it is expected that the quality of recycling for this flow can be updated at least every two years.

5 Test of Model on Biowaste

5.1 Introduction to the waste stream

Bio-waste was selected as a case study for implementing the quality of recycling framework due to its specificities as a waste stream; namely, that it is mainly composed of water (up to 90%) and it naturally degrades, defying our common understanding of recycling (Caro et al., 2023). In that sense, it is necessary to verify if the methodology for the quality of the recycling framework can still be implemented on this waste stream or whether adaptations are required.

According to ECN (2022), bio-waste consists of biodegradable garden and park waste, food and kitchen waste from households, offices, restaurants, wholesalers, canteens, caterers and retail premises and comparable waste from food processing plants. It does not include forest or agricultural waste, manure, sewage sludge or other waste such as natural textiles, paper and processed wood.

Bio-waste can be recycled using two leading technologies:

- Composting is carried out in the presence of oxygen, usually in open-air windows or vessels. The process generates compost, a moist substance derived from the biodegradation of organic solids that can be used as fertiliser, soil improver or growing media. The process is optimised when the mixture is easily degradable, with wet organic substances such as food waste and structure-improving organic matter such as garden waste (EEA, 2020).
- Anaerobic digestion (AD) is a process carried out in closed vessels without oxygen, which produces biogas that can be used to generate electricity or heat or upgraded to fuel and a digestate that can be used as a fertiliser or soil improver. The process can use different input organic materials but does not break down lignin, a central component of wood. This technology recovers both energy and material (EEA, 2020).

It is important to note that AD is not always technically feasible, for example, when large amounts of garden waste are present. Although garden waste can be processed using this technology, it usually reduces the energy yield of the process due to the presence of lignin, which is not broken down without oxygen (EEA, 2020).

The digestate can be applied directly as fertiliser in the soil or further composted to ensure its sanitation and stabilisation, resulting in a compost with similar quality and composition to the compost obtained from direct bio-waste composting (JRC/IES, 2011). Conversely, compost does not have the nutrients as readily available, releasing them more slowly into the soil, thus contributing to the soil's long-term fertility (ECN, 2019; JRC, 2014).

5.2 Scope of assessment

The assessment is focused on municipal biowaste, mainly generated in households, but it also includes biowaste from similar sources such as shops, offices, and public institutions. Generally, it comprises bio-waste from households and the HoReCa sector (Hotels, Restaurants and Catering) that is collected by municipal authorities and disposed of through waste management systems (ECN, 2022).

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In-depth research was carried out to define the main recycling pathways that currently exist for bio-waste in the EU:

1. **Collection:** Several collection systems have been implemented for bio-waste in the EU. Separate bio-waste collection is widely implemented in several European countries: Austria, Germany, Luxembourg, Netherlands, Poland, Slovenia, and Sweden (EEA, 2024). Many door-to-door collection systems focus on food waste, leaving garden waste for less frequent collection. In several Member States, drop-off points are strategically located in accessible areas within communities for the collection of bio-waste (EEA, 2020; Zero Waste Europe, 2020).

   About 50% of municipal biowaste is collected as residual (mixed) waste (EEA, 2020). The mixed waste can then be sent to mechanical and biological treatment (MBT), where the organic fraction is separated and subjected to biological treatment. A data search was carried out to determine whether this pathway should be included in the scope of assessment. Eurostat only provides data concerning the final treatment of municipal solid waste (MSW). However, there were several constraints with the available data. Eurostat reports data regarding the final treatment of waste according to the following categories: landfill, incineration, material recycling, composting/digestion and other treatment. The 'other treatment' category captures the differences between the amount treated and the amount of waste generated due to uncollected waste while also reflecting the effects of import and export, weight losses, double-counting of secondary waste (e.g. landfilling and recycling of residues from incineration), differences due to time lags, temporary storage and, the use of pre-treatment, such as MBT (Eurostat, 2024). Mechanical biological treatment (MBT) is not covered directly as a category in the reporting of MSW treatment. Eurostat (2024) states that the amounts delivered to MBT should be reported based on the subsequent final treatment steps. Still, the reporting approach varies significantly between countries, with many only reporting on the first (pre-) treatment step. Given the lack of available data, the pathways associated with the collection of bio-waste as mixed waste and subsequent MBT were excluded from the scope of assessment. In addition, it is essential to highlight that separate collection of bio-waste will become mandatory as of January 1, 2024, and that from 2027, the processing of bio-waste as mixed waste in MBT plants will not count towards national recycling targets, as countries will only be able to report recycled bio-waste if it comes from the separate collection (EEB, 2020).

2. **Sorting:** A pre-treatment stage before recycling can be implemented where the bio-waste is sorted (e.g. resorting to magnets and screening technologies) to remove unwanted contaminants (e.g. packaging, bags) (Caro et al., 2023). There is no EU-average data available on the efficiency of bio-waste sorting. For this reason, sorting is not differentiated in the scope of assessment. It is worth noting that losses at this stage are highly influenced by the efforts made for separation at the source (households) to avoid contamination and for separate collection.

3. **Recycling:** Bio-waste is typically recycled via biological recycling through composting or anaerobic digestion (Caro et al., 2023). Composting, the process of transforming bio-waste into compost, is currently the predominant recycling method in Europe. However, AD, for the production of digestate, stands out as an emerging and advanced method for treating bio-waste as it also allows for biogas production. The choice between the two methods usually depends on the composition of the bio-waste (EEA, 2020; Interreg Europe, 2021). It is

important to note that it was impossible to disaggregate bio-waste into food and garden waste under this scope of assessment. This is a limitation of this assessment, which is particularly relevant in AD, as it is not technically feasible to treat garden waste via this process.

AD and composting are often combined to maximise the advantages of both approaches (Lewerenz et al., 2023). There is a lack of consistent data reporting concerning this pathway over the two available periods. According to the ECN, in 2019, this pathway represented around 10% of the total separately collected bio-waste sent to recycling (ECN, 2019), whereas, in 2022, this data was incorporated into the reporting on composting to avoid data duplication (ECN, 2022). For these reasons, this pathway is currently not included in the scope of assessment.

In the case of AD, nutrient management strategies are necessary to avoid the risk of soil contamination and to allow the full use of the fertilising potential of digestate, which allows the isolation of the intended nutrients and removal the unwanted compounds (Angouria-Tsorochidou et al., 2022). Currently, post-treatment technologies are not commonly used to improve efficiency in recycling nutrients and reduce the associated environmental impacts (Angouria-Tsorochidou et al., 2022) but should be taken into account when assessing the quality of recycling of bio-waste in the upcoming years if they become relevant.

4. **Application:** Agriculture is the dominant market segment for compost and anaerobic digestate (ECN, 2022). Compost is a versatile soil improver used in agriculture, horticulture, landscaping, and land restoration. It enhances soil properties physically, biologically, and chemically, acting as a soil improver and, to a lesser extent, an organic fertiliser. Digestate, primarily used in agriculture, is valued for its fertilising properties, rich in nitrogen (N), phosphorous (P) and potassium (K) while also contributing to soil improvement (ECN, 2019; JRC, 2014). Post-treatment technologies can be implemented to separate digestate into liquid and solid fractions, each with significantly distinct compositions, providing added benefits (Chojnacka & Moustakas, 2024). Due to the lack of available data, the assessment did not consider this differentiation.

As already described, energy recovery is also part of anaerobic digestion. In this case, biogas from anaerobic digestion could be considered an inherent loss. Therefore, the quantity of material used as fuel or other means to generate energy would not be considered a recycled quantity. However, as the Waste Framework Directive establishes, biogas can be considered a recycling process when digestate is used as a recycled product, material or substance (Caro et al., 2023).

