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Improving Sustainability and Circularity of European Food Waste Management with a Life Cycle Approach

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Abstract

In the past years, several research initiatives have been promoted in the area of food waste. Many of these were focused on the identification of key drivers of food wastage and on the quantification of food waste generation. While these initiatives provided fairly accurate information over European food waste generation patterns and management routes, they did not always deliver comprehensive and comparable knowledge on the sustainability of food waste management and on ways to mitigate negative consequences at environmental, economic and social levels.

Building on most recent methodological advancements and policy needs, the work presented in this report provides decision makers and waste managers with a life-cycle based framework methodology to quantify the environmental and economic sustainability performance of European food waste management. This methodology makes use of multi-objective optimization and Pareto optimality concepts in order to help identify most sustainable management options for food waste, intended as those that minimize environmental and economic impacts.

Therefore, applying this methodology can offer relevant insights to the decision making process. The social dimension of sustainability is also addressed, though the assessment of the social performance is kept separated from the proposed framework methodology.

A numerical case study is also developed. This is meant to give an example of simplified application of the proposed methodology to a fictitious European food waste management context. The environmental dimension has been evaluated with the Life Cycle Assessment (LCA) software EASETECH, while the economic assessment is conducted based on a number of different indicators expressing the costs associated with food waste management.

1. Introduction

1.1 Overview

Worldwide, over 1.3 billion tonnes of food for human consumption is wasted or lost annually (FAO, 2011) throughout the food supply chain (FSC)[1], which represents about 1/3 of the global food production. Such huge fraction, paradoxically, does not thus reach the part of the world population that is estimated to be undernourished: about 11%, corresponding to nearly 800 million people (FAO, 2015)[2]. In Europe, 2014 estimates show up to 100 million tonnes of food waste generation per year[3], corresponding to approximately 200 kg per capita[4]. To guarantee access to a proper amount of food to the increasing world population without provoking unsustainable environmental pressures is undeniably one of the major challenges for the current and future generations.

Such a massive generation of food waste, in fact, leads to significant environmental impacts, as well as to economic and social costs. For instance, worldwide figures provided by FAO (FAO, 2013) on the consequences of food produced for human consumption that had been lost or wasted include (in 2007): 3.3 Gtonnes of CO_2 eq. emitted to the atmosphere, 250 km^3 of surface and groundwater consumption (i.e. $2.5*10^{11} m^3$) and 1.4 billion hectares (per year) of land occupation. When putting these figures into perspective, it can be found that food waste (for human consumption) is worldwide associated with: GHG emissions that equals about half of the total GHG emissions of the USA (USA emits around 7 Gtonnes of CO_2 eq.), consumption of water that corresponds to almost three times the volume of Lake Geneva and wasted food production that occupies an area of about 28% of the world's agricultural land (FAO, 2013). In Europe, food waste is responsible for about 170 Mt of CO_2 eq. per year (EC, 2010a). Worldwide, the economic costs of food waste were estimated to be 1055 billion USD (FAO, 2013).

The social consequences associated with food waste is tightly interlinked with environmental and economic concerns. Impacts on the environment, for instance, reduce the availability of natural resources, lead to increased food prices and affect people's livelihood, health and wellbeing (EU FUSIONS, 2015). FAO estimated a monetary value for these social costs up to 882 billion USD (FAO, 2013).

BOX 1 – Important Definitions

In the context of the 2015 EU Circular Economy package (EC, 2015a), the Circular Economy Action Plan (EC, 2015b) states that "*Food waste takes place all along the value chain: during production and distribution, in shops, restaurants, catering facilities, and a home*".

