# Review of economic and socio-economic studies on recycling topics

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**Abstract**

The New Circular Economy Action Plan (Commission, 2020) mandates increasing recycling rates for several waste streams, e.g., plastic waste or municipal biowaste. There are several ways by which these waste streams could be recycled, each with a different environmental and socio-economic footprint. To inform the assessment of different options in a comprehensive and efficient way, the socio-economic literature on recycling pathways will be reviewed systematically in this report. Three main findings emerge: Firstly, market failures are common in waste management given the particularities of waste markets, as for example, the fact that waste might have very low or negative prices. Secondly, the approaches and methods used to assess the economic and socio-economic impacts of recycling policies and pathways found in the literature are multiple and diverse. Thirdly, the number of publications assessing the economic and socio-economic impacts of municipal food/bio-waste and municipal dry recyclables (priority on plastics) in an EU-context are not enough to be able to generalize their findings.

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# 1 Introduction

The New Circular Economy Action Plan (Commission, 2020) mandates increasing recycling rates for several waste streams, e.g., plastic waste or municipal biowaste. There are several ways by which these waste streams could be recycled, each with a different environmental and socio-economic footprint. To inform the assessment of different options in a comprehensive and efficient way, the socio-economic literature on recycling pathways will be reviewed systematically in this report.

The objective of the present study is carrying out a systematic review of research and studies on the economic and socio-economic impacts of different recycling policies and pathways.

Three questions shall be answered by the review:

1. What are the theoretical arguments in favour of recycling, as opposed to, e.g., energy recovery or landfill? Is there a need for intervention to "correct" a market failure? Under what conditions is no (or too little) recycling a market failure? What types of market failures are reported in the literature? What are the proposed solutions to market failures?
2. What formal approaches and methodologies are used to assess the economic and socio-economic impacts of different recycling policies and pathways?
3. What are the main findings from currently available economic and socio-economic studies for the following specific waste streams: (i) municipal food/bio-waste and (ii) municipal dry recyclables (priority on plastics)?

This is not the first review of this kind: several review articles can be found in the literature regarding methods to assess solid waste management options (Allesch and Brunner, 2014; Bonanomi et al., 2017; Campitelli and Schebek, 2020; De Menna et al., 2018; Finnveden et al., 2007; Medina-mijangos and Seguí-amórtegui, 2020; Parthan et al., 2012; Simões and Marques, 2012). Even though none of them completely addressed all the questions posed here, they have been reviewed and their main insights are included in this report.

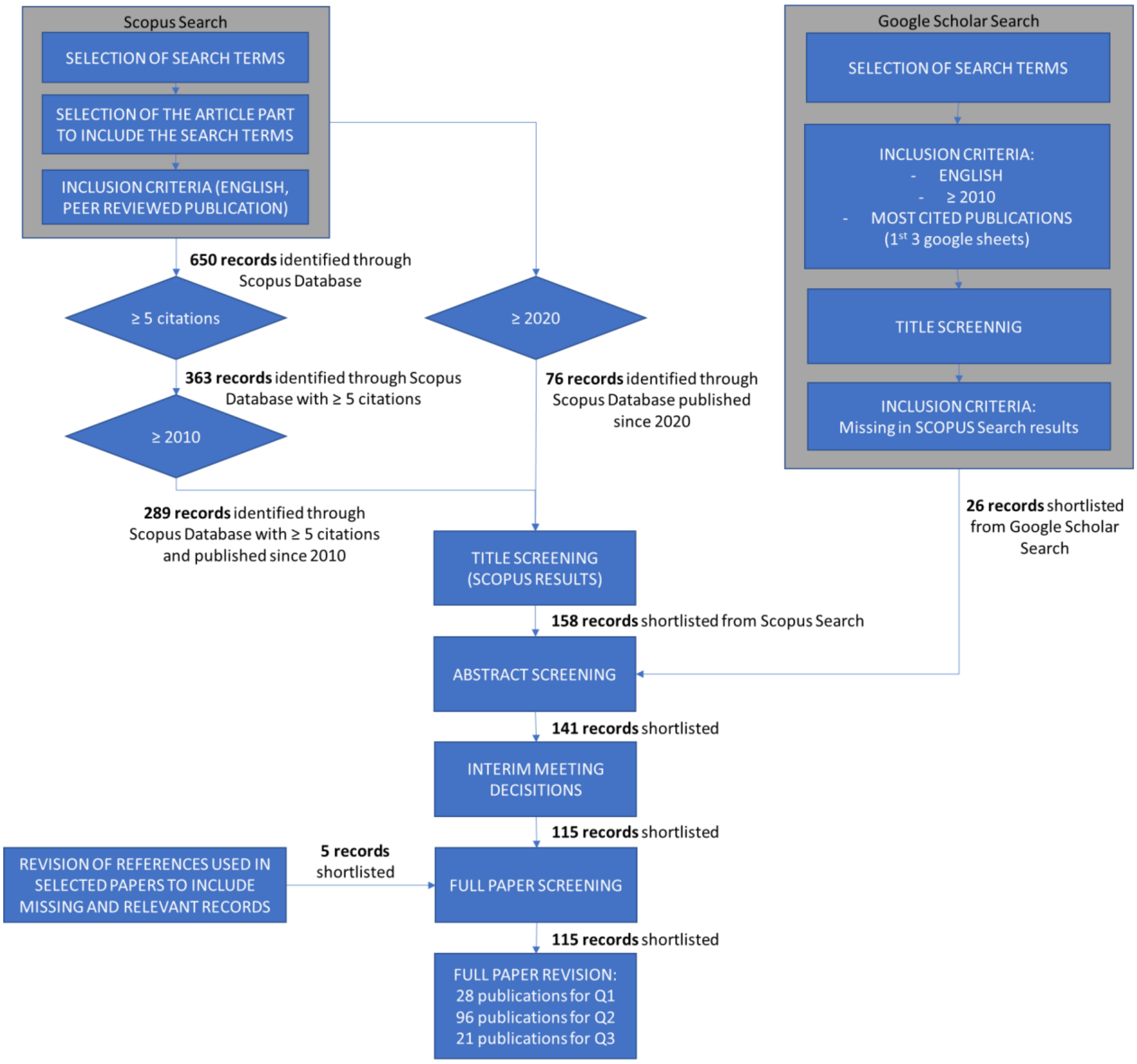
In Section 2, the methodology used in the study is described. The following sections address the three abovementioned questions of the reports, i.e., Section 3 addresses question 1, Section 4 question 2 and Section 5 the third and last question. Finally, Section 6 summarizes the most important findings of the study.

# 2 Methodology

As question 1 is more theoretical and general in scope than the rest, since it is not limited to Municipal Solid Waste (MSW), a systematic review was performed to answer questions 2 and 3 and a more narrative review was carried out to answer question 1. However, the systematic review for questions 2 and 3 was also used to identify some publications to reply to question 1.

Figure 1 shows an overview of the screening process used to identify relevant publications. The details of all the publications identified, rejected, and shortlisted in the different steps of the literature review strategy can be found in the excel file accompanying this report (available upon request).

Figure 1. Overview of the screening process to identify relevant publications to be reviewed.




Source: Own Elaboration

## 2.1 Scopus search

In the search done in the Scopus Database, it was necessary to choose a wide variety of terms, mainly due to the lack of standardization in the names of the methods used to estimate socio-economic impacts.

At the same time, it was necessary to narrow the search to the title of the publications to get a feasible number of publications for screening. Annex 1 shows the results obtained when searching different terms in different parts of the articles.

In addition, the inclusion criteria used in Scopus were: 1) Peer reviewed publications and 2) published in English.

When analysing the first set of results, there were many publications from the energy sector focusing on waste heat recovery, without relation to solid waste management. To remove these articles from the search, articles with "waste heat" in the title were excluded.

The search string used in Scopus was:

*TITLE ( waste ) AND TITLE ( economic AND impact OR benefit OR potential OR ass essment OR analysis OR evaluation OR estimation ) OR TITLE ( "Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs" ) OR TITLE ( "Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA" ) OR TITLE ( [ "socio-economic" OR "socioeconomic" ] AND [ AND impact OR benefit OR potential OR " assessment" OR "analysis" OR "evaluation" OR "estimation" ] ) OR TITLE ( "welfare " AND "economic" AND [impact OR benefit OR potential OR "assessment" OR "an alysis" OR "evaluation" OR "estimation" ] ) OR TITLE ( financial AND [ AND impact OR benefit OR potential OR "assessment" OR "analysis" OR "evaluation" OR "esti mation" ] ) AND NOT TITLE ( "waste heat" ) AND ( EXCLUDE ( DOCTYPE , "cp" ) OR EXCLUDE ( DOCTYPE , "no" ) OR EX CLUDE ( DOCTYPE , "er" ) ) AND ( LIMIT-TO ( LANGUAGE , "English" ) ) AND ( EXCLUDE ( LANGUAGE , "Russian" ) OR EXCLU DE ( LANGUAGE , "Ukrainian" ) )*

Due to the large number of records obtained, we decided to limit the shortlisted papers to the most cited (more than 5 citations) and recent ones (published since 2010). As the citation filter could penalise recent publications, we decided to skip this exclusion for papers published since 2020.

## 2.2 Title screening (Scopus' results)

In the title screening of the publications identified through Scopus, 207 publications were excluded. The main reasons for exclusion were:

- No MSW – The focus of the publication was not on Municipal Solid Waste (MSW). This was the case for 170 publications whose focus was on waste different than MSW such as: agricultural waste, industrial waste, construction and demolition waste, hazardous waste, medical waste, etc.
- No economic evaluation - The publication did not include neither review socio-economic impacts of waste management options. This was the case for 21 publications.
- Duplication – 8 publications appeared duplicated in the Scopus database.
- Method/Process too specific/novel – 8 publications were considered as too specific regarding either the cost evaluation method or the waste process/technology.

After the title screening of the Scopus results, 158 records were shortlisted for the next step (i.e., abstract screening).

## 2.3 Google Scholar search

Google Scholar was used as a complementary database to verify that the most cited publications on the topic of research were included in the Scopus search results. This search was done using 3 search terms:

- Costs of waste recycling
- Costs of waste management

— Economic impacts of waste

In Google Scholar, records found were ordered by importance (i.e., number of citations) and only the first three pages of results gathered for each of the 3 search terms were screened. The screening was mainly done on the title content and year of publication. Regarding the year of publication, as in Scopus, the shortlisted publication needed to be published since 2010. An exception was applied to one publication that based on the title seemed relevant for the 1st question of the project.

## 2.4 Abstract screening

The abstract screening was done separately for publications identified through Scopus and Google Scholar.

In the abstract screening of the publications identified through Scopus, the reasons for exclusion in this step were:

— Too specific method – 23 publications were rejected because either the economic evaluation method or the waste process/technology was too specific.

— No MSW – 6 publications were rejected because from the abstract it could be understood that the focus was not on MSW.

— No economic evaluation – 5 publications were rejected because from their abstract it could be understood that socio-economic impacts of waste management were not estimated nor reviewed.

— Minor role of economic evaluation – 1 publication was rejected because the focus on the socio-economic impacts was minor.

In the abstract screening applied to the publications identified through Google Scholar, the reason for exclusion was:

— No Economic evaluation – 5 publications were rejected because from their abstract it could be understood that socio-economic impacts of waste management were not estimated or reviewed and 2 publications were rejected because they were an editorial (Da Cruz et al., 2014b) and a conference proceeding (Bernardo et al., n.d.).

## 2.5 Final selection of articles

Due to the large number of publications shortlisted after the abstract screening, it was decided together with the JRC to exclude some publications whose focus was less relevant for the present study (31 publications were excluded). In addition, the reference list of all the selected studies was reviewed to identify potentially relevant missing records (5 publications were added to the shortlisted list). Subsequently, during the screening of the full publications, 5 publications were excluded, as they did not focus on Municipal Solid Waste (MSW), their methods were not relevant for the study or because the full paper could not be accessed.

Finally, the literature review included 115 records: 28 records with potential interest for question 1, 96 records relevant for question 2 and 21 records for question 3.

For question 3, the focus was exclusively on four specific waste streams of MSW, namely: the organic fraction of MSW, municipal food waste, packaging waste and plastic waste.

— The **organic fraction of MSW** is defined as the biowaste collected and treated by or on behalf of municipalities. Biowaste is defined as "biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. It does not include forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles, paper, or processed wood. It also excludes those by-products of food production that never become waste."[1]

— **Municipal food waste** is defined as the food waste collected and treated by or on behalf of municipalities. It is a large part of the organic fraction of MSW.