Biogas can be used as a source of energy to replace electricity, natural gas and even vehicle fuel (JRC/IES, 2011). The benefits associated with each of these scenarios differ significantly, as substituting natural gas or fuels results in more significant avoided impacts than substituting an electricity mix (the difference will be even more significant with the expected decarbonisation of electricity in the EU) (Ardolino et al., 2018). The detailed modelling of energy recovery is outside the scope of recycling quality and, therefore, is not analysed. However, interest in biogas upgrading to biomethane with the possibility of injection into the national gas grid is expected to increase in the upcoming years (Calise et al., 2021), so the analysis of energy recovery scenarios can become more relevant in future assessments.

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It is essential to clarify that composting and anaerobic digestion are not single technological processes, as different technologies are being implemented. In the case of composting, turned windrows and in-vessel systems are the most common composting systems. For AD, digestion systems can be dry, semi-dry or wet, continuous, semi-continuous or in batch, and thermophilic or mesophilic (JRC/IES, 2011; Vieira & Matheus, 2019). The lack of available data makes it impossible to differentiate the recycling pathways at this technology level. Additionally, this level of detail is not considered relevant as it is expected that the differences in technologies within composting and AD are not so significant that they should require this level of detail.

Differences in performance are, in fact, more relevant for the AD followed by the composting pathway, which is currently excluded from the analysis. If compost obtained from direct composting and composting of digestate is similar in composition and quantity, then it is likely that the pathway AD followed by composting has the best environmental performance compared to direct composting, given the energy recovery that takes place (Caro et al., 2023).

The research and data collection on the different stages of the bio-waste value chain culminated in the definition of the most representative recycling pathways for bio-waste, namely:

- Bio-waste separate collection – Composting – Soil application (compost);
- Bio-waste separate collection – Anaerobic Digestion – Soil application (digestate).

Although the project aspired to assess the quality of bio-waste recycling over 10-15 years, this case study, like PET, showed no time series data available for this period. This was also confirmed during the stakeholder consultation carried out during the project, which involved the European Compost Network (ECN).

Extensive research was carried out to evaluate the bio-waste case study. The dynamic data comes from the ECN's status report publications (Figure 7). The first was published in 2019 and refers to 2016-2017 (ECN, 2019), and the second publication dates from 2022 and refers to 2019-2020 (ECN, 2022).

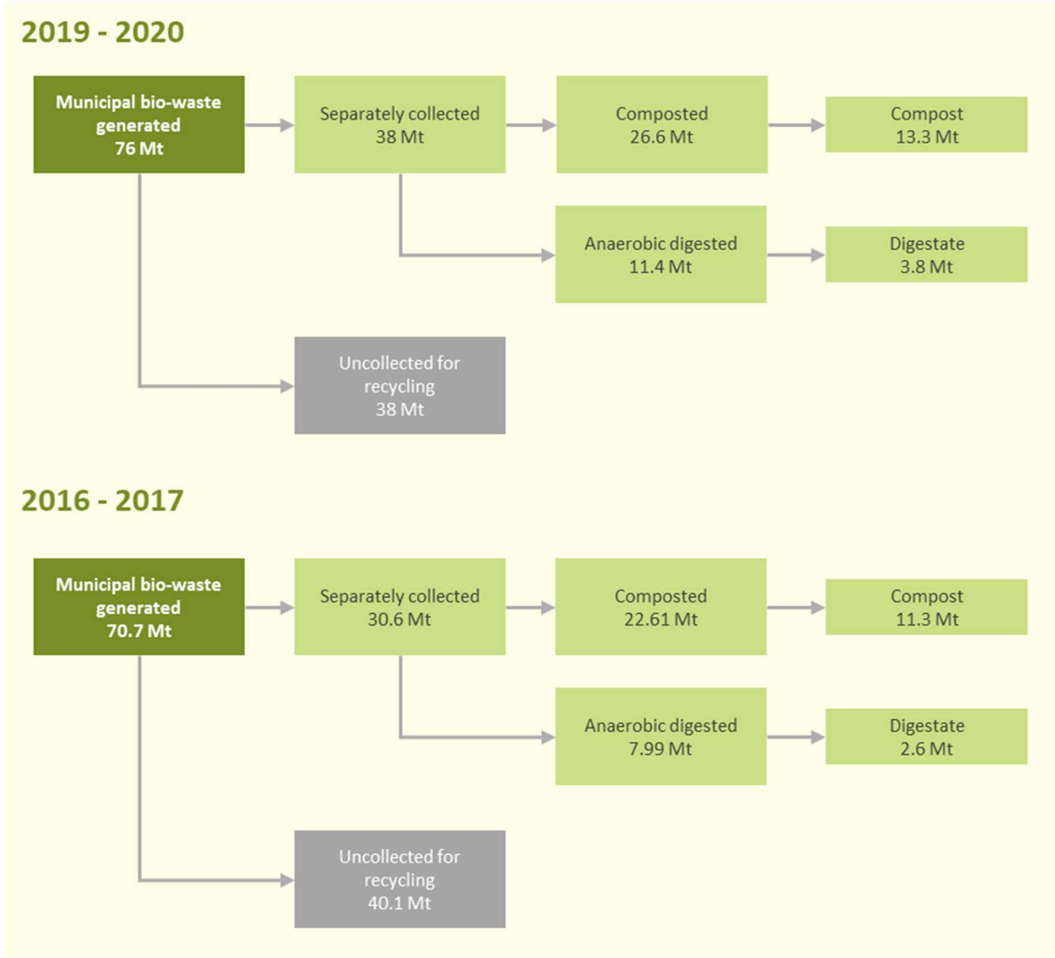

Figure 7. Market data for municipal bio-waste used for the assessment

5.3 Application of the quality of recycling metric

The quality of the recycling (QR_{RPi}) of each pathway defined for bio-waste was determined by Equation 1). The implementation of the VCM is detailed in the following sections.

5.3.1 Efficiency

For bio-waste, the efficiency of the recycling pathways (E_{RPi}) was calculated according to Equation 2, with no differentiation in the assessment scope for sorting technology. It is important to note that recycling efficiency and the technical substitution ratio are data expected to remain static over the next few years unless a disruptive technology emerges that significantly influences these values. Table 10 presents the parameters used in the efficiency calculation, while the subsequent sections outline data availability and treatment.

Table 10: Parameters for the calculation of the efficiency dimension of the VCM for bio-waste

Data nature	Parameter	Processes	2019-2020	2016-2017	Data source
Dynamic	Efficiency of collection and sorting	Separate collection	50%	43%	(ECN, 2019, 2022)
Static	Recycling efficiency	Composting	50%		(ECN, 2022)
		Anaerobic digestion	33%		(ECN, 2019)
Static	Technical substitution ratio	Compost	0.63		Based on Møller et al. (2009) and Sardarmehni et al. (2021)
		Digestate	0.61		

- **Efficiency of collection and sorting** – The collection and sorting system was evaluated together due to the lack of available data.

  The ECN estimates that 224 Mt of municipal waste was generated during 2016-2017 and that 34% of this is bio-waste (34%-46%). This results in an estimated generation of around 76 Mt of bio-waste. For 2019-2020, the ECN reports that 38 Mt of municipal bio-waste was collected separately, which signifies a collection and sorting efficiency of 50 %.

  For 2016-2017, the ECN does not differentiate the data between municipal and non-municipal bio-waste. This differentiation is needed since only municipal bio-waste falls within the scope of this study. It is stated that 47.5 Mt of municipal and non-municipal bio-waste were collected separately (ECN, 2019). The following publication reports 38 Mt of separately collected municipal bio-waste and 21 Mt of non-municipal bio-waste, meaning that municipal bio-waste represents 64% of the total bio-waste collected (ECN, 2022). For lack of better data, this percentage was assumed for 2016-2017, totalling 30.6 Mt of municipal bio-waste collected. It was also estimated that 70.72 Mt of municipal bio-waste was generated, assuming 208 Mt of municipal solid waste was generated in Europe in that time frame, according to the ECN 2019 report, and that 34% of it was bio-waste. Therefore, the efficiency of collection and sorting was estimated at 43%.