Food waste is a main constituent of the broader group **bio-waste**, which was defined in the Waste Framework Directive 2008/98/EC (EC, 2008) as *"[…] biodegradable garden*

[1] The food supply chain (FSC) typically includes agriculture production and/or animal farming, transport, processing, distribution and consumption.
[2] The same source reports detailed figures of the trend of undernourishment around the world. These are progressively decreasing, e.g. ranging from about 19% of the world population in 1990-1992, to the forecasted 11% in 2014-2016.
[3] EC Food Waste Website - http://ec.europa.eu/food/safety/food_waste/index_en.htm
[4]Considering a population of 503 million people in the EU27 in year 2014, as from Eurostat data: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=demo_pjan&lang=en

and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants"**.**

The Landfill Directive 1999/31/EC (EC, 1999), clarifies that an even broader group exist called **biodegradable waste**. The latter includes "*any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and cardboard".*

Consistently with the definition proposed by WRAP (2009) and by the Commission (EC, 2010a), **edible food waste** includes **avoidable food waste** which is "*food that is thrown away that was, at some point prior to disposal, edible (e.g. slices of bread, apples, meat)"* and **possibly avoidable food waste** which is "*food that some people eat and others do not (e.g. bread crusts, potato skins)"*.

Inedible food waste includes waste arising from food preparation that is not, and has not been, edible under normal circumstances (e.g. bones, egg shells, pineapple skins).

While this report also provides an overview of key aspects related to food waste along the entire food supply chain (FSC) (e.g. identification of drivers and sources of food waste generation), the key focus is the final part of such supply chain, i.e. that representing the management of the generated food waste. This report is in fact an attempt to provide life cycle based methodological support to help taking decisions involving food waste management, with a view of improving its overall sustainability.[5]

1.2 Legislative context

In addition to try to reduce/prevent food waste generation, Europe is also committed to designing and implementing measures to improve the management of food waste, so to reduce unwanted consequences at environmental, economic and social levels. In 2011, the European Commission (EC) identified food waste as one of the main problems that needed to be addressed to increase resource efficiency (EC, 2011a and 2011b) and invited all Member States (MS) to address food waste in their National Waste Prevention Programmes as foreseen in the Waste Framework Directive 2008/98/EC (EC, 2008). The Commission is also working together with MS and stakeholders towards designing strategies and solutions to ensure food safety[6] (EC, 2013).

In 2014, the EC announced the intention of reducing generation of food waste of at least 30% by the end of 2025 compared to 2017 levels. This proposal was part of the first Circular Economy package (EC, 2014a & 2014b, Figure 1), which was withdrawn in February 2015 with the intention of replacing it with a more ambitious and coherent one.

On December 2nd 2015, the new Circular Economy package (EC, 2015a) was launched. In its action plan (EC, 2015b) it establishes a 50% reduction target of food waste generation, in line with the target set by the United Nation General Assembly as part of the 2030 Sustainable Development Goals. It indicates that "*The Commission will elaborate a common EU methodology to measure food waste in close cooperation with Member States and stakeholders*". It recognises that "*Food waste is an increasing concern in Europe. The production, distribution and storage of food use natural resources and generate environmental impacts […] and causes financial losses for consumers and the economy. Food waste has also an important social angle […]".* Furthermore, the new package stresses that "*Action by Member States, regions, cities,*

[5] More details on the objectives of this report are provided in section 1.4

[6] EC Food Waste website, http://ec.europa.eu/food/safety/food_waste/index_en.htm last accessed December 2015

and business along the value chain is essential to prevent food waste […]. The Commission supports […] the dissemination of good practices in food waste prevention." The new Circular Economy package, thus was clearly recognised that reducing food wastage and improving food waste management are key steps towards increasing the circularity of the European economy. In addition, the new package includes proposals for amendment and revision of several pieces of legislation, including the Waste Framework Directive (EC, 2015c).