— **Packaging waste** is defined as "any packaging or packaging material covered by the definition of waste laid down in Article 3 of Directive 2008/98/EC, excluding production residues"[2]. Common packaging materials are plastic, metals, carton beverages and glass.

— **Plastic waste** is defined as waste originated from plastic objects, including packaging and non-packaging items.

While the methodologies used for the economic and socio-economic assessments in a non-EU context are of relevance for this project and have been therefore included in the answer to question 2, the main findings of the publications regarding the results obtained with the economic assessment methods were not used to answer question 3, since their applicability to EU conditions could not be generally ensured.

This exclusion applied to:

— 8 publications on the **organic fraction of MSW** including cases from Turkey (Yalcinkaya, 2020), USA (Lee et al., 2020), Bangladesh (Mia et al., 2018), Vietnam (Otoma and Diaz, 2017), Brazil (Gaeta-Bernardi and Parente, 2016), Malaysia (Bong et al., 2017; Lim et al., 2016), and UK (Yang et al., 2018).

— 12 publications on **food waste** whose cases were based in USA (Chen et al., 2015; Franchetti, 2013; Mu et al., 2017), Mexico (Chan Gutiérrez et al., 2018), South Korea (Kim et al., 2011), Singapore (Ahamed et al., 2016; You et al., 2016), Japan (Koido et al., 2018; Takata et al., 2012), China (Xuan-feng et al., 2019) and Australia (Edwards et al., 2018),

— 3 publications on **plastic waste** in India (Choudhary et al., 2019), China (Zhang et al., 2020) and USA (Gracida-Alvarez et al., 2019).

1 https://ec.europa.eu/environment/topics/waste-and-recycling/biodegradable-waste_en

2 Directive (EU) 2018/852 of the European Parliament and of the Council of 30 May 2018 amending Directive 94/62/EC on packaging and packaging waste (Text with EEA relevance). https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:32018L0852

# 3 Theoretical arguments in favour of recycling

## 3.1 Existence and identification of market failures

In neoclassical economics, market failures occur when the outcome of an economic allocation of goods and services is not Pareto-optimal under welfare economics basic principles: *"(1) if there are enough markets, (2) all consumers and producers behave competitively, and (3) an equilibrium exists"* (Arrow and others, 1951). Therefore market failures arise when some of the reference conditions of a perfectly competitive and complete market are not fulfilled since, as formulated by Bator (1958), it is "*the failure of a more or less idealized system of price-market institutions […]*". Bator also points out that *"[m]any things in the real world violate such correspondence: imperfect information, inertia and resistance to change, the infeasibility of costless lump-sum taxes, businessmen's desire for a "quiet life," uncertainty and inconsistent expectations, the vagaries of aggregate demand, etc."*.

Typical market failures are often associated with:

- **Public goods**: when goods are non-rivalrous and non-excludable, prices might not reflect the costs and benefits to the agents involved in the economic transaction, leading to non-optimal allocations of resources.
- **Price controls**: some goods and services might be subject to price controls either through minimum or maximum prices, which might be above/below equilibrium.
- **Information asymmetries**: information is basic to assess prices both on the production and consumption sides. Information asymmetries might lead to a reduction/increase of benefits/cost which can result in non-optimal allocations.
- **Non-competitive markets**: There might be market conditions where free competition cannot be achieved because either a reduced number of producers or consumers have the power to influence prices (monopoly, oligopoly, monopsony, oligopsony).
- **Externalities**: when costs or benefits involved in an economic transaction are borne by third parties not involved in such transaction (e.g., pollution), resource allocation is not efficient.

Other types of market failures would be conflicts of interest (Haubrich, 1994), factor immobility (Casas, 1984), time-inconsistent preferences (Hoch and Loewenstein, 1991) or resource failure (Karp, 1993).

Market failures, and particularly externalities, occur for a great variety of goods and services, among which, solid and other types of waste are a typical case: "*manufacturers are usually given no incentive to consider the ultimate disposal cost or recycling cost of their products and packages, and the result is products and packages that are too heavy, too complex, and too difficult to recycle.[H]ouseholds are usually given no incentive to consider the disposal cost or recycling cost of the solid waste they generate, and the result is too much waste, too little reuse, and too little recycling"* (OECD, 2004). "*Factors such as information failures, technological externalities, and market power can affect the prices, quantity, and quality of [recycled] materials traded. Ultimately, such market failures and barriers can even undermine the development of the market entirely*" (OECD, 2006).

As seen from a more heterodox point of view, a market failure is just evidence of the maladjustment of non-empirical economic models based on doubtful axioms about human behaviour failing to be validated in real life (Naredo, 2015). In particular, as exposed by ecological economists, conceiving the economic process as a socioeconomic metabolic system, what welfare economics takes as sporadic market or government failures can be considered a systemic part of the economic process itself (Martinez-Alier, 2003).

In turn, from an economic point of view, waste can be conceptualised as an unintended while inevitable physical output resulting from production and consumption[3]. These outputs are varied in their form and value, ranging from materials that may still contain some utility for other companies (i.e., with positive value) to unprofitable materials with a high harmful potential for health and the environment.

Since the accumulation of waste either for firms or households is physically impossible and entails disutility to their holders, waste needs to be "managed", which triggers both the supply of waste and the demand for waste management services. In the context of market economies, some of this waste can be placed on fully operational markets, particularly that with high positive value, e.g., metals, since waste will work as a secondary raw material. Whereas other materials will not be subject to such markets even if they have positive value. In both cases, waste can be subject to market failures.

It has to be mentioned that, where waste producers are subject to specific types of management involving private costs and/or public charges, they have an incentive for illicit activities to externalise these costs (Müller, 2019), which in turn may interact with criminal activities (INTERPOL, 2020). Criminal activities related to waste management occur because some types of waste, particularly those of low or negative value, entail a cost for waste producers because of the legal obligations for proper treatment along with taxes/charges. Skipping these costs implies savings/increasing profits for waste producers. One example would be the illegal exports of plastic waste from the EU to Asian countries (INTERPOL, 2020). The disposal cost of low-quality plastic waste in the EU following sound environmental practices ranges from 100 to 150 €/t. However, shipping this waste illegally to Malaysia costs around 70 €/t.

### 3.1.1 **Market failures found in waste management**

#### 3.1.1.1 ***On waste as a commodity***

Waste management comprises a wide range of products and materials. Whereas some of them have common features, market conditions may significantly differ in different contexts, e.g. low income vs. high income economies (Beede and Bloom, 1995). In this section we will focus on the context of the EU, since there are several traits of the EU market that frame waste management within specific boundaries. The EU has adopted a waste hierarchy establishing a priority for management options along with targets for recycling (The European Parliament and the Council of the European Union, 2008) plus regulation to prevent landfilling (Council of the European Union, 1999). This implies that there is an explicit acknowledgement of the externalities posed by waste dumping on health and the environment. Furthermore, extended producer responsibility (EPR) schemes exist for several waste streams (e.g., packaging, end-of-life vehicles, tyres, batteries, etc.) which again acknowledges the need for waste producers to reduce waste generation and properly manage waste, whose costs would be externalised in the absence of such intervention.

It is also important to point out that the term "recycling" is often confused with "separate collection" and other terms in the economic literature. Separate collection is very often a prerequisite to recycling, but materials separately collected might not end up being recycled.

By recycling, we take the definition included in the amended version of the Waste Framework Directive (The European Parliament and the Council of the European Union, 2008): recycling "*means any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the*

3 Indeed, what we call "waste" is just the materialisation of the second law of thermodynamics (Georgescu-Roegen, 1971). In this sense waste is inherent to the economic process and to some extent inevitable, since it forms part of the socioeconomic metabolism of firms and households (Schaffartzik et al., 2014).

*reprocessing into materials that are to be used as fuels or for backfilling operations*". So, recycling implies a reconnection of materials with the production process, therefore sorting either in households or firms does not mean recycling.

Taking these aspects into account, waste management significantly differs according to several criteria. First the **physical features of waste**: waste management comprises both mono **material** waste and **products**. Products made of several materials (i.e., waste electronic and electric equipment) inevitably require disassembling and pre-treatment until they can be converted into individual materials re-introduced in a production process. Mono material waste can often be recycled either directly or through very little processing (i.e., glass, paper).

Also, its **economic value**: there are wastes ranging from high positive market value to some which are very costly to manage. For products with a positive value, differences can also be found according to waste **performance as substitutes for virgin materials** (Viau et al., 2020). For example, ferrous metals, glass, and paper[4] would be examples of waste materials that have a positive value and perform reasonably well as substitutes for virgin materials, which enhances their potential for being reintroduced in production processes. Some types of plastics such as PET have a positive value in the market, but substitution for virgin products relies on physical features such as colour, density, additives, etc. and therefore its potential for recycling might be lower than for metals.

Biowaste has a negative price though it can perform reasonably well as a substitute for inorganic fertilisers if quality is ensured in collection and treatment. The environmental impacts and costs may significantly vary according to local conditions and treatment technology (Martínez-Blanco et al., 2013) as compared to landfilling and incineration. In pecuniary terms, biowaste competes with inorganic fertilisers that are often synthetised using large amounts of electricity in turn distorted by low oil and gas prices (i.e., not including their externalities) which prevents demand for organic fertilisers from expanding. It follows that **the extent to which externalities are internalised in the prices of the potentially substituted raw materials** and along the value chain is a relevant factor, i.e., when raw materials are cheap at the expense of health and the environment, recycled materials are not competitive.

Other types of waste do not have such potential for recycling and its market value can be low and even negative, whereas some other waste has no possibility of being used in any further production processes and its value is generally negative and in the case of harmful substances its management can be costly (e.g., nuclear waste).

It must be also considered that for some materials **markets might be global** (e.g., some plastics, ferrous metals), whereas markets for other materials tend to be **local or regional** (e.g., construction and demolition waste) depending on the material nature of waste, transport costs, scaling effects and overall treatment/disposal costs.

Given these considerations it can be stated that each waste material/product has a unique context and therefore is subject to different market failures when it comes to comparing management options. In the following, some of these market failures in waste management are analysed.

#### 3.1.1.2 *Common market failures in waste management*

Among the most common market failures related to waste management, the existence of **externalities** is referred to as the most relevant and consequently is the one receiving most attention (Andersen, 2007; Chung and Poon, 1997; Eshet et al., 2007; Kinnaman, 2009; McKinsey Center for Business and Environment, 2016; Nahman, 2011). Externalities related to waste management occur, in general terms, when the full cost of waste management is not borne by waste producers. As a result, part of these costs is

4 Although for paper it took several decades to overcome market failures related to information and the perceived quality of recycled paper (Mansikkasalo et al., 2014; Mobley et al., 1995).

externalised to other agents not participating in the benefits of the waste producer. These externalities should be considered in an integrated manner, alongside the value chain from raw material extraction and transportation, industrial processing to waste management in post-consumer stages, i.e. from a life-cycle point of view (Ayres et al., 1969; Pearce and Turner, 1993).

This implies that, to the extent that externalities generated in this whole process are not internalised at optimum levels by the corresponding agents[5], prices of products and services will be subsidised at the expense of human health and the environment (Kinnaman, 2014). Furthermore, in the absence of mechanisms to correct these externalities, alternatives in the form of improved product design for recyclability, enhanced energy and material efficiency and processes increasing recycling will receive no signals from the market to be developed.

According to the point of the value chain where environmental and health impacts are externalised by households or firms, they can have several effects on waste management, leading to a decreased degree of recycling and the corresponding excess of disposal. For example, if companies extracting virgin materials do not internalise the environmental externalities of extraction (Martínez Alier and Roca Jusmet, 2013), virgin materials will not be priced correctly, therefore preventing recycled materials from being competitive against raw materials. On the other end of the value chain, if disposal rates also exclude external costs, again no economic signal is in place to stimulate separate collection and recycling. Also, when municipal waste services are charged so that only part of the internal costs of waste management are transferred to households and businesses (while skipping external costs entirely), the agents will not be incentivized towards increased source sorting and recycling rates.

Another type of market failures found in waste management is **insufficient competition**, which might occur in different contexts. For example, landfilling and incineration are relatively localised services for which there exists a certain tendency to natural monopolistic and oligopolistic situations. Another example might be the oligopolistic behaviour stemming from producer responsibility organisations in charge of EPR schemes for entire waste streams (Fleckinger and Glachant, 2010; Wang et al., 2017).