- **Recycling efficiency** – The ECN reports a 50% conversion of bio-waste to compost in Europe (ECN, 2022). However, in this report, there is no reference to the conversion of bio-waste to digestate, nor is the amount of digestate produced reported. Consequently, an estimate was made by relying on the quantities outlined in the preceding report from the ECN. According to the previous publication, 12.4 Mt of bio-waste was recycled through AD, yielding 4.1 Mt of digestate (ECN, 2019). Based on this, the conversion rate from bio-waste to digestate was estimated at 33%.

  It is important to emphasise that this estimated conversion for digestate may have a significant error because the ECN 2019 report does not differentiate between municipal bio-waste and non-municipal bio-waste. As such, the assumed values shown in Table 10 for the two efficiencies have a certain level of associated uncertainty that should be addressed in future data reporting. It is worth noting that the ECN was consulted about the data reporting for digestate during the consultation process, indicating difficulties in obtaining these values.

- **Technical substitution ratio –** The TSR considered in other waste streams has a straightforward interpretation, but in the case of compost and digestate, one must consider more factors. Nutrients can be present in different forms, affecting how available these are for different purposes (e.g., agriculture and soil restoration). The scientific community has approached this question with the Mineral Fertiliser Equivalency (MFE) factor (Sardarmehni et al., 2021), further described in section 5.3.3. This parameter indicates the amount of fertiliser offset according to the availability of the applied nutrients to plants in relation to conventional fertilisers. The organic matter in compost and digestate is also vital as a soil improver. A fraction of the initial organic carbon content remains stored in the soil after 100 years, whereas the rest is converted aerobically to biogenic CO_2, thus providing additional ecosystem benefits such as water retention and reducing soil erosion (Joseph & Stichnothe, 2023). This fraction varies significantly in the literature for compost.

  Boldrin et al. (2009) indicates that the fraction of carbon still bound to soil after 100 years has been estimated to be between 2% and 14% of the input in compost according to different authors, depending on the soil type and the crop rotation. Sardarmehni et al. (2021) indicates that 2-16% of the initial carbon content remains stored in the soil. Studies analysed by (ISWA, 2020), on the other hand, have shown that over 4-12 years, between 11% and 45% of the organic carbon applied to soil as compost remained as soil organic carbon.

  Regarding digestate, the long-term benefits to soil are less clear-cut compared to compost, and it is thought to have a negligible effect on soil organic matter in the long term (ISWA, 2020). However, Møller et al. (2009) considers that the sequestered carbon not released as CO_2 during the 100-year period, is in the range of 4–14% of the applied amount of carbon.

  In the case of bio-waste, carbon and a set of nutrients are thus provided in soil applications, and it was assumed that every element has an equal weight to calculate the global TSR value for bio-waste (Equation 18). In this context, TSR can be interpreted as the MFE for nutrients and the binding factor for carbon. As detailed in section 5.3.3, there is a high level of uncertainty on MFE and carbon binding for both compost and digestate. Further studies on the availability of carbon and nutrients in compost and digestate are required to increase the robustness of the results.

$$TSR_{element} = \frac{quantity\ available\ as\ fertiliser/soil\ improver}{quantity\ of\ element\ added} \qquad \text{Equation 17}$$

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$$TSR_{bio-waste} = u_N \times TSR_N + u_P \times TSR_P + u_K \times TSR_K + u_C \times TSR_C \qquad \text{Equation 18}$$

$$u_N + u_P + u_K + u_C = 1$$
$$u_N,\ u_P,\ u_K,\ u_C \in [0; 1]$$

In this assessment, the data from Table 11 and Table 12 was used to calculate the $TSR_{bio-waste}$ parameter.

Table 11. MFE values for compost and digestate obtained from literature

MFE	Compost	Digestate
	(Sardarmehni et al., 2021)	(Møller et al., 2009)
N	60% (20-80%)	40%
P	90% (80-100%)	100%
K	90% (80-100%)	100%

Table 12. Carbon binding factors for compost and digestate obtained from literature

MFE	Compost	Digestate
	(Sardarmehni et al., 2021)	(Møller et al., 2009)
C_{stored}	10% (2-16%)	4% (4-14%)

5.3.2 Loop potential

According to the scale set out in Table 3, which will require validation through expert elicitation, the soil application of compost and digestate is scored as 0.6 and 0.4, respectively. As in the TSR, the definition of the loop dimension for biowaste requires careful consideration. The loop dimension should represent the potential for keeping the material circling within the economy. For bio-waste, it is necessary to consider nutrients as the materials to be kept circling. Theoretically, if all nutrients become available to be integrated into biomass again, then the loop dimension should be 100%. Compost and digestate might be similar in terms of applications, but the literature indicates differences in physicochemical properties. The majority of the total nitrogen concentration in digestate is in the form of ammonium/ammonia, which provides higher short-term fertilisation potential and exhibits a higher susceptibility to losses. This causes concerns regarding the use of digestate on land, such as phytotoxicity and eutrophication. On the other hand, the total nitrogen concentration of stabilised compost is mainly in organic forms, which are less susceptible to leaching and volatilisation (Vieira & Matheus, 2019). Additionally, there is little scientific evidence of humification during anaerobic digestion, indicating a lesser contribution to the soil's organic matter than compost (ISWA, 2020).

5.3.3 Environmental performance

An extensive literature review of LCA studies concerning bio-waste recycling was developed. Vieira & Matheus (2019) reviewed LCA studies of the composting and anaerobic digestion of municipal biowaste and concluded that the studies presented methodological differences and contradictions in modelling, which prevents a comparison of results. A careful assessment of the available studies is thus required to ensure the data's adequacy to the assessment's objectives, following the recommendations provided in section 3.2.1.3.

Data from Colón et al. (2015) was considered to model the environmental impacts of the recycling pathways AD and composting. The authors analysed the impacts of different composting technologies and bio-waste AD in a region of Spain, including the collection and transport stage. In the case of AD, energy recovery, namely the production of electricity from biogas, was considered in the analysis. The authors did not consider a system expansion to include the avoided impacts of substituting synthetic fertilisers. For this reason, this modelling was carried out based on other LCA studies and further explained in this section.

Studies and methodologies addressing the LCA impacts of digestate and compost use on land are still lacking (Vieira & Matheus, 2019). LCA studies typically model the soil application of compost and digestate and the resulting replacement of nitrogen (N), phosphorous (P) and potassium (K) fertilisers (Vieira & Matheus, 2019). Other types of applications, such as landfill covers or biofilter media, are typically not considered in LCA studies (Oviedo-Ocaña et al., 2023).

A significant constraint in estimating the environmental benefits of using compost and digestate is its diverse nutrient composition and quality (Oviedo-Ocaña et al., 2023). Not all the nutrients present in bio-waste are ultimately used by plants. There are losses of nutrients to surface water runoff, ground leachate, and the air. Additionally, nutrients in compost and digestate are not as readily available to plants as mineral fertilisers. For this reason, to adjust the actual amount of fertiliser offset according to the availability of the applied nutrients to plants in relation to conventional fertilisers, a mineral fertiliser equivalent (MFE) is used (Sardarmehni et al., 2021). It is essential to highlight that this approach has several limitations regarding nutrient dynamics in soil. This equivalency approach substitutes short-term available NPK in compost and digestate, thus discarding organic forms of NPK in fertilising potentials or other unavailable forms (Vieira & Matheus, 2019).

Based on the analyses of nutrient composition of both compost and digestate from the Institute of Crop Science (2016), and the MFE values for each product (Table 11), the amount of avoided mineral fertilisers was estimated. It was considered that the elements N, P and K would avoid the mineral fertilisers ammonium nitrate (NH_4NO_3), phosphorus pentoxide (P_2O_5) and potassium oxide (K_2O), respectively. The impacts associated with the production of mineral fertilisers were modelled using data from the Ecoinvent database. The CML 2001 impact method was used to quantify the benefits associated with the avoided production of mineral fertilisers, as this was the method used by Colón et al. (2015) to model the direct impacts of both recycling pathways.