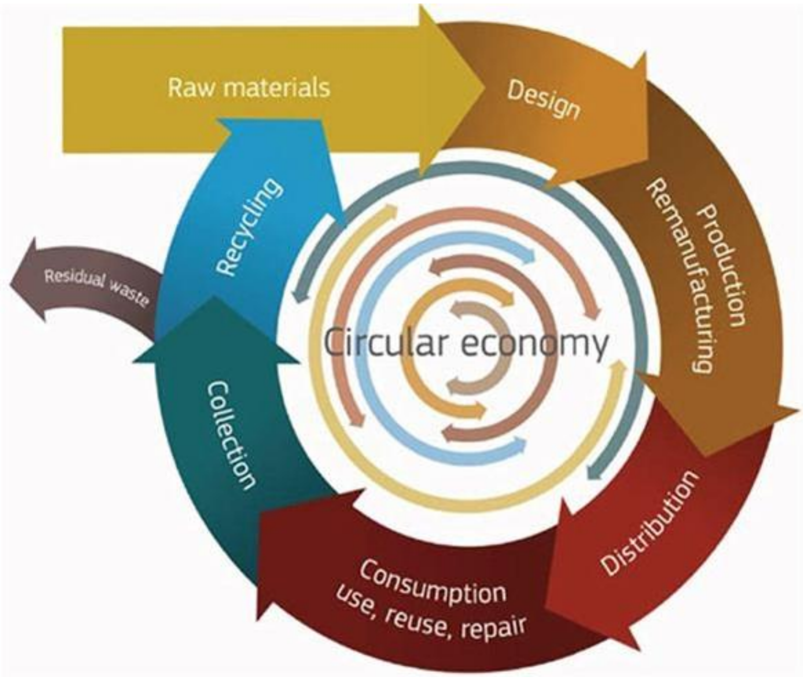


Figure 1. Schematic representation of the Circular Economy concept.

Several guidance documents were prepared to help governments and other stakeholders preparing their food waste prevention programmes (EC, 2011c; UNEP, 2014). As a response, several MS introduced non biding reduction targets ranging from 20% to 50% to be met up to 2025 (e.g. UK, 2014) and measures were implemented at national and local levels in Europe, such as consumer sensibilisation campaigns, food donation, food redistribution and food re-use in feeding.

At the moment, the Landfill Directive 1999/31/EC (EC, 1999) and Waste Framework Directive 2008/98/EC (EC, 2008) tackle the food waste management issue only indirectly and food waste appears within the broader group of bio-waste and biodegradable waste (see Box 1). In 1999, the Landfill Directive had set mandatory targets to progressively reduce the share of biodegradable municipal waste put into landfills, e.g. a reduction of 35% of the total amount produced in 1995 should be achieved by 2016. As of today, some MS (e.g. Denmark, Germany, The Netherlands) have already completely banned landfilling of biodegradable waste. The Waste Framework Directive 2008/98/EC does not include specific provisions on food waste, nor even a definition of what food waste is/includes. However, according to this directive measures shall be taken by MS to achieve environmental sound management of waste by following the so-called "waste hierarchy" (art 4(1)). Such hierarchy considers waste prevention as the most environmentally sound option (but no prevention targets were established), while landfilling is considered as the worst option[7].

Biological treatment of food waste through anaerobic digestion and composting are well established technologies and currently available at small and large scale. They allow for

[7] See chapter 2.1 for more details on the waste hierarchy.

waste to be treated in a way that the process output(s) can (partly) be re-used in the production of bio-products and energy (fertilizer, biogas and biomethane). For instance, an average anaerobic digestion plant can generate more than 2000 MJ (as biogas) per tonne of food waste, excluding the biogas used for own energy consumption in the plant (EC, 2010b), while the use of composting in agriculture can decrease the necessity of using synthetic fertilisers[8].

By promoting the re-use of waste in the production of energy and products, the Renewable Energy Directive 2009/28/EC (EC, 2009) and the communication "Innovating for Sustainable Growth: A Bioeconomy for Europe" COM(2012)60 (EC, 2012a) includes also few initiatives that can contribute to address the food waste problem. For instance, the Renewable Energy Directive stipulates a 20% mandatory target for renewables in energy consumption in EU by 2020 and a 10% target in the transport sector; at the same time, it specifies that biofuels and bioliquids produced by wastes and residues will count double for each member state renewable target of 10%. In fact, this target boosted the re-use of used cooking oils for biofuels production and the construction of new installations of anaerobic digestion for biogas production from bio-waste.

Improving the sustainability of food waste management systems and strategies is a complex task because it requires a coherent and integrated consideration of several environmental, economic and social aspects. The EC recommends the use of Life Cycle Thinking (LCT) to support sound waste management (e.g. Waste Framework Directive 2008/98/EC and COM(2010)235), so that all life cycle stages are taken into account and shifting of burdens (among e.g. life cycle stages, environmental impacts, geographical areas) are minimised.