**Information asymmetries** occur in waste management in several ways. Typical examples are treatment plants (sellers) having more information on sorted materials than recyclers (buyers) which might jeopardise trust between parties when quality standards are not clearly revealed. Also, landfill users (buyers of waste management services) assuming that landfill managers (sellers) comply with regulations regarding environmental and health related issues. Other examples might include final consumers perceiving recycled products or products made of recycled materials as of lower quality as compared to traditional products (Mansikkasalo et al., 2014; Schreck and Wagner, 2017), as it was the case for recycled paper. This is also the case for some industrial supplies (e.g., recycled plastic). Also, when waste holders do not have proper information about the positive and negative impacts of products regarding waste management and recyclability (e.g., the benefits of source sorted waste or the impacts of landfilling). Furthermore, for some types of waste its content and composition can be hardly ascertained by buyers, which might be an opportunity for sellers to overprice certain materials (e.g. bales of plastic mix, or bales of PET including impurities at unknown rates). In relation to externalities, the lack of information on the external costs of waste management leads to non-optimal prices for certain products. Another case of information asymmetry might be the one described in Hu *et al* for the construction and demolition waste market in China (Hu et al., 2019). In

5 In turn, the optimal internalisation of environmental and social externalities requires measurement, which entails making decisions about boundaries, time frames and discount rates (Chung and Poon, 1997; Kinnaman, 2014). Even considering the significant contributions of LCA and LCC techniques to the systematic measurement of marginal external costs associated with waste disposal and recycling, methodological discussion remains when it comes to assign pecuniary values over environmental and social impacts (Cleary, 2009; Nahman, 2011).

this case, the asymmetry emerges from recycling companies underreporting their technological capacity to the government to receive subsidies for improving such technologies. Current studies in industrial symbiosis have also demonstrated that information asymmetries in the form of lack of market coordination prevent some types of recycling from arising in the industrial waste arena[6] (Desrochers, 2004; Paquin and Howard-Grenville, 2012).

## 3.2 Rationale for intervention and possible policy measures

The rationale for intervention in waste management from a general welfare economics point of view can be grounded in the search for economic optimality. Given the number and relevance of market failures identified in the previous section, the need for intervention in the waste management sector has been acknowledged and broadly applied in the last decades (DEFRA, 2011; Goddard, 1995; OECD, 2004; Pearce and Turner, 1993).

Intervention implies a choice of instruments to address market failures. In this sense, not all measures are fit for all purposes and it is generally highlighted that the tailoring of policy measures has to be taken into account whereas a mix of policies is often referred to as the most convenient approach in most contexts (European Environment Agency, 2005).

Therefore, in principle a cost-benefit approach would be required to consider whether intervention is required, and when this is the case, which are the most convenient instruments. It is worth noting that cost-benefit analysis is not exempt from uncertainties and that the transformation of certain phenomena into monetary values entail significant challenges in theoretical (Martinez-Alier et al., 1998) and practical (Soltani et al., 2017) terms. In practice, cost-benefit analysis forms part of a broader framework for decision-making where other criteria are also considered (e.g., equity).

### 3.2.1 Command-and-control/standards/regulation

A first type of intervention is typically referred to as **command and control** which involves the **regulation and the implementation of standards.** This means that for certain variables (e.g., recycling), legally binding thresholds or targets are established (e.g., separate collection obligation, recycling levels, management specifications, recycled content of products) and penalties are applied to those agents not meeting such targets. These measures require good knowledge of the economic and environmental functioning of the waste stream where these standards are applied, to be able to set the standards at levels where efficiency is improved, and about the reversibility or manageability of environmental/health damages. It is also true that, in the presence of uncertainty, further management guiding principles such as the precautionary principle (Funtowicz et al., 2000; Ravetz, 2004) can be applied when it comes to setting standards. In any case, this measure has the potential to avoid severe environmental damage and for this reason, it has been broadly applied to harmful substances and hazardous waste, particular in the industrial sector.

Command and control measures are relatively simple to implement and are a clear signal to agents to achieve certain environmental targets. It must be considered, though, whether standards affect all agents similarly in terms, for example, of cost of compliance. In general, it is commonly recommended to announce these types of measures in advance, so agents have time to adapt and perform the required changes timely. Furthermore, depending on the context, specific regulatory options such as technology standards might be easier to implement and more cost-efficient.

For regulatory management options two additional aspects to consider are the fact that standards might not provide clear incentives for innovation and that, as demonstrated by

6 It has been widely demonstrated that a significant number of wastes can be reinserted in the production process of other companies with none or low processing requirements and that the only barrier preventing this to occur is lack of information and coordination between the companies of a given territory. One of the most relevant examples is the industrial area of Kalundborg (Denmark): http://www.symbiosis.dk/en/

the implementation of the EU waste regulation, monitoring can become controversial unless the standards are defined in a clear and univocal manner for all agents. An example of such a failure in standards design is the definition of "recycling" in the Waste Framework Directive, which until 2020 was calculated through up to four different methods to be chosen by the Member States (Official Journal of the European Union, 2008). Plus, none of them effectively referred to the reintroduction of waste into production processes, so in practice the so-called recycling rates of the EU Member States were, to a great extent, incomparable. Furthermore, issues regarding the illegal exports of sorted waste from the EU to Asian countries, but being accounted for as recycled have arisen, and are being addressed in the last years (INTERPOL, 2020).

These types of measures are quite widespread in the European Union and involve the implementation of a regulatory framework regarding specific waste streams. Two clear examples of this type of intervention are the municipal solid waste recycling targets included in the Waste Framework Directive:

- *by 2020, the preparing for re-use and the recycling of waste materials (such as paper, metal, plastic and glass) from households shall be increased to a minimum of overall 50% by weight*
- *by 2025, the preparing for re-use and the recycling of municipal waste shall be increased to a minimum of 55 %, 60% and 65% by weight by 2025, 2030 and 2035 respectively*[7].

Also, the Landfill Directive includes a target related to the reduction of biodegradable municipal waste (BMW). With 1995 as the reference year, BMW to landfill had to be reduced to:

- 75% in 2006
- 50% in 2009
- 35% in 2016

In the light of the differing results reached by the Member States[8], it seems that despite the overall increase in what is defined as "recycling" in the WFD, the amount and format of the penalties (i.e., being applied long after the year of non-compliance) might not provide enough incentive to enforce such standards. Furthermore, the monitoring issues related to these regulations (e.g., the reporting of data on recycling for waste streams subject to EPR schemes relying on private firms) suggest that this aspect is key for the efficacy of command-and-control intervention to be reliable.

### 3.2.2 **Market based instruments**

Market based instruments (MBIs) entail intervening in the formation of prices (e.g., applying taxes) and quantities (e.g. tradable permits) with the intention of signalling agents towards specific behaviours (e.g., diverting waste from landfilling/incineration). Applying MBIs is particularly efficient when price signals are passed through the value chain. MBIs are recommended when external costs to individuals and firms cannot be ascertained, as well as if these costs are not uniform across agents. For certain MBIs, there are specific requirements such as a minimum number of agents for trading schemes. considering the fixed transaction costs involved in the creation of such institutions.

As taken from the European Environment Agency definition of MBIs applied to waste management (European Environment Agency, 2005), the following instruments are described:

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7 https://ec.europa.eu/environment/topics/waste-and-recycling/waste-framework-directive_en. It should be noted that the definition of recycling is more stringent for targets after 2020.
8 https://ec.europa.eu/eurostat/web/products-datasets/product?code=sdg_11_60

- **Tradable permits**: tradable permits are designed to achieve reductions in pollution (such as emissions of CO2) or use of resources (such as fish quotas) in the most effective way through the provision of market incentives to trade. There are several experiences on tradable permits related to waste management. For example, Calaf-Forn et al (Calaf-Forn et al., 2014) describe the Landfill Allowance Trading Scheme in England. Under this scheme, a limited and decreasing number of tradable permits for the landfilling of biowaste was allocated to local authorities, which were allowed to sell and transfer the permits between them. This instrument was implemented to ensure compliance with the Landfill Directive's targets on the landfilling of biowaste.
- **Environmental taxes/levies**: this instrument increases prices and thus signals the behaviour of producers and consumers, while raising revenues. Examples of this instrument would be levies imposed on the raw materials ideally related to the potential for recycling and reuse and to the environmental damage caused by these materials (Söderholm, 2011, 2006). Levies can also be applied to specific products as in the case of plastic bags (Anastasio and Nix, 2016) or the recent taxes on single use plastics (Walker et al., 2020). Landfill and incineration taxes are among the most common instruments of this kind (Martin and Scott, 2003; Mazzanti et al., 2009; Morris et al., 1998). By taxing the landfilling of waste, externalities are to a certain extent internalised making alternative management options (e.g., recycling) more competitive. A particularly interesting case is earmarked landfill taxes devoted to recycling programmes, as is the case for Catalonia (Spain), where part of the revenues from municipal waste landfill taxes are refunded to the municipalities in order to develop separate collection programmes (Puig-Ventosa, 2004; Puig-Ventosa et al., 2012). These schemes have also been applied in Latvia, Flanders and Poland (European Environment Agency, 2005).
- **Environmental charges** are designed to cover the costs of environmental services and abatement measures such as wastewater treatment and waste disposal. Charges can be also designed as to provide incentives for separate collection, as it is the case for "pay as you throw" charging schemes. Under these schemes, households and businesses are charged according to the volume/weight of the different waste streams, establishing differentiated fees for separately collected streams as compared to comingled waste. By making the collection of mixed waste more expensive, separate collection is incentivised (Bonelli et al., 2016; Elia et al., 2015; Huang et al., 2011; Puig Ventosa, 2008).
- **Environmental subsidies** are incentives designed to stimulate the development of new technologies, to help create new markets for environmental goods and services, to encourage changes in consumer behaviour and to temporarily support achieving higher levels of environmental protection (European Environment Agency, 2005).
- **Mixes of taxes and subsidies (e.g., deposit-refund schemes, earmarked taxes):** The consumer is given the right to a refund if the waste product is returned to the seller, through an authorised recycling/reuse point. This right is achieved by paying a deposit at the time of the purchase of a given product. These schemes are typically applied to beverage bottles, batteries, tires, motor oil, electronic equipment, etc. (Kulshreshtha and Sarangi, 2001; Walls, 2013). Whereas some authors describe these types of deposit refund schemes and advance disposal fees as mixes of taxes and subsidies (Pearce and Turner, 1993) these are also typically described within extended producer responsibility instruments (OECD, 2016a).
- **Extended producer responsibility:** Extended producer responsibility (EPR) has been applied since the early 90s in industrialised countries as a measure for improving and correcting waste management policies (European Commission, 2014; Hanisch, 2000; OECD, 2016b, 2016a; Sachs, 2006; Walls, 2006)**.** The objective of EPR should be maximising social welfare by both reducing the amount of waste and the impacts derived from waste management (Walls, 2006). EPR

schemes acknowledge the either physical or financial responsibility of waste by firms, thus implementing the "*polluter pays principle*" in practice. Within EPR schemes, producers should organise either individually or collectively, so that proper separate collection and recycling for a given product (e.g., packaging waste) are provided. These services are financed by collecting payments from producers of the commodity subject to EPR, according to the amount and other features of products (e.g., recyclability) placed in the market. EPR is a very widespread instrument in the OECD, and more specifically in the EU where EPR applies to packaging waste, batteries and accumulators, tires, end-of-life vehicles and waste electric and electronic equipment (European Commission, 2014) among other products.

- **Liability and compensation schemes** aim at ensuring adequate compensation for damage resulting from activities dangerous to the environment and provide for means of prevention and reinstatement (Wilde, 2002).

### 3.2.3 Other instruments

Beyond regulatory and market-based instrument, other relevant tools are available for addressing market barriers and failures.

For example, in the case of information asymmetries, **information and awareness raising campaigns** (DEFRA, 2011) can be useful when it comes to ensuring that all the agents have access to proper information about key issues. This tool can be useful in improving cost-effectiveness of other measures since in general, information and awareness raising are inexpensive measures as compared to other policies. This tool permits a clear exposure of key ideas, which can be delivered over time in order to provoke a medium-large term behavioural change. For example, the change in consumers attitudes towards plastic littering could be a case of an increasing success of information and awareness campaigns (Adeyanju et al., 2021; McNicholas and Cotton, 2019). In this sense, certification (e.g., EMAS, ISO) has made a difference for crediting environmental performance of products and companies.