The avoided CO_2 emissions that result from the sequestration of organic carbon in the soil in the long term was estimated, according to Equation 19 (Boldrin et al., 2009), and was based on the carbon composition from Institute of Crop Science (2016) and the carbon binding factors presented in Table 12.

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$$CO_{2,bind} = C_{input} \times C_{bind} \times \frac{44}{12} \qquad \text{Equation 19}$$

For the reference scenarios required for the normalisation of the EI dimension, the worst-case scenario was defined as the landfill of bio-waste with no pre-treatment. The LCA results of this scenario were defined according to Colón et al. (2015). The best-case scenario was considered to be the total recovery of the primary nutrients present in compost (NPK) in a recycling process with zero burden. As was the case for PET, and by the methodology described in section 3.2.1.3, only the Global Warming Potential impact category results were considered and normalised to obtain the score for the EI dimension (Table 13).

Table 13. Results of environmental assessment of bio-waste scenarios

Scenarios	Description	Value (kg CO_2eq/kg)	Normalized values	Data source
Best case scenario	Total recovery of the present nutrients with zero burden recycling process	-60.07	1	(Institute of Crop Science, 2016) Ecoinvent data for synthetic fertilisers
Anaerobic Digestion	Collection of bio-waste and recycling process. Energy recovery (production of electricity). Benefits with substitution of synthetic fertilisers were estimated.	21.35	0.85	(Colón et al., 2015; Institute of Crop Science, 2016) Ecoinvent data for synthetic fertilisers
Composting	Collection of bio-waste and recycling process. Benefits with substitution of synthetic fertilisers were estimated.	11.00	0.87	(Colón et al., 2015; Institute of Crop Science, 2016) Ecoinvent data for synthetic fertilisers
Worst case scenario	Landfilling of bio-waste	490.46	0	(Colón et al., 2015)

The close results of the environmental performance of the two pathways do not allow to assert that one is better than the other, as there is a high level of uncertainty associated with modelling the avoided impacts associated with synthetic fertiliser substitution and carbon sequestration.

5.3.4 Market weights

A detailed analysis was carried out to obtain data at European level that would allow the quality of recycling to be estimated for the different pathways (W_{RPi}) in Europe. This data is dynamic and needs to be updated according to the data published in the ECN reports. Data collected refers to the years 2019-2020 and 2016-2017. Table 14 summarises the estimated weights for each stage.

Table 14: Parameters for the calculation of market weights of recycling pathways in the VCM for bio-waste

Data nature	Parameter	Processes	2019-2020	2016-2017	Data source
Dynamic	Weight of collection system (not considering the uncollected)	Separate collection	100%	100%	-
	Weight of recycling technology	Composting	70%	74%	(ECN, 2019, 2022)
		Anaerobic digestion	30%	26%	
	Weight of application	Compost	100%	100%	-
		Digestate	100%	100%	

- **Weight of collection system –** The weight of the collection system was assumed to be 100% since only one collection pathway is relevant when the uncollected bio-waste was not considered.
- **Weight of recycling technology** - Estimating the weight of each recycling technology requires an understanding of the quantities of municipal bio-waste collected separately and the quantities sent for each recycling technology (composting and AD).

  For 2019-2020, these quantities are reported directly by ECN: 70% of the collected municipal bio-waste is sent for composting (26.6 Mt), and the remaining 30% is sent for AD (11.4 Mt) (ECN, 2022).

  For 2016-2017, and as already mentioned, the ECN 2019 report does not differentiate between municipal and non-municipal bio-waste, so it was assumed that 64% of the bio-waste collected was municipal (assumption explained in section 5.3.1). Another data constraint was that in the ECN 2019 report, the quantities of bio-waste collected were distributed across three recycling technologies: composting (64%), AD (26%) and AD, followed by composting (10%). AD, followed by composting, is not established as one of the pathways in this study. As such, the amount sent to AD followed by composting was included in the composting pathway (ECN, 2022). These ratios (74% for composting and 26% for AD) were applied to the estimated amount of municipal bio-waste collected, i.e. 30.6 Mt (ECN, 2019), thus obtaining the amounts sent for each recycling technology: 22.61 Mt sent for composting and 7.99 Mt sent for AD.
- **Weight of application -** The two applications are associated with the two recycling pathways, as they have different characteristics influencing their application in the soil.

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5.4 Consolidation

Finally, given the weights of the different recycling pathways (W_{RPi}), as well as the recycling quality score for each of these pathways (QR_{RPi}), an aggregate recycling quality score for Europe ($QR_{bio-waste}$) was estimated, according to Equation 10.

Table 15. Calculation of the quality of recycling of bio-waste in Europe

	2019-2020		2016-2017	
Pathway	W_{RPi}	QR_{RPi}	W_{RPi}	QR_{RPi}
Composting – Soil application (compost)	70%	0.463	74%	0.426
Anaerobic Digestion – Soil application (digestate)	30%	0.330	26%	0.301
$QR_{bio-waste}$	0.423		0.394	

The final overall recycling quality scores presented for the bio-waste stream in Europe are, as already mentioned, conditioned and affected by the discrepancies and uncertainty in the available data described in the previous sections. This resulted in assumptions being defined in the analysis, which can impact the final scores. It would be necessary for policy purposes to improve data collection and the body of knowledge on the substitution potential and environmental performance of composting and AD.

5.5 Limitations and uncertainty of case study

As discussed in the PET test, the most relevant uncertainty and limitation stems from the dependence on industry information. Official statistics at the European level do not have the granularity and consistency to characterise the recycling quality for each recycling pathway. The primary data sources used were the 2019 and 2022 ECN reports, which report data from the 2016-2017 and 2019-2020 time periods. As an overall limitation, quantitative data before 2016 is scarce, not consolidated in a single data source and therefore insufficient to assess recycling quality for years before 2016.

By using industry data, the recycling quality ultimately incorporates the same uncertainties and limitations. In the case of biowaste, the most relevant data source for market and efficiency was the ECN report, which has two uncertainties that could not be addressed in this test. The first is not differentiating food waste and garden waste, which have different recycling pathways and should lead to different environmental outcomes, but countries have different approaches on how to quantify and aggregate the two streams. ECN does not have the data necessary to consider AD with or without composting downstream, which is why the two options are combined. Also, data on the sorting of bio-waste is scarce, making it difficult to carry out a detailed analysis; thus, the sorting efficiency is evaluated together with the collection.

For the analysis, only two recycling technologies were defined as pathways: composting and AD, but it should be noted that several composting and AD technologies currently exist in Europe. This assessment did not address the quantitative disaggregation of these technologies due to the lack of robust data. Data may be available in the future and thus will no longer be a limitation.

Another limitation arose when defining the recycling pathways, specifically the collection pathway, on whether to include MBT plants. Eurostat provides data on the final treatment of MSW, but MBT plants are not directly covered as a treatment category. In addition, the reporting approach of treatment technologies varies significantly from country to country, leading to the pathway associated with the collection of bio-waste as mixed waste and subsequent MBT being excluded from the scope of the assessment.

Composting and AD have significantly different operation conditions and material inputs. This is reflected in the scientific literature regarding the environmental performance of these methods. Many studies have inconsistent assumptions and vague or missing goal definitions. It is difficult to compare them due to different assumptions in system boundaries, how impacts are allocated to different streams and difficulties in capturing influential local specificities (e.g., the inclusion of representative waste compositions into the inventory. As suggested for testing the model on PET packaging, improving the knowledge on this topic and performing consistent and comparable LCA studies for different treatment pathways and material inputs will be necessary.

Finally, a limitation identified in the biowaste test is how the model considers contaminants that might limit how the final product/recyclate can be used. A suitability parameter is included in, for example, the JRC method, but it was considered that for simplicity, the VCM should not include such a parameter. Instead, the model should explicitly disaggregate recycling pathways according to this contamination level. For example, if bio-waste from the residual collected is composted, then it will have lower quality and can only be used in applications with low-quality standards – to approach this, one can split composting into two recycling pathways and consider different collection methods and substitution (avoided products). One could also consider a recycling pathway where bio-waste is collected from residual waste but is subject to sorting processes that remove those contaminants, and the output is also high-quality; in this case, a new recycling pathway is modelled, and the sorting efficiency is significantly lower than one where bio-waste is collected separately. At the EU level, these disaggregations are difficult due to the lack of data; at the regional or Member State level, these are possible if regional or national authorities have the data.