1.3 Environmental Footprint and Food Round Table

Initiatives such as the EC Product Environmental Footprint (PEF) pilot and the Sustainable Consumption and Production Food Round Table – although not specifically targeting food waste – inevitably integrate the issue of food waste and waste management in their studies. A description of these initiatives and their relation with food and food waste is given in the following sections.

1.3.1 European Commission Product Environmental Footprint

In the context of the Communication "Building the Single Market for Green Products" COM(2013)196 (EC, 2013b), the European Commission (EC) recommends a method to measure the environmental performance of products and organisations, named the Product Environmental Footprint (EC, 2013c) and Organisations Environmental Footprint (EC, 2013d).

The PEF is a multi-criteria measure of the environmental performance of goods and services from a life cycle perspective. PEF studies are produced for the overarching purpose of identifying and seeking to reduce the environmental impacts associated with goods and services, taking into account supply chain activities (from extraction of raw materials, through production and use, to final waste management). As the PEF guidelines are overall guidelines that have to be applicable to all products, additional product specific guidelines are needed. To address this issue, the EC launched in 2013 a three-year pilot project to develop Product Environmental Footprint Category Rules (PEFCRs) that provide category-specific guidance for calculating and reporting life cycle environmental impacts of products in a harmonised way.

[8] More info on food waste treatment and management options are provided in chapter 3.3

The main reason for this activity is that existing life cycle-based standards do not provide sufficient specificity to ensure that consistent assumptions and measurements are made to potentially enable comparable environmental claims. In order to address that limitation, the use of PEFCRs will play an important role in increasing the reproducibility, relevance, and consistency of PEF studies (and therefore comparability among PEF calculations within the same product category).

The EC launched in January 2014 a call for volunteers from the food, feed and drink sectors to test the development process of PEF/OEF guides. Eleven pilots have been retained by the EC out of thirty applications. Most of them came from EU based organisations, but applications from Australia, New-Zealand, Sri Lanka and Tunisia demonstrate the interest of non EU countries to participate in this pilot phase. The food, feed and drink pilots selected are:

1. **Beer,** proposed by Brewers of Europe;
2. **Coffee,** proposed by the European Coffee Federation;
3. **Dairy,** proposed by the European Dairy Association;
4. **Feed for food-producing animals,** proposed by the European Feed Manufacturers' Federation;
5. **Fish for human consumption,** proposed by the Norwegian Seafood Federation;
6. **Packed fresh meat from bovine, pigs and sheep,** proposed by the European Livestock and Meat Trades Union;
7. **Uncooked pasta,** proposed by Union of Organizations of Manufactures of Pasta Products of the EU;
8. **Packed water,** proposed by the European Federation of Bottled Waters;
9. **Pet food (cats & dogs),** proposed by European Pet Food Industry Federation;
10. **Olive oil,** proposed by CO_2 consulting S.L.;
11. **Wine,** proposed by the Comité Européen des Entreprises Vins.

The first task of the pilots studies is to carry out a screening study to identify the elements that most contribute to the product overall impact, the analysed elements include: life cycle stages, processes, environmental impact categories and elementary flows of a representative product that describes the average product sold in the European markets. The results of the screening study are used as a basis for the drafting the PEFCR. Once the draft PEFCR has gone through a public stakeholder consultation9 and has been approved by the Environmental Footprint steering committee, it will be tested in supporting studies, which will apply the PEFCR for real products. During the supporting studies, also various ways of communicating the environmental footprint results to consumers and businesses will be tested. The PEFCR will be revised based on the lessons learned from the supporting studies, after which the stakeholders have another opportunity to provide comments on the PEFCR. Before final approval of the PEFCR by the EF steering committee, the PEFCR will be reviewed by external reviewers. The final PEFCRs are scheduled to be released by end of 2016.