**Facilitation** can be included as a specific type of information and awareness raising measure which is particularly relevant for industrial waste and industrial symbiosis processes. Facilitation has been signalled as a tool for market coordination in industrial waste contexts where industrial symbiosis is feasible and economically viable but requires a coordination agent (e.g. mediator, facilitator) to enable market opportunities (Paquin and Howard-Grenville, 2012). Through facilitating industrial symbiosis, externalities and certain information asymmetries are being overcome in the EU (Costa et al., 2010), e.g. in Catalonia, several facilitation programmes for industrial symbiosis are being currently carried out in order to close the material, energy, water and heat loops within the industrial sector. These facilitation activities include organisational skills and the sharing of data across companies for identifying and materialising opportunities for recycling. For example, there are several companies in Catalonia using an online platform called "Siner"[9], which is being used as a tool to centralise information for facilitating industrial symbiosis.

**Voluntary agreements** are also a tool for improving economic efficiency (Croci, 2005; Nunan, 1999; Rennings et al., 1997; Sairinen and Teittinen, 1999). Whereas the main advantage of this approach is the saving of regulation costs for public authorities, these agreements might entail significant transaction costs as the ambition of measures grow. Also, market power and information asymmetries could distort such processes in favour of specific firms. According to the experiences with such agreements in Germany, the weaknesses of voluntary agreements are related to goal-conformity, system-conformity, cost-efficiency and institutional controllability (Rennings et al., 1997). The authors recommend using voluntary agreements as a part of a policy mix either in parallel to

9 https://www.sinerplatform.com/

economic instruments (e.g., MBI and voluntary agreements) or accompanying command and control-policy measures.

**Public green/circular procurement** has become an instrument with an increasing relevance in the last years, particularly since the recent developments in circular economy policies have given this measure a central role (Alhola et al., 2019; European Commission, 2017; Lăzăroiu et al., 2020; Sönnichsen and Clement, 2020). Green procurement is an instrument by which public authorities introduce environmental criteria for public funds spent on supplies. This way, those companies/products accomplishing certain criteria (e.g., recycled materials content) are given a competitive advantage, therefore encouraging measures at the firm level that lead to the internalisation of environmental impacts. Examples of these criteria might be a restriction to companies holding environmental certificates (e.g., ISO, EMAS certifications)

## 3.3 Summary of market failures and intervention in waste management

As shown in this section, market failures are not uncommon in waste management given the particularities of waste markets, as for example, the fact that waste might have very low and negative price. Externalities are the most widespread market failure, although insufficient competition and information asymmetries also play a role.

In the EU, a set of interventions devoted to correcting such failures have been implemented.

Externalities are being addressed from several standpoints including command and control measures and MBIs. Command and control measures such as standards in the form of recycling targets, contents of recycled materials, maximum amounts to be landfilled and specific regulation for several waste streams (e.g. plastics) are aimed at preventing and controlling the environmental burdens derived from harmful management practices such as excessive dumping and littering, for example. These instruments have a long tradition within the EU waste regulation, as is the case with the Waste Framework Directive or the Landfill Directive.

Market based instruments are also well-known and widespread tools in waste management when it comes to correct prices and signalling agents towards the internalisation of their environmental burdens, while making recycling more competitive. Among these instruments, landfill taxes, pay-as-you-throw charges and extended producer responsibility schemes are the most popular instrument across EU countries, along with deposit refund schemes and product taxes. Landfill taxes increase the price of landfilling to take social and environmental externalities into account, so that alternatives such as improved separate collection for recycling become viable and profitable. Pay-as-you-throw has proven to be a very effective instrument when it comes to improving the quality of separate collection at source, whereas extended producer responsibility has channelled financial resources to address the separate collection and management of several waste streams, such as packaging waste, tyres or waste from electric and electronic equipment.

For its part, interventions devoted to counteracting information asymmetries such as awareness campaigns and facilitation for industrial symbiosis have been developed.

All in all, it is widely acknowledged that in waste policy there is not one measure fitting all possible issues to be solved given the heterogeneity of waste streams and related markets. In general, a policy mix containing the most appropriate measures for each case tends to be applied (e.g. mixes of recycling targets and landfill taxes).

# 4 Methods used to assess the socio-economic impacts of waste management

This section aims at answering the second question of the project, i.e., which formal approaches and methodologies are used to assess economic and socio-economic impacts of different recycling policies and pathways?

Macroeconomic assessment tools are beyond the scope of this report, thus publications using macroeconomic simulation tools such as fixed price social accounting matrices (Campoy-Muñoz et al., 2017), flexible-price computable general equilibrium models (Philippidis et al., 2019; Rutten et al., 2013) and input-output models (Reynolds et al., 2015) are excluded. These models are used to predict effects on national economies due to changes in waste generation in terms of Gross Domestic Products, employment, food security, etc. This report also excludes publications dealing with costs functions, such as Chifari et al. (2017) and Abrate et al. (2014). This type of publications is used to estimate cost elasticities and efficiencies of the WMS.

## 4.1 Method terminology

The terminology used for the methodologies applied to assess the economic impacts of solid waste management is broad and lacks standardization. For example, sometimes the same name is given to methods including different types of cost, and vice versa. Other times, no name is given to the method used, or different names are given within the same publication. The review carried out by De Menna et al. (2018) for Life Cycle Costing (LCC) on food waste demonstrated that the standardization of LCC is still in a very early stage since there is no consensus on definitions or approaches. This lack of standardization has generated a blurring of definitions, approaches, and methods.

According to the authors of the present report, the socio-economic impacts of waste management can be assessed only in monetary terms or with methodologies integrating different types of units.

There are two main approaches that use exclusively monetary terms: 1) financial assessment (FA) and 2) welfare economic assessment. Multiple names are given to these two main types of assessment, and sometimes both are named "economic assessments". Often, when the publication combines the economic assessment with Life Cycle Assessment, the costs assessment is called Life Cycle Costing and sometimes the terminology used is aligned with Ciroth et al. (2008) and Swarr et al. (2011). These were the first two publications about Life Cycle Costing as economic assessment tools to complement LCA, they distinguish 3 types of LCCs: Conventional LCC (C-LCC), Environmental LCC (E-LCC) and Societal LCC (S-LCC). As it can be seen in Table 1, the first two type (C- and E-LCC) can be considered financial assessments, while the third (S-LCC) is more a welfare economic assessment.

Financial analysis describes the income flows of a project, i.e., flows of money between the different actors involved in a project. It only concerns money flows, and thus consequences not traded on the market, such as environmental loads, are excluded from the assessment unless they are internalized as transfers (e.g., environmental taxes). For example, (Cheng et al., 2019) discuss different options to internalize the environmental impacts of Waste Electronic and Electric Equipment (WEEE) recycling in the recycling fee that producers have to pay in the EPR scheme for WEEE in Taiwan.

The foundation of welfare economic assessment is utility. Benefits represent increases of utility and costs represent reductions in utility. The purpose of these types of assessment is calculating consequences for utility of re-allocating society's scarce resources that could be used for producing other welfare generating products. The consequences of a project should include market goods (consumption goods, production factors, produced production goods) and non-market goods (health, recreational possibilities, aesthetical values) (Møller and Martinsen, 2013). In this type of assessments, all costs and benefits are converted into monetary units.

In practice, the main difference between these two main types of methods (when assessing the same project) is the type of costs included. While financial assessment includes budget costs and transfers, welfare economic assessment includes budget costs and externality costs. On the other hand, there is a tendency to use welfare economic assessment from a society perspective, while financial assessment is used more from the perspective of one single actor of the chain.

Table 1: Two main approaches for socio-economic assessment, main name, alternative names, type of costs included and common perspective.

| | Financial Assessment | Welfare economic assessment |
|---|---|---|
| Alternative names | • Economic assessment<br>• Cost accounting/assessment<br>• Conventional Life Cycle Costing<br>• Environmental Life Cycle Costing (when done in parallel with an LCA)<br>• Financial LCC | • Economic assessment<br>• Socio-economic assessment<br>• Cost-Benefit Analysis<br>• Social Cost-Benefit Analysis<br>• Societal Life Cycle Costing |
| Type of costs included | • Budget costs<br>• Transfers | • Budget costs<br>• Externality costs |
| Perspective | • Single or multiple actors assessed individually | • Society |

Source: Own elaboration.

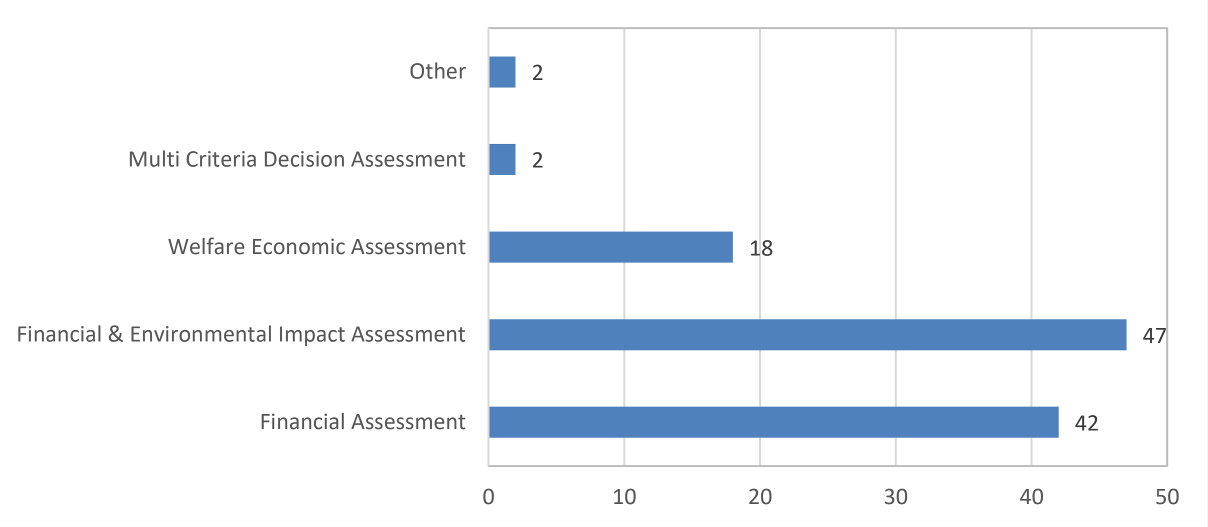
As mentioned previously, there is also the possibility of assessing the socio-economic perspective of solid waste management without converting all impacts into monetary terms, it could be an externality without its valuation. For example,

- Wohner et al. (2020) identified the most sustainable product in terms of packaging for ketchup in Austria by integrating LCA, LCC, and a third criterion (organic/conventional farming of tomatoes to produce ketchup) in a multi-criteria decision analysis (MCDA)
- Willersinn et al. (2017) complemented the quantitative assessments (including LCA and full-cost calculation) with consumer preferences between alternatives with an online survey conducted in the German-speaking part of Switzerland (1) to assess consumers' general preference for the introduced measures to reduce potato losses and (2) to analyse the factors which might influence the intensity of these preferences.
- Fernández-González et al. (2017) compared different MSW-to-Energy technologies (WtE) in Spain using Analytic Hierarchy Process (AHP) multi-criteria method that combined economic, environmental, and territorial criteria (such as public acceptance and employment generation).

Figure 2 shows the methods used in the reviewed publications based on the classification mentioned above. As can be seen, most of the publications used a financial assessment approach in combination with some environmental indicators (LCA or just GHG), followed by Financial Assessment alone. Only 18 of the 96 publications relevant for question 2 used Welfare Economic Assessment, 2 publications used MCDA and 2 publications used methods different than the rest.

It should be noted that this classification does not necessarily represent the name given by the authors of each publication to the method used. It should also be noted that publications assessing costs in different manners (such as Martinez-Sanchez et al. (2016)) are accounted for multiple times, once for each method.

Figure 2: Methods used in the reviewed publications.




Source: Own Elaboration

## 4.2 Cost bearer’s perspective

As costs (direct and indirect) can be incurred by different actors/stakeholders of the waste management systems, it is important to mention the perspective used in the assessment and the costs bearers considered, i.e., whose costs are we considering (De Menna et al., 2018). This is not always clearly mentioned in the publications. However, most of the times it can be guessed from the costs included.

For publications assessing the costs of single processes, e.g., cost of composting, often the perspective taken is the facility operator. However, when the project being assessed affects more than one stakeholder, costs incurred by different agents should be stated separately, i.e., distribution of costs among stakeholders. By including the costs and benefits incurred by the main affected stakeholders, it is possible to identify economic winners and losers and relate them to economic incentives or compensations.