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6 Test of Model on Textiles

6.1 Introduction to the waste stream

According to Regulation (EU) No 1007/2011, a textile product is any raw, semi-worked, worked, semi-manufactured, manufactured, semi-made-up or made-up product composed of at least 80% textile fibres, regardless of the mixing or assembly process used (Huygens et al., 2023).

Textile waste is generated at different stages of its life cycle, namely (Huygens et al., 2023):

- Post-industrial waste, which arises during the manufacturing of textile products;
- Pre-consumer waste, which is generated at retail stages (e.g. unsold textiles);
- Post-consumer waste, which comprises textiles discarded after use by citizens or end-users in commercial and industrial activities (e.g., hotels, care, automotive), commonly denoted as household and commercial post-consumer textile waste, respectively.

Post-consumer textile waste represents 87% of the total textile waste generated in 2019 (Huygens et al., 2023). Currently, no legislation in the EU mandates the separate collection of post-consumer used textiles. However, in line with the EU strategy for sustainable and circular textiles and the impending legislation such as the Waste Framework Directive (Art. 11(1)), which requires mandatory separate textile collection from households throughout Europe by 2025, it is expected that all EU Member States will implement separate collection systems for used household textiles. Various charity organisations separately collect post-consumer textiles through outside bring banks, second-hand shops or kerbside collection (Trzepacz et al., 2023). This form of separate collection is classified differently between EU Member States. In line with the Commission proposal for a targeted revision of the Waste Framework Directive, it is considered as used and waste textiles, textile-related and footwear products as waste upon collection (Huygens et al., 2023).

Three main categories of post-consumer waste products are considered:

- Apparel (trousers, t-shirts, sweaters, coats, footwear, dresses, apparel accessories such as scarves, handkerchiefs, etc.);
- Household textiles (curtains, bed linen, carpets, etc.);
- Used textiles from professional textile applications (technical textiles such as automotive applications, medical textiles, agro-textiles, protective equipment, etc.).

When textiles have been collected, these are either sorted locally or exported for sorting in other countries. The sorting process includes the elimination of non-textile items from the collected materials and the categorisation of either reusable or recyclable textiles. Notably, the sorting process has an economic incentive to prioritise sorting for reuse as this fraction is worth more than textiles sent to recycling. Some of the textiles are sold second-hand or sent to recycling in Europe; however, the majority are exported outside of Europe for either reuse or recycling (ETC CE, 2023). For the fraction sorted for recycling, mechanical recycling is predominant in Europe, while chemical recycling is still in its early stages and can be divided into closed-loop mechanical recycling and open-loop mechanical recycling (Huygens et al., 2023). Closed-loop recycling is a mechanical process where textiles are torn to obtain individual fibres. The output material obtained following tearing consists of yarn-spinnable fibres of variable fibre length, shorter fibres that cannot be re-spun (fluff), and dust. Open-loop recycling is the direct use of shredded textiles in specific industrial applications outside the retail apparel sector (e.g. wiping rags) or further processed into non-wovens (e.g. insulation materials). The processes are pretty similar as currently closed-loop recycling only generates a small amount of long fibres that can be yarn-spinned.

6.2 Scope of assessment

The scope of this assessment focuses on post-consumer textile waste. Detailed research was carried out to define the main recycling pathways for European textiles.

- **Collection:** Separate collection of textiles for reuse and recycling through pick-up and drop-off bins was considered. Denmark and Finland have the highest rates of separate collections of textiles, as they have well-established systems for managing textiles for longer than the European average (Huygens et al., 2023).
- **Sorting:** Manual sorting is the predominant methodology and the only one considered as a pathway. Automatic sorting is in its infancy (less than 1% of post-consumer textile waste is sorted automatically), but it is expected that in the future, automatic sorting will supplement the sorting of textiles destined for recycling. Still, manual sorting will continue to play a central role in assessing the quality of textiles, especially when their reuse is in question since there is currently no automatic sorting technology that can perform actions beyond fibre and colour assessment. It is important to note that manual sorting is mainly carried out in countries with low labour costs, both inside and outside the EU (which explains why considerable amounts of waste are sent for export) (Huygens et al., 2023).
- **Recycling:** Textiles can be recycled using mechanical, chemical and thermal technologies. Mechanical recycling is the most representative process in the EU, with chemical recycling in its initial stages. Two pathways were considered for mechanical recycling: open-loop and closed-loop recycling (Huygens et al., 2023).
- **Application:** In the case of closed-loop recycling, there are two outputs for the process resulting in two types of applications: spinnable fibres, which are used to produce apparel and home textiles, and non-spinnable fibres which are used to produce other textile applications, such as cleaning cloths, car insulation, roofing felt, panel linings or other non-woven composite materials. Considering only the application of spinnable fibres, which is the main objective of the recycling process, would result in a low-efficiency process, disregarding that the remaining output can still be recycled into lower-value applications. In that sense, the three dimensions of the quality of recycling are analysed for the set of applications resulting from closed-loop recycling. In the case of open-loop recycling, only the application of non-spinnable fibres into other textiles is considered.

The development of the textiles case study was only possible for 2019 due to the lack of coherent public data for previous years, resorting to the data from the study 'Techno-scientific assessment of the management options for used and waste textiles in the European Union', published by the JRC in 2023 (Huygens et al., 2023), which consolidates the most recent and available information on this material fraction (Figure 8).

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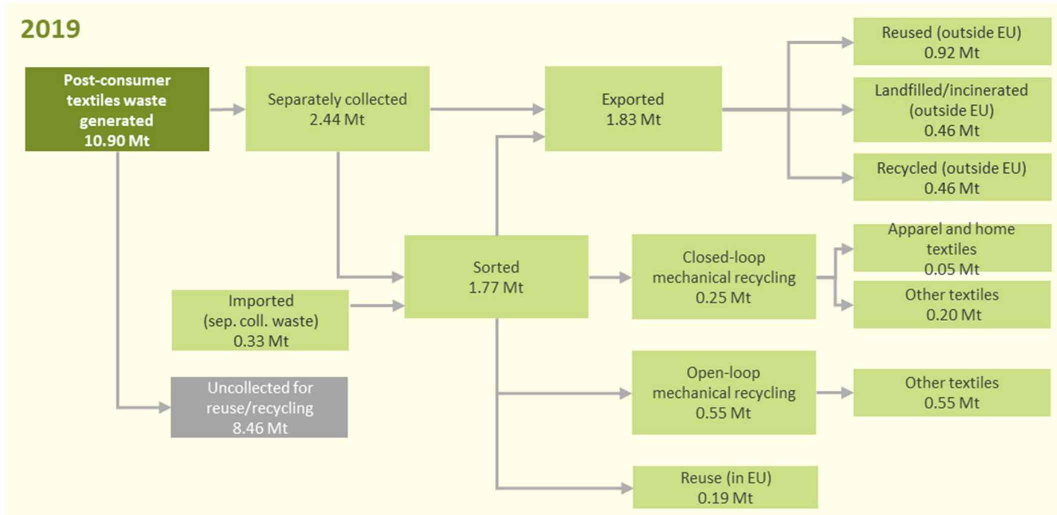

Figure 8. Market data for post-consumer textile waste used for assessment

6.3 Application of the quality of recycling metric

The recycling quality for each defined pathway (QR_{RPi}) was determined based on the three parameters: efficiency, loop dimension and technical substitution ratio (Equation 1). The following sections present the data analysis and assumptions that allow it to be determined.

6.3.1 Efficiency

The efficiency of the recycling pathways for textiles was determined using Equation 2. The data available for textiles allows for the disaggregation of collection and sorting efficiencies. Although there is no available time series data for the management of textile waste in the EU, and data from the one-shot assessment study by Huygens et al. (2023) is being used, the efficiency parameters are considered dynamic (Table 16).