1.3.2 The Sustainable Consumption and Production (SCP) Food Round Table

Since 2009, Food Round Table members have been working together on a commonly-agreed and science-based framework for assessment and communication of the environmental performance of food and drink products in Europe. An analysis of relevant data, methodologies and guidelines for assessing the environmental performance of food and drink has been conducted. The analysis led to a harmonised methodology for

[9] Stakeholders can register to follow pilots on Environmental Footprint wiki-page: https://webgate.ec.europa.eu/fpfis/wikis/display/EUENVFP/EU+Environmental+Footprint+Pilot+Phase

environmental assessment, the ENVIFOOD Protocol. The Protocol provides guidance to support environmental assessments of food and drink products conducted in the context of business-to-business and business-to-consumer communication and the identification of improvement options.

The Round Table (RT) is co-chaired by the EC and food supply chain partners on equal footing and supported by the UN Environment Programme (UNEP) and European Environment Agency (EEA). When applying a life cycle approach, the RT's unique structure based on transparency and dialogue facilitates an open, results-driven and evidence-based dialogue among all players along the food chain which leads to further harmonization. The RT has delivered the publication of the ten "Guiding Principles on the voluntary provision of environmental information along the food chain" (European Food SCP Roundtable, 2010), the Reports on "Communicating environmental performance along the food chain" (European Food SCP Roundtable, 2011) and "Continuous Environmental Improvement" (European Food SCP Roundtable, 2012) and the ENVIFOOD Protocol (European Food SCP Roundtable, 2013).

In the EU Commission PEF Food Pilot phase, the ENVIFOOD Protocol is used as a complementary guidance to the PEF/OEF guides (EC, 2013c and 2013d). The RT supports the PEF/OEF testing by:

- Facilitation of coordination and consistency between pilots, including through participation in PEF pilot consultations and organisation of technical workshops;
- Providing technical support for the interpretation of the ENVIFOOD Protocol, in relation with the EF Technical Helpdesk;
- Participation of the WG1 industry co-chair in the PEF Technical Advisory Board;
- Help PEF pilot testers to come up with a common approach on cross cutting issues.

1.4 Objectives

While assessment of the performance of waste management systems and strategies keeps being extensively covered by scientific literature (e.g. Arafat et al., 2015; Pressley et al., 2014), more recently, several research initiatives have been promoted focusing on specific waste streams. For instance, a number of studies were undertaken to identify key drivers of food wastage and to quantify food waste generation (e.g. EU FUSIONS 2015; FAO 2015, 2013, and 2011; EC 2010a).

These initiatives provided fairly accurate information over European food waste generation quantities and management routes, but they did not always deliver comprehensive and comparable information on the sustainability of food waste management and on ways to mitigate negative consequences at the levels of the three so-called "sustainability dimensions": environmental, economic and social. Most studies, in fact, only focuses on one specific sustainability dimension, e.g. only the environmental (Nakakubo et al., 2012; Kim and Kim, 2010) or the economic (Kim et al., 2011) dimension.

This is currently changing due to increasingly challenging sustainability targets and requirements enforced by recent legislation (e.g. EC, 2011a and 2011b) and the 2015 Circular Economy package (EC, 2015a) – which are boosting research also on the methodological side, e.g. towards developing methods to evaluate resource efficiency and sustainability. Room for improvements exists, especially to harmonize existing assessment approaches and adapt them to the specific context of food waste management. This report is an attempt to reduce such gaps. Its content is in line with a publication from Manfredi & Cristobal (2016) where methodological aspects related to sustainability assessment of food waste management are further analysed.

Building on existing knowledge, most recent methodological advancement and policy needs, the work presented in this report aims at providing:

1. An overview of how the sustainability (environmental, economic, social) of food waste management can be evaluated in a life cycle perspective (Chapters 2);
2. An overview of European food waste generation and management routes (Chapter 3);
3. A life-cycle based framework approach to quantitatively assess the environmental and economic sustainability performance of European food waste management options (Chapter 4);
4. Examples of quantitative assessment of the environmental and economic performance of food waste supply chains, via a numerical application of the proposed methodology (Chapter 5).

The report is aimed at policy makers and waste managers at local, regional or national level.

2. European food waste: generation and management

To understand the size and importance of the European food waste sector, the following aspects should be considered:

- Definition of the supply chain(s) in which food waste is generated (within the so-called "food supply chain" – FSC);
- Identification of the main causes and drivers of food waste generation;
- Quantification of food wastage throughout the FSC.