For example, both Rigamonti et al. (2015) and Andreasi Bassi et al. (2020) performed an economic analysis of packaging waste in Italy, but while Rigamonti and colleagues used a single-actor perspective, Andreasi Bassi and co-authors used a multi-actor perspective. Rigamonti et al. (2015) assessed the extra-cost incurred by local authorities due to the procedures, equipment, and infrastructure necessary for the recycling of packaging waste, excluding financial transfers between the Producer Responsibility Organization (PRO), who owns the packaging waste after paying the collection fee, and the recyclers. Andreasi Bassi et al. (2020) performed an economic analysis using a multi-stakeholder approach for calculating a separate cost-benefit analysis for each stakeholder to identify potential imbalances and/or economic losses between Municipality, PRO, Material Recovery Facility (MRF) and Recyclers for 5 different packaging waste management scenarios.

Most of the reviewed publications adopted a single actor perspective, either explicitly or not, and only a few studies used multi-actors/stakeholders’ approach.

## 4.3 Cost models

In most of the cases, the publications include costs and benefits incurred by the stakeholder assessed, but there are some exceptions, such as Zaman (2016), which accounted only global benefits from recyclables sales without considering any cost associated with the collection and recovery of such materials from the waste. Ayodele et al. (2018), Burneo et al. (2020), Cucchiella et al. (2015) and De Feo et al. (2019) also limited their economic assessment to the economic benefits obtained from the sale of recyclables.

Most of the cost results are shown in relation to waste tonnage, but in publications dealing with Waste-to-Energy (WtE) options, costs are often presented per unit of energy generated, e.g., Fernández-González et al. (2017). In addition, some publications present costs in relation to GHG reduction, e.g., Wang et al. (2016) or impacts on health, e.g., Woon and Lo (2016).

To take into account the variability of cost data input in the results, few publications work with probability distributions (Andreasi Bassi et al., 2020; Faraca et al., 2019) and one with cost functions (Colvero et al., 2020). However most of the reviewed publications worked with single values cost data, e.g. Cimpan et al., (2016) and Martinez-Sanchez et al. (2015).

Some publications provided the year of the cost figure, but not all of them. Most of the publications included simple costs calculations (namely: amortization, conversion of CAPEX and OPEX to €/tonne, NPV, etc.), but others, such as Giuseppe et al. (2014) included more complex calculations related to profit optimization.

### 4.3.1 **Type of costs**

The costs included in each publication depend mainly on the system being assessed, the cost bearers and the aim of the study. But in most cases, the cost calculations included budget costs (CAPEX and OPEX), except for: 1) publications whose focus was exclusively on externality costs, such as Vlachokostas et al. (2020a), and 2) publications that included only economic benefits from recyclables sales, such as Ayodele et al. (2018), Burneo et al. (2020), Cucchiella et al. (2015), De Feo et al. (2019) and Zaman (2016). In addition to budget costs, some publications included transfers (31) and/or externality costs (14). Furthermore, some publications considered opportunity costs of waste, land, and capital (36) and a few non-monetized impacts (4).

The publications here classified as financial assessment often included budget costs and transfers, while publications classified as welfare economic assessment combined budget costs and externalities costs. In some cases, publications including only budget costs were considered FA and publications only using externality costs were considered WEA.

#### 4.3.1.1 ***Economic transfers***

Transfers (taxes, subsidies, fees, etc.) are monetary flows that represent income redistribution between stakeholders. In waste management systems, the most common economic transfers are incineration and landfill taxes/levies, waste charges, EPR fees, and feed-in-tariffs (less common in EU).

Sometimes transfers are excluded when the perspective used is society or the whole value chain, since they are monetary flows that cancel each other out if the payer and the receiver of the transfer are both included in the system. However, reporting transfers, even if they can be cancelled out, can provide interesting information about economic distribution, economic winners and losers, as done by Andreasi Bassi et al. (2020), as well as their potential to compensate extra costs of separate collection or recycling, as done in the break-even analysis of Edwards et al. (2018).

#### 4.3.1.2 ***Externality costs***

Externalities are costs or benefits arising in an economic process or transaction which are imposed on third parties who did not participate in the process or transaction. Externalities are often health-related and/or environmental but can also be related to social aspects (e.g., access to employment, public acceptance, Not-In-My-Back-Yard effects). To be included in a cost assessment they should be first quantified in non-monetary terms. Often externalities remain beyond the scope of the publication or are simply described in a narrative manner because they have not been quantified.

If they are quantifiable, they can be included in economic assessments by using conversion factors that are estimated using different valuation techniques. Eshet et al. (2006) provide comprehensive information on valuation techniques applied in relation to waste management. De Menna et al. (2018) highlighted the lack of standardization regarding monetary valuation of environmental impacts. This is supposed to be solved with the publication in 2019 of the ISO standard "ISO 14008:2019 - Monetary valuation of environmental impacts from specific emissions and natural resources"(ISO, 2020).

These externalities can also be included as non-monetary impacts. For example, (Woon and Lo, 2016) did not monetize part of the human health impacts and rather integrated these impacts in an Eco-efficiency Indicator.

In waste management systems common externalities relate to: emissions, mainly air emissions (e.g. CH4 or dioxins), but also emissions to water and soil (e.g. leachate from landfills), and disamenities (e.g. littering). Manni and Runhaar (2014) and Medina-Mijangos et al. (2020) provided comprehensive descriptions of externality costs to be potentially included in social cost-benefit analysis of waste management projects. From the reviewed publications, some externalities could be mentioned here:

- Woon and Lo (2016) considered the opportunity cost of land as externality costs.
- The case study of Martinez-Sanchez et al. (2015) considered the time used by households to sort their waste as externality costs and demonstrated that valuing this time may significantly affect the results of the Societal LCC (Welfare Economic Assessment). It can be argued if this cost can be classified or not as an externality cost. Bruvoll et al. (2000) estimated this cost in Norway and classified it as "social costs".

#### 4.3.1.3 ***Opportunity costs***

Opportunity costs is a concept often applied in project appraisal. It relates to the fact that resources are limited, and choosing a project has the consequence of not being able to use the resource for something else. The opportunity cost of a project is represented by the value of its best alternative (often maximizing profits) and it can be used to estimate costs within common markets or to value externalities.

According to Medina-Mijangos et al. (2020), the concept of opportunity cost applied to MSW systems can be applied by considering alternative ways of using the waste and, if there are no alternatives, with the opportunity costs given by a financial instrument.

Da Cruz et al. (2012) and Rigamonti et al. (2015) assessed the economic impacts of packaging waste recycling and included the opportunity costs as the costs associated with the packaging waste management in a scenario with no selective collection or sorting. The opportunity cost of recycling was accounted for as a benefit for local authorities (avoided costs).

Conversely, in the case study presented by Medina-Mijangos et al. (2020), it was considered that there were no better alternative uses for waste or for land use (industrial land); for this reason, opportunity cost accounted for was the interest earned by the use of a financial instrument, i.e. the return on the money if it had not been used for waste management.

Woon and Lo (2016) did not account for the opportunity cost of waste but did account for the opportunity costs of land used for MSW facilities in Hong Kong as externality costs.

The opportunity cost of capital is also considered in many publications to evaluate the investment in waste facilities, e.g. Guo and Yang (2019), Innocenzi et al. (2017), Mabalane et al. (2021) and Nunes et al. (2018). Even if these publications are not explicitly mentioning "opportunity costs", they do apply financial discounting rates in their calculation of indicators such as Net Present Value or Internal Rate of Return. The social and financial discount rates applied in the publications vary. For EU studies, suggested financial and social discount rates can be found in European Commission. DG for Regional and Urban policy (2014).

## 4.4 Non-monetary criteria

Within the reviewed publications, 47 complemented economic assessment with environmental impact assessments, such as LCA. In addition, a few studies complemented economic data with some social indicators, such as employment generation (Fernández-González et al., 2017; Hestin et al., 2017) and public acceptance (Fernández-González et al., 2017).

The Decision Support System (DSS) model proposed by Nie et al. (2018) to analyse the benefit of the four MSW scenarios in Pudong (Shanghai, China) combines cost benefit analysis, life cycle assessment and analytical hierarchy process (AHP). Within the AHP, the DSS accounts for three social aspects: educational level of the personnel participating in the survey, income status based on the annual household income and distance from dumping site to residential houses.

In addition, some studies using macroeconomic assessment tools, such as Philippidis et al. (2019) included other types of impacts such as effects on EU market price and food security. As these methods remain beyond the scope of this report, the impacts accounted for with this type of methods are not discussed further.

## 4.5 Methods' application

The application of the methodologies described previously varies across the reviewed publications in terms of type of question posed, waste component and waste stream being assessed, as well as type of results given.

Generally, the question posed in the publication to be answered by the (socio-)economic assessment were: "what are the current costs of the WMS?" and "what are the costs for the alternatives being considered?". In some cases, there were additional questions on the cost bearers, i.e., who incurs the costs estimated within the publication. This is the case of a group of publications focusing on EPR schemes for packaging waste, whose main question was whether industry is paying for recycling costs (Andreasi Bassi et al., 2020; Cabral et al., 2013; Da Cruz et al., 2012; De Feo et al., 2019; Marques et al., 2014; Rigamonti et al., 2015).

While some publications focus exclusively on single waste components, such as collection or treatment, others addressed more than one component together to be able to assess the waste management scheme (see Table 2). Only 3 publications addressed prevention, even if prevention has the highest waste priority. This was also concluded in the review carried out by De Menna et al. (2018) for Life Cycle Costing of food waste. A few publications assessed impacts of waste policies and mention them explicitly, as the group of publications addressing the EPR schemes of packaging waste, or Hestin et al. (2017) that evaluated impacts of achieving plastic recycling targets in the EU. This was also found in the review conducted by (Campitelli and Schebek, 2020), who recommended further research assessing waste "control and regulation" to ensure a holistic view of the assessment.

The reviewed publications included different types of results: 1) Some publications only assessed the economic performance of current treatments or pathways (e.g., (Cimpan et al., 2016; Medina-Mijangos et al., 2020)) and identified (or not) potentials for improvements, 2) other publications (in addition to point 1) compared current state vs alternative scenarios using different assessment methods (e.g., (Martinez-Sanchez et al., 2015)) and 3) some publications (in addition to point 2) made specific recommendations on how to implement potential improvements, such as policy recommendations (e.g. Andreasi Bassi et al. (2020), De Feo et al. (2019) and Mayer et al. (2020)).

Table 2: Waste component and waste stream assessed as well as the method used in the reviewed publications.

| **Waste Component** | **Waste stream** | **Financial Assessment** | **Financial Assessment &Environmental Impact Assessment** | **Welfare Economic Assessment** | **Multi Criteria Decision Assessment** | **Other** | **Reviewed Publications** |
|---|---|---|---|---|---|---|---|
| Prevention | Food waste | | X | X | | | (Martinez-Sanchez et al., 2016) |
| | | | | | | X | (Philippidis et al., 2019) |
| | | | X | | | | (Willersinn et al., 2017) |
| Collection | Packaging waste | | X | | | | (Pires et al., 2017) |
| Transportation | MSW | X | | | | | (Zis et al., 2013) |
| Sorting | Packaging waste | X | | | | | (Cimpan et al., 2016; Pires et al., 2017) |
| | | | | x | | | (Medina-Mijangos et al., 2020) |
| | Bulky waste | | | x | | | (Medina-Mijangos et al., 2020) |
| Recycling | Plastic waste | | x | | | | (Choudhary et al., 2019; Faraca et al., 2019; Hestin et al., 2017) |

| **Waste Component** | **Waste stream** | **Financial Assessment** | **Financial Assessment &Environmental Impact Assessment** | **Welfare Economic Assessment** | **Multi Criteria Decision Assessment** | **Other** | **Reviewed Publications** |
|---|---|---|---|---|---|---|---|
| | | x | | | | | (Gracida-Alvarez et al., 2019) |
| | | | | X | | | (Zhang et al., 2020) |
| | WEEE | X | | | | | (Cucchiella et al., 2015; Ghodrat et al., 2016; Innocenzi et al., 2017) |
| | | | X | | | | (de Oliveira Neto et al., 2017) |
| | MSW | | X | | | | (Ayodele et al., 2018) |
| | | x | | | | | (Diaz and Otoma, 2014) |
| | Textile | | x | | | | (Gounni et al., 2019) |
| Anaerobic (co-) digestion | OFMSW and/or food waste | x | | | | | (Aleluia and Ferrão, 2017) |
| | | | x | | | | (Chan Gutiérrez et al., 2018; Dedinec et al., 2015; Faizal et al., 2018; Franchetti, 2013; Singh and Basak, 2018; Tan et al., 2015; Yalcinkaya, 2020) |