Table 16: Parameters for the calculation of the efficiency dimension of the VCM for textiles

Data nature	Parameter	Processes	2019	Data source
Dynamic	Efficiency of collection	Separate collection	22%	(Huygens et al., 2023)
Dynamic	Efficiency of sorting	Manual sorting	74%	
Dynamic	Recycling efficiency	Closed-loop	100%	
		Open-loop	100%	
Static	Technical substitution ratio	Apparel and home textiles + other textiles	75%	
		Other textiles	80%	

- **Collection efficiency –** The collection efficiency calculation considers the amount of post-consumer textile waste generated and how much is collected through separate collection systems for recycling/reuse. In 2019, the JRC reported 10.9 Mt of post-consumer textile waste generated and 2.44 Mt collected separately, resulting in an estimated collection rate of 22% (Huygens et al., 2023). It is worth noting that this efficiency considers the quantities of textiles collected for reuse, as it is impossible to distinguish at this stage if textiles are adequate for reuse.

- **Sorting efficiency –** In the sorting stage, textiles are separated and sent to treatment options in the EU (reuse, recycling and elimination (landfill and energy recovery)) or exported. Reuse is outside the scope of the methodology, so the quantities of reused textiles are excluded from the sorting efficiency calculation not negatively to impact the recycling efficiency.

  The quantity of collected waste sent to sorting includes the 2.44 Mt of separately collected textile waste in the EU, as well as 0.33 Mt imported textiles from separate collections. From this quantity, the 0.19 Mt of textile waste reused in the EU and the 0.92 Mt of exported waste reused are discounted. Considering that 0.78 Mt of post-consumer textile waste was recycled in the EU and 0.46 Mt was exported for recycling (Huygens et al., 2023), a sorting efficiency of 74% is obtained.

- **Recycling efficiency –** The recycling efficiency of the pathways was determined based on the material flow analysis presented by the JRC (Huygens et al., 2023) and considers the quantities sent to each recycling pathway (closed-loop and open-loop), as well as the quantities produced in each application pathway.

  The study (Huygens et al., 2023) indicates that 0.2-0.3 Mt of textile waste is sent to closed-loop recycling, resulting in 0.03-0.05 Mt of spinnable fibres and 0.15-0.25 of non-spinnable fibres. In the case of open-loop recycling, 0.5-0.6 Mt results in 0.5-0.6 Mt of non-spinnable fibres. From this data, it is impossible to determine the losses that occur in the two recycling processes; efficiencies of 100% were considered for both recycling processes, given that in the case of closed-loop recycling, the non-spinnable fibres are also considered to be an output of the process thus contributing to its efficiency. This is a limitation in the assessment which needs to be addressed in future developments of this case study.

- **Technical substitution ratio –** The technical substitution ratio indicates the amount of recycled product that can replace virgin material. According to the JRC report (Huygens et al., 2023), spinnable fibres produced through recycling can replace virgin fibres by 52.5%, and non-spinnable fibres (fluff) can replace insulation material by 80%. In the case of closed-loop recycling, a weighting of the previously mentioned TSR values for both spinnable and non-spinnable applications according to their respective weights was carried out (75%). The TSR is lower for spinnable fibre applications due to difficulties in maintaining the technical properties of virgin fibres.

6.3.2 Loop potential

The application of non-spinnable fibres into other textile applications from open-loop recycling was classified as 0.2, according to the scale proposed in Table 3, as the material value is lost after one cycle. In the case of closed-loop recycling, it was considered that 80% of its output, which is non-

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spinnable fibres, have the previously indicated loop potential score of 0.2; the remaining 20% of the output is spinnable fibres that can replace virgin fibres, is given a score of 0.8 (the material loses some functionality and technical properties, but it is possible to continue to cycle the materials). The result is a global loop potential score of 0.4. This dimension should be validated by expert elicitation.

6.3.3 Environmental performance

The JRC study (Huygens et al., 2023) carried out an extensive technical characterisation of existing LCA studies to quantify the potential environmental impacts of textile waste re-use, recycling, incineration and landfilling. The functional unit was considered to be the "*management and treatment of 1 metric tonne of a selected textile waste material fraction, wet weight including impurities from collection*". The processes included in the analysis are collection, sorting, recycling (including pre-treatment), energy recovery via incineration, and landfilling. The analysis also considers the benefits of substituting market materials, products and/or energy recovery.

The assessment considers the comparison between different waste management scenarios, including reuse, closed-loop and open-loop mechanical recycling and landfilling, which were analysed resorting to the EF 3.0 Life Cycle Impact Assessment (LCIA) method.

According to the study, the foreground data (consumptions, emissions, yields, etc.) used to describe the waste treatment technologies were taken from previous JRC studies or the recent literature. On the other hand, the background data describing the input to the foreground system (e.g. electricity supply, fuels, chemicals, ancillary materials) are represented with datasets from the Ecoinvent database.

The results of the assessment of the different waste management scenarios were presented in the study per type of fibre, focusing on the set of fibres that together make up more than 80% of the total EU-27 textile waste by weight, namely 1) cotton, 2) polyester, 3) wool, 4) polyamide, 5) viscose, and 6) mixed polycotton. Figure 9 presents the climate change results presented in the study for both mechanical recycling processes under analysis, closed-loop and open-loop, for each of the aforementioned fibres. It is possible to observe that the results vary significantly depending on the type of fibre.

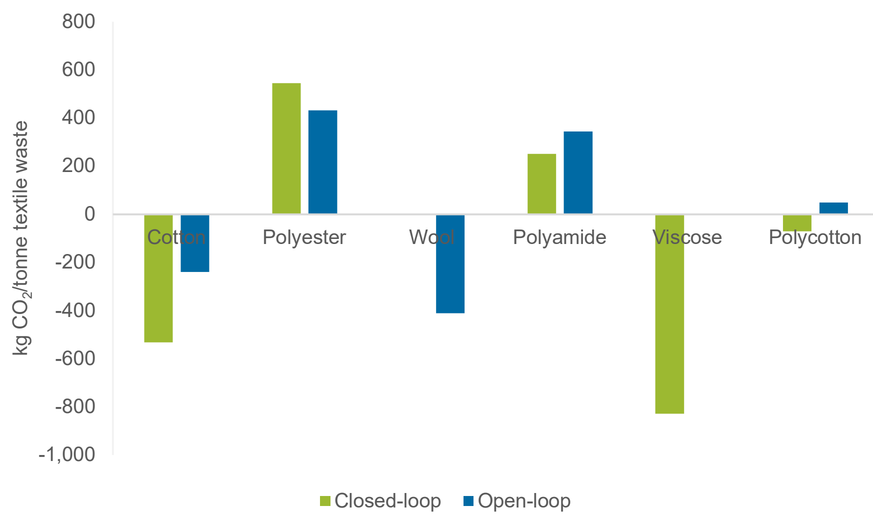

Figure 9. Results of environmental assessment of mechanical recycling processes for textile fibres (Huygens et al., 2023)

The average fibre composition in post-consumer textile waste was considered to obtain a weighted result for estimating the environmental performance of the waste management scenarios under analysis. The worst-case scenario is the landfilling of textile waste, and the best-case scenario is reuse, in which virgin fibres are substituted by second-hand textiles. As described in section 3.2.1.3, the results for the Climate Change impact category presented in the study were considered and normalised to obtain the score for the EI dimension (Table 17).

Table 17. Results of environmental assessment of Waste Textile Scenarios

Scenarios	Description	Value (kg CO_2eq/kg)	Normalised values	Data source
Best case scenario	Substitution of virgin fibres (modelled as reuse as defined by the JRC)	-3 317	1	(Huygens et al., 2023)
Closed-loop	-	40	0.26	
Open-loop	-	67	0.24	
Worst case scenario	Landfilling of textile waste	1 130	0	

The environmental assessment results are pretty similar between the two mechanical recycling processes, given the significant weight of non-spinnable fibres resulting from closed-loop recycling. Analysing the results of Table 17, one can conclude that mechanical recycling of textiles does not have sufficiently positive environmental impacts when compared to reuse.