| **Waste Component** | **Waste stream** | **Financial Assessment** | **Financial Assessment &Environmental Impact Assessment** | **Welfare Economic Assessment** | **Multi Criteria Decision Assessment** | **Other** | **Reviewed Publications** |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Composting | OFMSW | x | | | | | (Aleluia and Ferrão, 2017; Diaz and Otoma, 2014; Lim et al., 2016) |
| | | | x | | | | (Bong et al., 2017; Dedinec et al., 2015; Faizal et al., 2018; Mehta et al., 2018; Mu et al., 2017; Singh and Basak, 2018; Tan et al., 2015) |
| MBT | MSW | | x | | | | (Dedinec et al., 2015) |
| | | x | | | | | (Zis et al., 2013) |
| WtE | MSW | x | | | | | (Leme et al., 2014; Luz et al., 2015; Mabalane et al., 2021; Santos et al., 2019; Wang et al., 2016; Zhao et al., 2016) |
| | | | x | | | | (Chen et al., 2019; Faizal et al., 2018; Psaltis and Komilis, 2019; Singh and Basak, 2018; Tan et al., 2015) |

| **Waste Component** | **Waste stream** | **Financial Assessment** | **Financial Assessment &Environmental Impact Assessment** | **Welfare Economic Assessment** | **Multi Criteria Decision Assessment** | **Other** | **Reviewed Publications** |
|---|---|---|---|---|---|---|---|
| | | | | x | | | (Haraguchi et al., 2019; Vlachokostas et al., 2020b) |
| | | | | | x | | (Fernández-González et al., 2017) |
| | OFMSW | x | | | | | (Yang et al., 2018) |
| | Green waste | x | | | | | (Abdoulmoumine et al., 2012) |
| | Textile waste | x | | | | | (Nunes et al., 2018) |
| RDF production and utilization | MSW | x | | | | | (Aleluia and Ferrão, 2017) |
| | | | x | | | | (Dedinec et al., 2015; Singh and Basak, 2018) |
| Incineration | MSW | x | | | | | (Aleluia and Ferrão, 2017; Udomsri et al., 2010; Zis et al., 2013) |
| | | | x | | | | (Faizal et al., 2018; Islam and Jashimuddin, 2017; Menikpura et al., 2016; Singh and Basak, 2018; Tan et al., 2015; Woon and Lo, 2016) |

| Waste Component | Waste stream | Financial Assessment | Financial Assessment &Environmental Impact Assessment | Welfare Economic Assessment | Multi Criteria Decision Assessment | Other | Reviewed Publications |
|---|---|---|---|---|---|---|---|
| | | | | x | | | (Woon and Lo, 2016) |
| Landfill | MSW | x | | | | | (Zis et al., 2013) |
| | | | x | | | | (Faizal et al., 2018; Mehta et al., 2018; Menikpura et al., 2016; Singh and Basak, 2018; Tan et al., 2015; Zis et al., 2013) |
| | | | x | x | | | (Woon and Lo, 2016) |
| | Food waste | | x | | | | (Franchetti, 2013) |
| Waste management | Food waste | | x | | | | (Edwards et al., 2018; Kim et al., 2011; Takata et al., 2012; Willersinn et al., 2017; Wohner et al., 2020) |
| | | | x | x | | | (Martinez-Sanchez et al., 2016) |
| | Recyclables | | x | | | | (Ayodele et al., 2018; Burneo et al., 2020) |
| | MSW | x | | | | | (Colvero et al., 2020; Massarutto et al., 2011; Paes et al., 2020; Pinha |

| Waste Component | Waste stream | Financial Assessment | Financial Assessment &Environmental Impact Assessment | Welfare Economic Assessment | Multi Criteria Decision Assessment | Other | Reviewed Publications |
|---|---|---|---|---|---|---|---|
| | | | | | | | and Sagawa, 2020; Weng and Fujiwara, 2011) |
| | | | x | | | | (De Feo et al., 2019; Fei et al., 2018; Islam and Jashimuddin, 2017; Maghmoumi et al., 2020; Marino et al., 2017; Nie et al., 2018; Paes et al., 2020; Rigamonti et al., 2016; Zaman, 2016) |
| | | | | x | | | (Massarutto et al., 2011; Paes et al., 2020) |
| Waste regulation | Packaging Waste EPR Principle | | x | | | | (Andreasi Bassi et al., 2020; De Feo et al., 2019) |
| | | x | | x | | | (Cabral et al., 2013; Da Cruz et al., 2012; Marques et al., 2014; Rigamonti et al., 2015) |
| | EPR for WEEE | x | | | | | (Cheng et al., 2019) |

| Waste Component | Waste stream | Financial Assessment | Financial Assessment &Environmental Impact Assessment | Welfare Economic Assessment | Multi Criteria Decision Assessment | Other | Reviewed Publications |
|---|---|---|---|---|---|---|---|
| | Pay-As-You-Throw Schemes | | | x | | | (Manni and Runhaar, 2014) |
| | Plastic recycling targets | | x | | | | (Hestin et al., 2017) |

Source: Own Elaboration

# 5 Socio-economic impacts of key waste streams

This part of the report describes the main findings from peer-reviewed papers dealing with specific waste streams of Municipal Solid Waste, namely: the organic fraction of MSW, municipal food waste, packaging waste and plastic waste.

## 5.1 Organic fraction of Municipal Solid Waste (OFMSW)

From the reviewed publications, there was only one publication dealing exclusively with OFMSW in the EU (Mayer et al., 2020). However, there were other publications that assessed the impacts of organic waste source separation and subsequent management, although not exclusively focusing on this fraction, their scope was on mixed municipal solid waste or residual solid waste (Fernández-González et al., 2017; Martinez-Sanchez et al., 2015). The latter have not been included in this section of the report, but their methods have been considered in section 4.

Mayer et al. (2020) used LCA and levelized costs of exergy[10] (LCOE) to compare four waste treatment options for the OFMSW generated in Germany: (i) anaerobic digestion followed by composting, (ii) incineration of OFMSW, (iii) incineration of separately pre-dried OFMSW and (iv) anaerobic digestion with incineration of digestate.

The levelized costs of the exergy (LCOE) approach was used to determine the marginal production costs of exergy. The formula used to compute the LCOE relies on the following parameters/variables: share of debt capital, Investment costs, annual operation and management costs (including revenues), interest rate on debt, rate of inflation, annually exported exergy, share of equity capital, interest rate on equity and lifetime of the plant.

Based on the LCOE results, anaerobic digestion followed by composting showed the lowest marginal generation costs for exergy and thus the preferred option.

The LCOE for incineration are twice as high (with and without pre-drying) and the combination "anaerobic digestion and incineration" showed the highest economic burden partly due to the transportation between the two facilities. Anaerobic digestion followed by composting was also the management option showing better results in the LCA. Thus, according to the results of this publication, source segregation of OFMSW in Germany is justified and should be maintained.

## 5.2 Food waste

(De Menna et al., 2018) carried out a literature review of LCC applied to food, food waste and waste systems and found that the publications assessing the economic costs of food waste management are scarce.

### 5.2.1 Food waste generated at retailer level

Food waste generated at retailer level accounts for approximately 5% of the total food waste generated in the EU (Stenmark et al., 2016) and two publications of the reviewed publications (Albizzati et al., 2019; Giuseppe et al., 2014) dealt with it from a retailers' perspective but using different methodological approaches.

Albizzati et al. (2019) performed a cost analysis in parallel to an LCA on the valorisation and prevention of surplus food in the French retail sector. The costs analysis mainly used gate fees of waste treatments (Anaerobic digestion and incineration), tax credit for retailers carrying out redistribution and food prices for surplus prevented.

Their results demonstrated that retailers have economic gains when handling surplus food, conforming to the current management (mainly redistribution). However, the costs for the current management of food surplus varied between retailers (from -40 €/t to -410 €/t)

10 Exergy is a concept used in thermodynamics to represent the amount of work that can be done with an available energy.

depending on quantity and type of food donated. Savings per tonne of donated food increased not only by the quantity donated but also with the value of the donated food, i.e., the donation of more expensive products (namely animal-based products) had larger savings associated. Thus, maximising the redistribution of expensive products should be encouraged.

The cost analysis fully supported the waste hierarchy: management involving redistribution and/or animal feed offered lower financial costs than the traditional waste management (anaerobic digestion and incineration).

Giuseppe et al. (2014) presented a deterministic mathematical model for the optimization of food redistribution composed of retailers and potential recipients of food donations. The model considers financial benefits and management costs of food recovery for retailers to determine the optimal time to withdraw the products from the shelves as well as the quantities to be donated to non-profit organizations and those to be sent to the livestock market, as fodder, maximizing the retailer profit. The applicability of the proposed model relies on the use of automated warehouse management system in which key information (e.g., best-before date and optimal time to withdraw the product from the shelves) is available.

### 5.2.2 **Food losses and waste related to packaging**

Wohner et al. (2020) proposed a sustainability evaluation method for food-packaging systems considering the food losses and waste induced by the "emptiability" of each packaging option. The method integrates "emptiability" into the LCA and LCC results. Finally, the most sustainable food-packaging systems is identified by using multi-criteria decision analysis (MCDA). The method is demonstrated using a case study on tomato ketchup.

For the LCC part, they used the value added approach introduced by Heijungs et al. (2013). This means that the results of the LCC represents the sum of the value added provided by the product life cycle.

The argument in favour of using the value added sum instead of the sum of all the costs across the life cycle of a product or a service (as done in LCA with environmental impacts) relates to the fact that costs in each life cycle process already include upstream costs. Thus, adding all the costs will include double or multi-counting of the same cost items. It is also important to consider that depending on the perspective not all the life cycle stages should be included, for example, disposal costs when users have this service provided for free (Heijungs et al., 2013).

Wohner et al. (2020) found out that a greater material intensity in tomato ketchup products (including the amount of packaging used but also the food waste related to poor emptiability of some packaging) leads to higher value added along the supply chain. Contrary the product with the lowest packaging weight per kg of product and with the best emptiability ended up with the worst value-added results. Thus, from the manufacturer's point of view, a higher food loss would be preferable to increase profits. Using the consumer's perspective, the results would be exactly the opposite.

### 5.2.3 **Food waste prevention measures**

According to the review by Campitelli and Schebek (2020) of 366 peer-reviewed publications assessing WMS, prevention is the WMS component which is considered the least, although it has the highest priority in the waste hierarchy. Within the reviewed publications, there are two publications dealing with prevention of food losses (Willersinn et al., 2017) and food waste (Martinez-Sanchez et al., 2016).

Willersinn et al. (2017) combined environmental and socio-economic attributes in an overall sustainability index and combined this index with the consumers' preferences to select the most preferable measure to prevent/reduce the loss of Swiss potatoes. In the socio-economic analysis, Willersinn and colleagues carried out a financial assessment including all the potato supply chain (agricultural production, wholesaler stage, storage and packing, retailer, transportation, and private households). The socio-economic indicators derived were: net profit, total production cost, income variability, dramatic yield loss, invested capital and return on investment. For the sustainability index, indicators were agregated into different categories and weights were assigned.

Martinez-Sanchez et al. (2016) performed an Environmental LCC and a Social LCC of the food waste management in Denmark to compare 3 food waste management options (anaerobic digestion, incineration, animal fodder valorisation) and prevention of avoidable food waste. The novelty of this publication was the inclusion of the income effects when assessing prevention. If there is prevention of food waste (consumers only buy what they will eat) and the level of savings remain the same, the savings generated by the unpurchased food commodities would be used for other consumptions. This would mean that: 1) the total economic cost of all the scenarios would be the same, but the actors incurring the costs will be different and 2) the environmental impacts of prevention will depend on the nature of the marginal consumption done with the money saved due to prevention.