6.3.4 Market weights

In order to assess the quality of recycling at European level, market data was analysed to determine the weight of each recycling pathway (W_{RPi}) in Europe. This requires knowledge of the market weights for the collection, recycling and application of textile waste. This data should be dynamic and reflect the evolution of European textile waste management over time. However, as mentioned, the only publicly available data relates to 2019 (Table 18).

Table 18: Parameters for the calculation of market weights of recycling pathways in the VCM for textiles

Data nature	Parameter	Processes	2019	Data source
Static	Weight of collection system	Separate collection	100%	(Huygens et al., 2023)
	Weight of recycling technology	Closed-loop	31%	
		Open-loop	69%	
	Weight of application	Apparel and home textiles + other textile applications	100%	
		Other textile applications	100%	

- **Weight of collection system –** The weight of the separate collection in 2019 in the EU was 100% as it was the only system considered within the scope of collection.
- **Weight of recycling technology –** The weight of the recycling technology was determined for closed-loop and open-loop recycling, considering the quantities sent for each technology. Taking into account the already mentioned quantities sent for closed-loop recycling and open-loop recycling, reported by the JRC, it was determined that the weight of the closed-loop recycling is 31% and the weight of the open-loop recycling is 69% (Huygens et al., 2023).
- **Weight of application –** The application of non-spinnable fibres resulting from open-loop recycling into other textiles was considered a single application, so the application's weight is considered 100%. In the case of closed-loop, as described in section 6.2, the application of spinnable fibres into apparel and home textiles and of non-spinnable fibres into other textiles is considered globally, resulting in an application weight of 100%.

6.4 Consolidation

The consolidation of the market weights of each pathway (W_{RPi}) in the EU and the scores for quality of recycling for the defined unit pathways (QR_{RPi}) are shown in Table 19. Using Equation 10, it is possible to estimate the aggregate recycling quality of textile waste in the EU.

Table 19: Calculation of the quality of recycling of textiles in Europe.

	2019	
Pathway	W_{RPi}	QR_{RPi}
Closed-loop	31%	0.213
Open-loop	69%	0.184
$QR_{textiles}$	0.193	

The lack of publicly available data highly conditioned the implementation of the VCM in the waste textiles case study. The results from an extensive study focused on the techno-scientific assessment of the management options for used and waste textiles in the EU, developed by the JRC, were used, which allowed for a one-shot assessment (reference year was 2019).

The quality of recycling scores for mechanical recycling of textiles are significantly low, mainly due to the low separate collection rate and losses in the functionality and material properties of the recycled materials. From a policy-making perspective, the environmental performance results suggest that the focus should be on expanding the lifespan of textile products and reuse strategies.

6.5 Limitations and uncertainty of case study

Evaluating the quality of textile waste recycling faced several challenges in choosing recycling pathways and populating the pathways ultimately chosen due to the inherent complexity of the waste fraction and the existence of publicly available data or lack thereof.

First, the lack of data on the weight of chemical recycling in the recycling market hindered this recycling technology from being included in the defined recycling pathways. Chemical recycling employs various chemicals allegedly producing a recyclate with a higher looping potential (Köhler et al., 2021). Although many sources are referring to the anticipated large-scale entry of the technology on the market (Dahlbom et al., 2023; Hedrich et al., 2022; Köhler et al., 2021; Policy Hub, 2021), it has not been possible to identify the concrete volumes. Given that recycling processes lead to the production of new material, production statistics from ProdCom were examined for data on output quantities for different recycling (processing) technologies. The manufacturing of textiles is chapter 13 in ProdCom, but none of the subchapters refer to recycled textiles (Eurostat, 2023), which was confirmed through direct communication with Eurostat. Textile recycling is considered a form of waste treatment and not production activity and is therefore not registered in ProdCom. Several studies estimate the recycling capacity of chemical recycling and mechanical recycling (European Commission, 2021; Hedrich et al., 2022; Huygens et al., 2023), but as the results are highly divergent, the methods unclear, and capacity cannot be considered representative of actual and real production output, these sources could not be used.

Production numbers were sought obtained through direct inquiry with recyclers (both mechanical and chemical) from the industry overview provided by Fashion for Good through the Sorting for Circularity project as well as EuRIC – the European Association for Recyclers, but few responded, and fewer provided useful data. Consequently, it was concluded that correct data for volumes of chemical recycling of textile waste could not be obtained within the project's scope.

Second, the range of sources, using different definitions and methodologies, presents a significant challenge in harmonising data for and across the different steps of the value chain in the chosen recycling pathways. This is further aggravated as the efficiency score is based on the share of textile waste going from one step to the next, and thus, the data for each step needs to be consistent and correspond with each other. For instance, Eurostat estimates that 1.4 million tonnes of textile waste went to recycling in 2020 (ETC CE, forthcoming), and JRC estimates a recycling capacity of 135.000 tonnes and 15.000 tonnes for respectively mechanical and chemical recycling (Köhler et al., 2021). The divergence of the data makes it challenging to rely on different sources for the different recycling steps.

Data for the different value chain steps were also somewhat limited and constituted uncertainty.

1. Total generation of textile waste:
    a. Huygens et al. (2023) estimate 10.9 Mt of post-consumer textile waste generated in 2019. The ETC CE report on Textile Waste Management (ETC CE, forthcoming) on the other hand cite an estimate of 6,93 Mt from Eurostat for 2020. Both include separately collected textiles and textiles present in the mixed household waste. The reason for the divergence has not been identified, but could come from the different definitions of textile waste across the EU. Nonetheless, this shows a significant uncertainty with the data. As statistical and dynamic sources are to be prioritised, data from Eurostat would be preferrable. However, Eurostat's data on "Generation of waste by waste category, hazardousness and NACE Rev. 2 activity (ENV_WASGEN)" includes all economic activities and waste generated by households and could hence not be used as the model only considered post-consumer waste (ETC CE, forthcoming).
2. Collection: Huygens et al. (2023) estimate that 2.44 Mt of textile waste was separately collected in 2019; (ETC CE, forthcoming) on the other hand estimates that 1.95 Mt was collected in 2020, again representing a significant gap and consequently an uncertainty of the data.
3. Sorting: Data on the share of different types of sorting i.e. manual or automatic sorting was not found. More detailed data on this step could provide valuable information for calculating environmental impact as automatic sorting requires technology and thus electricity which impacts its environmental performance.
4. Recycling:
    a. Eurostat provides data on the waste treatment of textile waste but does not differentiate on recycling method and it is unclear whether these numbers include the amounts of textiles donated to a non-profit or charity organisation of which a share also end up being sent to recycling, as the definition of textile waste vary amongst the Member States (ETC CE, forthcoming). This constitutes a lack of a statistic and dynamic data source, limiting the possibility to provide a time series for the quality of recycling.

b. A second limitation is the lack of data and information on the management of textile waste that is exported out of Europe. Trade data shows that the majority of exports are “used textiles” meaning that they are intended for reuse. Anecdotal evidence however, shows that there is a risk that a part of these export are not reused but end up as waste in landfill or in nature (ETC CE, 2023). Another part of the export is “rags and scraps” which are destined for recycling, but it is not known how much is in fact recycled as opposed to incinerated. As the model includes the treatment of textile waste that happens outside of Europe, the uncertainty of this data renders the final score uncertain. If the quantities exported for recycling according to the JRC study (Huygens et al., 2023) were disregarded for the assessment of efficiency under the scope of quality of recycling, then the overall score for waste textiles would be lowered to 0.153, representing a decrease on quality of 21%.

c. The efficiency of the recycling is set to be a 100 %, meaning that the process produces no waste, based on the estimations from Huygens et al. (2023). This is however unlikely, and more detailed data would be preferrable.