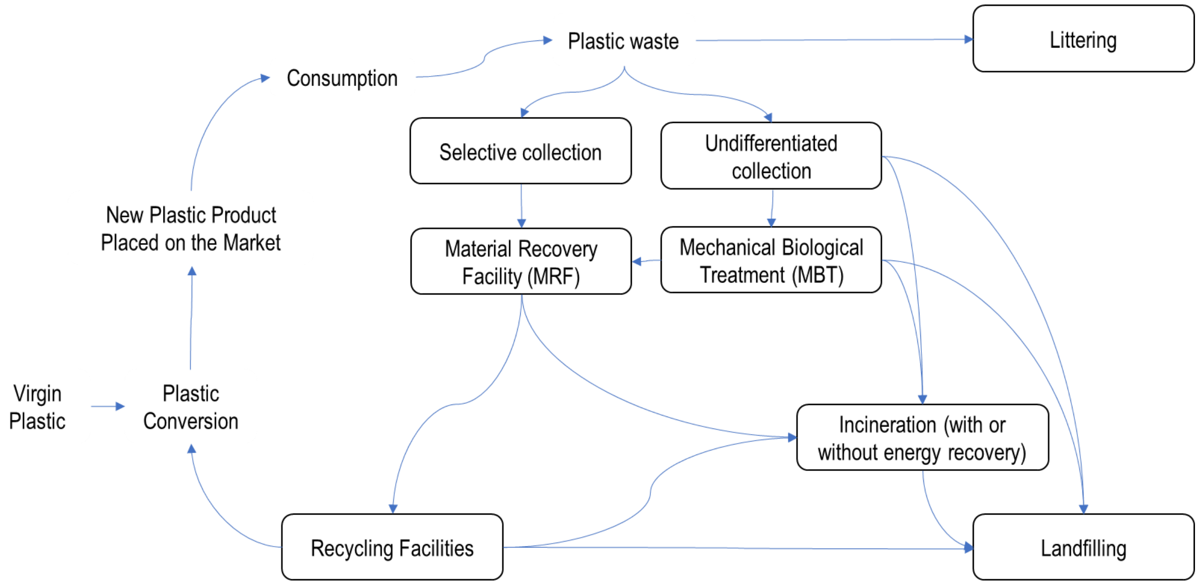
When considering only direct effects (and income effects are excluded), prevention of food waste appeared to be the preferred option in both LCCs (E-LCC and S-LCC) due to the resources saved by not producing the (prevented) food commodities.

In contrast, when including the indirect effects, prevention appeared to be environmentally worse than the alternatives (in LCA) when monetary savings from unpurchased food commodities were used for goods/ services whose production has larger environmental impacts than those of the (prevented) food. However, if the monetary savings were instead used for low-impact goods/services (such as health care, education, or insurances), the environmental impacts of prevention could be significantly reduced and ultimately be lower than those of the alternative management strategies. Hence, the environmental impacts of the income effects could be reduced if prevention measures were not only aimed at decreasing the purchase of unconsumed food items but also aimed at allocating monetary savings toward low-impact goods/services.

## 5.3 Municipal plastic waste

Figure 3 shows a simplified scheme including the most common plastic waste pathways in the EU. Some of the publications reviewed deal with multiple stages of the life-cycle, such as Andreasi Bassi et al. (2020), Da Cruz et al. (2012) and Hestin et al. (2017), while others focused on single stages, such as Cimpan et al. (2016) on MRF and Faraca et al. (2019) on recycling. Most of the publications focused on municipal post-consumer plastic waste, except Hestin et al. (2017) which included also industrial and commercial post-consumer plastic waste.

Figure 3: General diagram of the different pathways for plastic waste in Europe.




Source: Own Elaboration

### 5.3.1 **EPR principle for packaging waste**

Some of the reviewed publications dealt with the EPR principle for packaging waste. The EPR principle is one of the bases of the Packaging and Packaging Waste Directive (Directive 1994/62/EC). Thus, the directive stipulates that the industry is responsible for their packaging end-of-life, applying the EPR principle.

The research carried out by Da Cruz et al. (2012, 2014a), Ferreira et al. (2016), Marques et al. (2014) and Rigamonti et al. (2015) aimed at evaluating "whether the industry is actually covering the costs of collection and treatment/sorting of packaging waste" in different Member States with green dot systems.

These publications estimated the extra cost of packaging recycling for the public sector (mainly municipalities) and compared them to the financial transfers undertaken by the industry. They used a local authority perspective and focused on the life cycle of packaging waste from collection to the end of sorting (just before items are sold to recyclers). Thus, they exclude any further sorting, washing, and reprocessing carried out by recyclers.

The publications use two perspectives: financial and economic. Both perspectives included the same type of costs, except for the savings that derive from the diversion of waste from the residual waste collection and landfilling activities (on the benefits side), which are only included in the economic perspective. This element was called "opportunity costs" in Rigamonti et al. (2015) and defined as "the costs related to the packaging waste management in a scenario with no selective collection or sorting."

Da Cruz et al. (2012) admits that the valuation of environmental externalities (e.g., reduced $CO_2$ emissions) should be included for a full estimation of the economic benefits and costs of recycling. However, this was not done in any of these publications.

The overall idea of all these publications is based on the fact that if EPR is one principle of the PPWD, the sum of the Financial Support for Local Authorities (FSLA) and "other benefits attained from direct transactions with recyclers" should match all the costs included, i.e. operation costs of selective collection and sorting activities, depreciation of assets allocated to selective collection and sorting activities and the return on capital employed (debt and equity) for financing the assets allocated to selective collection and sorting.

The results for Portugal (Da Cruz et al., 2014a, 2012), France (Cabral et al., 2013), Belgium (Marques et al., 2014), Romania (Da Cruz et al., 2014a) and Italy (Rigamonti et al., 2015) point out that the industry is not paying the net financial cost of packaging waste management. If the EPR principle was to be strictly followed, the transfers to the local authorities would have to be increased by 35% in Portugal, 121% in France, 11% in Belgium and 119% in Italy (Da Cruz et al., 2014a). These increases should even be higher if the (public) subsidies for the investments made on the assets allocated to selective collection and sorting activities were not considered on the benefits side.

However, if the avoided residual waste collection and disposal costs with other treatments (landfill) are considered as a benefit for the local authorities, as avoided costs, the costs of the system are fully covered by 128% in Portugal, 135% in France, 204% in Belgium and 207% in Italy.

Andreasi Bassi et al. (2020) also studied the economic and environmental impacts of the Italian packaging waste and found out that Italian municipalities experienced losses of around 189-197 €/FU (FU = the management of 1,000 kg of household plastic packaging waste). The financial compensation from the Producer Responsibility Organization (PRO) covered only between 60-70% of the costs related to the collection and management of the source-separated plastic packaging waste in all the alternatives, while the municipality additionally had to cover the costs for managing the non-source-segregated plastic packaging material. This result is more or less aligned with Rigamonti et al. (2015) results (only 48% of the cost was being supported by the industry in Italy).

Andreasi Bassi et al. (2020) also demonstrated that: 1) an increase in collection rates resulted in increasing financial losses for the PRO and 2) recyclers are the weakest actor of the chain because they must deal with market fluctuations and have a relatively high fixed operational cost. Not all recyclers had the same situation, it depended on the type of plastic considered: while recyclers of PET had net profits, recyclers of FILM experienced net losses.

Andreasi Bassi et al. (2020) results also revealed that only between 15% and 36% of the generated plastic packaging waste in Italy can be transformed profitably into flakes and granules due to the lack of a stable demand for secondary plastic products. They also made three clear recommendations, based on their results:

- Environmental fees (EPR fees or Green Dot fees) should be increased to reflect a product's recyclability and the existence of a market for secondary material. This should be aligned with the introduction of economic incentives for "recycling-friendly" product designs.
- Financial transfers from industry to municipalities should be redefined to support increased separated collection of the highest possible material quality.
- "While deposit systems can bring both economic and environmental improvements compared to the baseline, their implementation should be carefully integrated with existing EPR schemes and with plastic waste management systems, to guarantee financial robustness and stability throughout the value chain."

### 5.3.2 **Plastic waste recycling targets**

Hestin et al., (2017), carried out an assessment of the environmental (GHG), economic (operating costs (including investments) and revenues), and social (direct jobs) impacts of increased plastic recycling in the EU-28, by 2020 and 2025.

The study considers targets included in the EC Directive proposal COM (2014) 397, as well as targets found in Directives 2008/98/EC on waste, 2000/53/EC on end-of-life vehicles, and 2012/19/EU on waste electrical and electronic equipment. Non-legislative targets were also considered for C&DW and agricultural plastic waste. The study included the waste management chain from the generation of plastic waste by the end user, to the production of final recycled plastic materials (e.g., flakes, pellets) and excluded plastic converters.

The results are presented in three parts: an economic part, an environmental part, and a social part. The results of the three parts are estimated as additional impacts due to target accomplishment by comparing the impacts of the business-as-usual scenarios for 2020 and 2025 (not meeting the plastic recycling targets) against the scenarios meeting the targets of 2020 and 2025.

In the economic part, the study estimated that meeting the plastics targets in the EU legislation by 2020 could result in a net cost of between 700 million EUR and nearly 1.6 billion EUR, compared to the BAU scenario. These net costs results are from the sum of the operating costs minus the revenues from the sales of recycled plastics. The large variation of results is due to the uncertain range of recycled plastics prices. Taking an 'average prices' approach, the net cost was approximately 1.1 billion EUR per year in 2020. The variation in the results for 2025 are broader than for 2020, showing a net cost of 720 million EUR (highest recycled material prices), a net cost of 2.3 billion EUR (lowest recycled material prices), and a net cost of about 1.45 billion EUR per year in 2025 considering average sale prices for all recycled plastic resins.

For both years, recycling appears to be the costliest step along the chain and accounts for more than half of the total operating costs. But on the other hand, the sales of recycled plastics at the end of the value chain represent significant revenues. Collection of plastics and sorting operations contribute with similar costs as recycling, while the transportation cost is very low. Landfilling costs decrease as more plastic waste is diverted from landfills to recycling (and energy recovery).

According to Hestin et al. (2017), given that the operating costs across the whole value chain are higher than the revenues from the sales of secondary raw materials (in the average situation presented), a share of these costs needs to be supported by other revenues than the sales from the materials, including Producer Responsibility Organisations' schemes and local charges which are established to supplement collection and sorting costs. Some drivers could progressively reduce the need for this external financing:

1) Increasing prices of secondary raw materials by regulatory measures (such as GHG taxes or mandatory recycled plastics integration in products) and/or future increase of the oil prices.
2) Increasing landfill taxes would virtually increase the revenues of the recycling value chain, and therefore improve the balance between the operating costs and the revenues from the sales of recycled plastics.

In addition, Hestin and colleagues estimated that meeting the targets for 2020 would lead to a reduction of GHG emissions of nearly 8 Mt $CO_2$ equivalent and a creation of 50 000 additional direct jobs (FTE). Meeting the targets for 2025 would mean a reduction of 13 Mt $CO_2$ equivalents and the creation of 80 000 additional direct jobs (FTE).

### 5.3.3 **Packaging waste collection and sorting**

Pires et al. (2017) compared three collection systems for packaging waste, namely 1) bring scheme, 2) curbside collection and 3) a combination of bring and curbside collection with respect to environmental and economic aspects in Portugal from a perspective of a private company operating the collection and the sorting of packaging waste.

The cost assessment included capital costs, operational costs, and maintenance costs of collection. It also included: costs of processing waste in their Material Recycling Facility (MRF), the payment of a landfill tax for MRF refuse and the costs of landfilling refuse. On the benefits side for the private company, it included revenues from the sale of recyclables to the PRO as well as other revenues related to tariffs charged to municipalities, which are charged for the fraction landfilled, as well as the refuse of the sorting plant.

According to Pires and colleagues' results, bring collection is more economical than curbside collection. The collection of the yellow stream (including metal, plastic, and beverage cartons) appeared to be more costly per tonne for the private company and per route than the collection of the blue stream (paper/cardboard packaging and non-packaging) due to waste density differences. However, when the sale of recycled material is introduced, together with MRF processing costs and landfill costs, the cost per tonne is higher for paper/cardboard than for the lightweight packaging stream. This difference can be explained by the different values of recyclables, higher for recyclables such as plastics (732 €/t), mixed plastics (245 €/t), and beverage cartons (693 €/t) than for paper/cardboard (122 €/t).

Cimpan et al. (2016) made a techno-economic analysis of Material Recovery Facilities processing commingled lightweight packaging waste (LWP) for Germany to estimate process and cost efficiency. According to their estimations, LWP MRFs in Germany operate at an overall net cost, which must be covered by the gate fees or sorting fees. The revenues from sales of recovered materials are significantly reduced or completely overturned (even when using optimistic market prices) by the disposal costs of sorting residues. Results suggest that economies of scale exist in LWP MRFs.

Due to the economies of scale, larger plants make use of a more comprehensive preparation of material streams before the sorting processes, thus maximizing the performance of the sorting equipment.

Medina-Mijangos et al. (2020) propose a methodology to carry out a techno-economic analysis for MSW management projects based on social cost-benefit analysis (sCBA) and applying the methodology to a private sorting facility of lightweight packaging waste and bulky waste in the Metropolitan Area of Barcelona (SEMESA).