5. Application: The JRC study (Huygens et al., 2023) provides a share of recycled material going to either open-loop or closed-loop recycling, providing input to different applications. This is based on assessments of the capacity of different recycling technologies to produce “spinnable fibres”. This is however an estimation which is presumably generalising across recycling companies and their specific technologies. The share between outputs can be further specified if more precise data was available.

To counter the data limitations and uncertainties and to enable a time series for the quality of textile recycling going forward, the following key aspects should be addressed:

- Data on generated textile waste and separately collected textiles should be transparent, similarly registered, and reported by all EU member states.
- The share of manual and automatic sorting for the environmental impact estimation is needed to deploy the VCM in its unfolded form.
- Recycling activities should be seen as production (as opposed to waste management) and registered in ProdCom to obtain data on the production volumes of textile recyclates.
- The ultimate treatment of textiles exported out of Europe should be closely monitored and recorded.

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7 Conclusions

The project's purpose has been to develop a suitable and readily operational metric to assess the quality of recycling for a given material, currently and in the recent past, and to test its application on three representative waste material fractions in the EU. The present report presents the metric and the results of three pilot tests on recycling pathways, thereby providing and documenting an option for measuring quality in recycling over time.

The proposed metric should be seen as a pilot model, heavily inspired by the work of the JRC, but with a strict focus on practical application in a context of lacking and insufficient data. It has demonstrated usefulness in assessing the quality of the investigated recycling pathways, but it cannot address every requirement. The tool must grow from the basis presented in this report.

The main challenges with the measurement of quality in recycling include:

- There is a lack of historical data to calculate dimensions/parameters for the quality of recycling (usually one-shot studies). The level of detail of market data does not always allow differentiation of recycling pathways, especially emerging pathways.
- Despite the number of studies focusing on the environmental performance of individual recycling pathways, these are usually not comparable. Data on applying the recyclates are, to a vast extent, based on expert consultations and are fraught with uncertainty. There is a need for standardised and systematic data collection covering the complete waste material value chain.
- An LCA study is time- and resource-consuming, so the existing body of relevant LCA studies should form the basis for the environmental assessment – with due consideration of the system boundaries, functional unit, and methodological approach. The LCA standards leave room for interpretation and, consequently, in the assessment results. Harmonised approaches to the allocation of environmental burdens and the definition of the substitution factor for a specific recyclate are of significant importance for the future use of the methodology.
- Climate change is considered the midpoint impact category in most LCAs but may not be the most relevant impact category for specific waste material fractions. The proposed model can be expanded to include other impact categories, accentuating data quality and comparability issues.

The challenges that were identified can be addressed in follow-up projects. The first recommendation is to establish a limited number of materials to be monitored and reported by EEA. This will help establish more focused projects to address specific data limitations and improve information flows from Member States and businesses. These projects can be crossed with other priority areas for the EU, such as the definition of best available techniques for waste management, the certification of recycling processes or even the improvement of tools such as the EWC codes list.

The model can also be improved by further testing, i.e. with other waste streams, but it is highly recommended to disseminate the model and foster its adoption and testing. For example, environmental agencies from each Member State can actively apply the model, resulting in information flowing from the national to the European level. Ultimately, this will help to increase the number of practitioners and projects adopting the model and provide feedback on how to improve the model.

Main takeaways of PET

- The quality of recycling has increased due to increased efficiency and the weight of bottle-to-bottle recycling.
- It can continue to increase with a shift towards DRS, which has increased efficiency.
- To improve robustness, it will be necessary to address data gaps such as the weight of DRS in Europe and the environmental performance of DRS (which might take many forms).

Main takeaways of bio-waste

- Data suggests that recycling quality has slightly improved due to improvements in each recycling pathway.
- Contrary to other recycling systems, where the concept of recycling is obvious, biowaste recycling can be considered as closing the nutrient cycle.
- The benefit of recovering energy is still captured in the environmental performance, but that benefit is diluted in the other dimensions.
- Significant uncertainty is associated with nutrient composition, availability, etc., which need to be addressed before applying the method for policy analysis.

Main takeaways of textiles

- The quality of recycling scores for mechanical recycling of textiles are significantly low, mainly due to the low separate collection rate and losses in the functionality and material properties of the recycled materials. There are significant knowledge gaps in this recycling system, especially considering mass flows and the final steps of the chain.
- Available information supports the conclusion that the recycling quality is still low due to low collection efficiency, low loop potential (i.e. keeping materials in loops) and low environmental benefit.
- Results on environmental performance suggest that policies should focus on extending the lifespan of textile products & reuse strategies.

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Annex I – Summary of project webinar

On January 26th, 2024, the project team organized a webinar in collaboration with the EEA to present the preliminary results of the project. The webinar included an overview of the project and an explanation of the developed framework for quantifying the quality of recycling of waste material fractions. The session was attended by 110 participants, including experts from the value chains of the materials analysed as case studies for testing the implementation of the framework, namely PET packaging, municipal bio-waste and post-consumer textiles.

The webinar was held according to the following program:

Times	Title
09:00 – 09:05	Opening remarks (EEA)
09:05 – 09:35	Presentation of framework for assessing quality of recycling
09:35 – 10:00	Presentation of the testing with three waste streams
10:00 – 10:05	Break
10:05 – 10:40	Parallel sessions for each stream
10:40 – 10:55	Discussion of policy implications
10:55 – 11:00	Closing remarks

Three parallel sessions, one for each case study, were held in the webinar, with the aim to present and discuss the preliminary results with experts and stakeholders. The collected contributions from the participating entities have been considered and incorporated into this technical report. In addition to the contributions gathered during the sessions, participants also had the opportunity to provide further input later on the Miro platform shared during the webinar.

The following were the main takeaways of the webinar:

- There was a recognition of the importance of assessing the quality of recycling of different waste streams at the European level with a standardised tool and methodology;
- The need for a collaborative and multidisciplinary approach to data sharing at the European level for the development of the quality recycling assessment metric was acknowledged;
- Challenges associated with incorporating particular recycling pathways were recognised, especially when these pathways involve emerging technologies for which data availability is still limited. Thus, the need for an adaptable and flexible assessment framework was emphasised;
- It was acknowledged that the data collected always has some uncertainty associated with it, as it is consolidated data on a European scale. Therefore, the importance of validation with the responsible entities was highlighted.

An important output that resulted from the webinar was the adjustment of the weight of the dimensions for assessing the quality of recycling: Efficiency, Loop and Environmental Performance. In the initial phase, the project team assigned equal importance to the three dimensions. The participants were consulted on the relative importance of each of the defined dimensions through the use of the Mentimeter platform and indicated that the environmental performance dimension should have more weight in relation to the two other dimensions when assessing the quality of recycling (Figure 10).

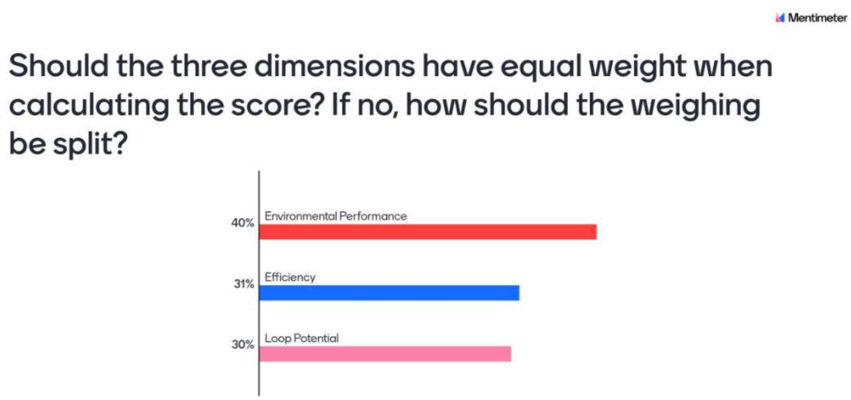

Figure 10. Mentimeter results from the webinar participants concerning the weights of the dimensions of the quality of recycling framework