The method considers internal and external impacts and aims at determining the total benefits (the difference between revenues and costs) generated by a project and to reduce uncertainty and risk of investing in a specific MSW management system. The total benefit of the project was calculated as the sum of the private benefit and externality benefit minus the opportunity costs. Even if the methodology comprises a wide range of cost items, the range was narrowed down in the case study application.

The study concludes that the sorting facility of SEMESA is operationally viable, since its private benefit (only including internal costs) is positive (42.94 €/t) as well as economically viable, since its total benefit including private and external benefits is also positive (87.73 €/t). The most representative revenue appeared to be the payment for the provision of service for the selection and treatment of light packaging waste and bulky waste (91.61 €/t) and the most representative costs is its capital costs (58.39 €/t). In this case study, the interest earned using a financial instrument is considered as opportunity cost, because there are no better alternatives for the use of waste or land where the facility is located. But if better alternatives are considered and the opportunity cost is greater than 89.5 €/tonne, the project could become economically unviable.

### 5.3.4 **Recycling of hard plastic waste**

Faraca et al. (2019) performed an environmental and financial assessment of three management scenarios for one tonne of hard plastic waste collected at Danish recycling centres. Recycling centres are manned collection points where the waste is sorted into around 40 material fractions (44–50% of source-separated postconsumer plastic waste in Denmark was collected in recycling centres in 2015).

Faraca et al. (2019) considered three recycling scenarios: two mechanical recycling scenarios (a simpler and a more advanced configuration) and a feedstock recycling scenario (chemical recycling) based on pyrolysis. The advanced mechanical recycling scenario provided the largest savings in the highest number of environmental impact categories (including global warming potential) and total costs.

According to Faraca et al. (2019), the financial viability of recycling depends on the market acceptance of the recycled pellets, as all scenarios achieved net financial revenues in case of market substitution factors above 0.6-0.7. The market substitution factors are often used in LCA studies to consider the fact that recycled material may not be used in the same applications as the virgin material. These factors represent the loss of material grade or quality of the secondary material produced with respect to its virgin substitute. For example, a market substitution factor of 0.6 means that one tonne of recycled PET would substitute 0.6 tonnes of virgin PET. Faraca and colleagues demonstrated that if high quality recycled plastic is achieved, both environmental savings and financial profits are possible.

# 6 Conclusions

Economic and socio-economic assessments of waste management systems can provide useful information for decision makers when designing and planning new waste policies and strategies. However, there is not one single methodology widely adopted by scholars and practitioners. This report provides a literature review of studies dealing with socio-economic impacts of waste management options to gather key aspects and particularities used in state-of-the-art publications.

From the literature review, three main findings emerge:

Firstly, market failures are common in waste management given the particularities of waste markets, as for example, the fact that waste might have very low or negative prices. Externalities are the most widespread market failure, although insufficient competition and information asymmetries also play a role. In general, in the presence of market failures the amount of waste being landfilled and incinerated would be larger than in those cases where these market failures are corrected, typically leading to an increase in recycling. From a welfare economics point of view, intervention to correct market failures in waste management is required to achieve a more efficient resource allocation.

In the EU, a set of interventions devoted to correcting such failures has been implemented. Externalities are being addressed from several standpoints including command and control measures (e.g. recycling targets) and MBIs (e.g. landfill taxes, PAYT, EPR schemes). In this sense, recycling tends to be associated with less generation of externalities, and thereby, from a welfare economics point of view, it implies a more efficient allocation of resources.

All in all, it is widely acknowledged that in waste policy there is not one measure fitting all possible issues to be solved, given the heterogeneity of waste streams and related markets. In general, a policy mix containing the most appropriate measures for each case tends to be applied (e.g. mixes of recycling targets and landfill taxes).

Secondly, the approaches and methods used to assess the economic and socio-economic impacts of recycling policies and pathways found in literature are multiple and diverse. Economic and socio-economic impacts of waste management can be assessed only in monetary terms or with methodologies integrating different types of units, such as MCDA. Methods using exclusively monetary terms can be classified as financial assessment or welfare economic assessment. Multiple names are given to these two main types of assessment, and sometimes both are named "economic assessments". Often, when the publication combines economic assessment with Life Cycle Assessment, the cost assessment is named Life Cycle Costing. In practice, the main difference between these two main types of methods (when assessing the same project) is the type of costs included. While a financial assessment includes budget costs and transfers, a welfare economic assessment includes budget costs and externality costs.

From the reviewed publications, we draw the following conclusions:

- Most publications used a financial assessment approach in combination with some environmental indicators (LCA or just GHG emissions). A Financial Assessment alone was also quite common. Only one fifth of the publications used Welfare Economic Assessment and fewer used MCDA or other types of methodologies.
- Most of the publications included budget costs (CAPEX and OPEX). In addition, some publications included transfers and/or externality costs. Fewer included non-monetized impacts.
- Most of the publications adopted a single actor perspective. Only a few publications used a multi-actors/stakeholders' approach. Thus, in most cases it was not possible to identify economic winners and losers and relate them to economic incentives or compensations.

- Most of the cost results are shown in relation to waste tonnage, but in publications dealing with Waste-to-Energy (WtE) options, costs are often presented per unit of energy generated.
- Most of the reviewed publications worked with single values for cost data. Few publications work with probability distributions to take cost data variability into account.
- The focus of most of the reviewed publications was on single waste components (e.g. treatment) or a combination of waste management components to assess the waste management scheme. Few publications focused on waste prevention or on assessment of waste policies.
- The questions posed in most of the reviewed publication were: "what are the current costs of the WMS?" and "what are the costs for the alternatives being considered?".

Thirdly, the number of publications assessing the economic and socio-economic impacts of municipal food/bio-waste and municipal dry recyclables (priority on plastics) in an EU-context are not enough to be able to generalize their findings. The goal and the scope of the publications found (and described in Section 5) were different from each other and consequently their findings were different and could not be compared against each other. The exception was a group of publications using the same methodology to assess the costs of the EPR schemes used for packaging waste in different EU countries. The main finding from this group of publications was that the packaging industry is not paying for the net financial cost of packaging waste management in different EU countries. If the EPR principle was to be strictly followed, the transfers to the local authorities would have to be increased.

The lack of an agreed methodology to assess the economic and socio-economic performance of waste management systems has generated a blurring of definitions, approaches, and methods in literature. Even if the findings of each single publication provide useful information for the decision makers of the specific case study, in most of the cases, their results could not be compared to other case studies because they assess different systems and/or used different methods.

Given the existing heterogeneity among the waste streams, related markets, waste components and policy interventions, there cannot be one single methodology to answer all the research questions about costs of waste management systems. However, it is important, first, to identify key costs to take into account when addressing each specific waste streams (e.g. plastic waste from packaging) or waste components (e.g. plastic recycling). Second, it is important to increase the number of studies for key waste streams or waste components. In addition, to be able to consider the diversity within the EU in terms of waste systems and policies, the same question should be answered using the same economic and socio-economic assessment method to capture the particularities of each context.

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## List of abbreviations and definitions

| | |
|---|---|
| AD | Anaerobic Digestion |
| CBA | Cost Benefit Analysis |
| CAPEX | Capital Expenditure |
| C-LCC | Conventional LCC |
| E-LCC | Environmental LCC |
| EPR | Extended Producer Responsibility |
| FA | Financial Assessment |
| FTE | Full Time Equivalent |
| LCC | Life Cycle Costing |
| LCOE | Levelized Costs of Energy |
| LF | Landfill |
| LFGTE | Landfill gas to Energy |
| MBI | Market based instruments |
| MBT | Mechanical Biological Treatment |
| MCDA | Multi Criteria Decision Assessment |
| MRF | Material Recovery Facility |
| MSW | Municipal Solid Waste |
| NPV | Net Present Value |
| OFMSW | Organic Fraction of Municipal Solid Waste |
| OPEX | Operating Expenses |
| PRO | Producer responsibility Organization |
| PAYT | Pay-As-You-Throw |
| Q1 | Question 1 of the project |
| Q2 | Question 2 of the project |
| Q3 | Question 3 of the project |
| S-LCC | Societal LCC |
| WEA | Welfare Economic Assessment |
| WEEE | Waste Electronic and Electric Equipment (also called e-waste) |
| WMS | Waste Management System |
| WtE | Waste-to-Energy |
| WTP | Willingness-To-Pay |

## Annex 1. Scopus search results based on the search term and article parts.

| | Seach string | TITLE | KEY | abstract | TITLE-ABS-KEY |
|---|---|---|---|---|---|
| Search 1 | TITLE (waste AND [economic AND assessment OR analysis OR evaluation OR estimation]) | 728 | 8472 | 12372 | 24460 |
| Search 2 | TITLE (waste AND "Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") | 38 | 319 | 535 | 837 |
| Search 3 | TITLE ("waste" AND "Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") | 48 | 5158 | 585 | 6475 |
| Search 4 | TITLE ("waste" AND [ "socio-economic" OR "socioeconomic" ] AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) | 23 | 622 | 1093 | 2161 |
| Search 5 | TITLE ("waste" AND "welfare" AND "economic" AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) | 1 | 27 | 112 | 216 |
| Search 6 | TITLE ("waste" AND financial AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) | 27 | 751 | 1776 | 3331 |
| Search 7 | TITLE (waste AND cost AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) | 304 | 13028 | 16853 | 37062 |
| Search 8 | (TITLE (waste AND [economic AND assessment OR analysis OR evaluation OR estimation]) OR TITLE (waste AND "Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") OR TITLE ("waste" AND "Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") OR TITLE ("waste" AND "welfare" AND "economic" AND [ "assessment" OR "analysis" OR "evaluation" OR " estimation" ]) OR TITLE ("waste" AND financial AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE ("waste" AND "cost" AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ])) | 1064 | 18661 | 25806 | 53121 |
| Search 9 | TITLE (waste) AND TITLE (economic AND assessment OR analysis OR evaluation OR estimation) OR TITLE ("Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") OR TITLE ("Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") OR TITLE ([ "socio-economic" OR "socioeconomic" ] AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE ("welfare" AND "economic" AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE (financial AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) | 1072 | 18907 | 26096 | 53668 |
| Search 10 | TITLE (waste) AND TITLE (economic AND assessment OR analysis OR evaluation OR estimation) OR TITLE ("Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") OR TITLE ("Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") OR TITLE ([ "socio-economic" OR "socioeconomic" ] AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE ( "welfare" AND "economic" AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE (financial AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) AND NOT TITLE ("waste heat") | 703 | | | |
| Search 11 | TITLE (waste) AND TITLE (economic AND assessment OR analysis OR evaluation OR estimation) OR TITLE ("Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") OR TITLE ("Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") OR TITLE ([ "socio-economic" OR "socioeconomic" ] AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE ( "welfare" AND "economic" AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE (financial AND [ "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) AND NOT TITLE ("waste heat") AND (EXCLUDE (DOCTYPE, "cp") OR EXCLUDE (DOCTYPE, "no") OR EXCLUDE (DOCTYPE, "er")) AND (EXCLUDE (LANGUAGE, "German") OR EXCLUDE ( LANGUAGE, "Portuguese") OR EXCLUDE (LANGUAGE, "Japanese") OR EXCLUDE (LANGUAGE, "Russian") OR EXCLUDE ( LANGUAGE, "Polish") OR EXCLUDE (LANGUAGE, "Spanish") OR EXCLUDE (LANGUAGE, "Chinese") OR EXCLUDE (LANGUAGE, "French") OR EXCLUDE (LANGUAGE, "Ukrainian")) | 557 | | | |
| Search 12 | TITLE (waste) AND TITLE (economic AND impact OR benefit OR potential OR assessment OR analysis OR evaluation OR estimation ) OR TITLE ("Life Cycle Costing" OR lcc OR "Life-Cycle Costing" OR "Life Cycle Costs") OR TITLE ("Cost Benefit Analysis" OR "Cost Benefit Assessment" OR "CBA") OR TITLE ([ "socio-economic" OR "socioeconomic" ] AND [ AND impact OR benefit OR potential OR "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE ("welfare" AND "economic" AND [impact OR benefit OR potential OR "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) OR TITLE (financial AND [ AND impact OR benefit OR potential OR "assessment" OR "analysis" OR "evaluation" OR "estimation" ]) AND NOT TITLE ("waste heat") AND (EXCLUDE (DOCTYPE, "cp") OR EXCLUDE (DOCTYPE, "no") OR EXCLUDE (DOCTYPE, "er")) AND (LIMIT-TO (LANGUAGE, "English")) AND (EXCLUDE (LANGUAGE, "Russian") OR EXCLUDE (LANGUAGE, "Ukrainian")) | 650 | | | |

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