This report is primarily focused on the European food waste situation, however, as the European FSC is obviously connected to the broader international food waste context, data and information of wider geographical scope will also be included, as necessary. After that, a description of the most relevant available technological options to manage food waste is provided.

2.1 Food waste generation: overview of quantities and drivers

The FSC is, in general terms, a complex chain that includes all the stages in which food can be wasted or lost, from the production of food for human consumption till the consumption itself (see Fig. 6). According to FAO (2011), five different stages can be identified:

- Agricultural production: it includes harvesting, fishing and breeding (for the case of animal feedstock) and sorting of the products;
- Post-harvest handling and storage: it includes handling (for fish it includes the icing and the packaging), storage and transportation between the production place and the distribution;
- Processing: it includes industrial or domestic processing - washing, peeling, slicing, boiling, baking, canning, smoking, etc.;
- Distribution: it includes all the market system, at e.g. wholesale markets, supermarkets, retailers and wet markets;
- Consumption.

The main causes of food waste generation have been extensively studied in recent years. For instance, the FUSIONS project (EU FUSIONS, 2015) identified 286 current causes of food wastage (and the drivers – see Box 2); these are grouped into a number of categories reflecting different ways through which food can be wasted:

- inherent characteristics of the food (e.g. unavoidable production, low perishability, low predictability of supply, temperature sensitivity);
- technological (e.g. non-use or sub-optimal use, misshapen products);
- supply chain and management inefficiency (e.g. bad logistics, bad stock management - overstocking, bad supply coordination);
- social factors, dynamics, attitudes and lifestyles (e.g. single-person households, take away food, portion sizes, awareness of food waste, taking leftovers home form restaurants, marketing standards and strategies);
- individual behaviors, habits and consumer expectations towards food (e.g. good aspect, preferences, bad planning, freshness, season, time).

BOX 2 – Drivers of food waste generation – FUSIONS project[10]

Detailed and aggregated classifications of the drivers (from which the causes are originated) have been reported in FUSIONS project. Drivers can influence positively or negatively the food waste generation dynamics (i.e., the decrease or increase in quantity, respectively) and they are classified in:

- Technological - technology development. Grouped in:
  - drivers inherent to characteristics of food and its production and consumption, where technologies are limited;
  - drivers related to collateral effects of modern technologies;
  - drivers related to sub-optimal use of, and mistakes in the use of available food processing technology and chain management.
- Institutional - food supply chain management at business management/economy level legislation/policies). Grouped in:
  - drivers not easily addressable by management solutions;
  - drivers addressable at macro level;
  - drivers addressable within the business units;
- Institutional - food supply chain management at legislation/policy level). Grouped in:
  - drivers concerned with the legislation derived from the agricultural policy and product quality;
  - drivers concerned with the legislation derived from food safety, consumer health and animal welfare policies;
  - drivers concerned with the legislation originated by waste, tax and other policies;
- Social - consumers' behaviors and lifestyles. Grouped in:
  - drivers related to social dynamics which are not readily changeable;
  - drivers related to individual behaviors which are not readily changeable;
  - drivers related to individual behaviors modifiable through information and increased awareness;

Theoretical estimations of global and European food waste generation are provided in different studies. Figure 2 provides a summary of the information collected from two of the studies analysed (FAO, 2011; EC, 2010a). Calculations based on the reported data on food waste generation from these two studies reveals considerable differences in food waste generation in Europe: 195 million tonnes based on FAO (2011) compared to 90 million tonnes reported by EC (2010a). This difference largely depends on the different geographical areas considered (FAO considers Europe as "Europe + Russia"; EC considers Europe as EU27), but also depends on the different definitions of what food waste is/includes (e.g. FAO also accounts for the production stage of the FSC, while EUROSTAT does not).

FAO (2011) reports production volumes for commodity groups (e.g. cereals, meat, roots and tubers, fish) in their primary forms for 7 world regions. The total worldwide food production was around 6.7 billion tonnes (i.e. 6.7×10^9 t) in 2007, 37% of which (i.e. 2.51 billion tonnes) is estimated to become food waste along the FSC. From this, it can be estimated that Europe[11] produces annually about 419 million tonnes of food waste (reference year 2007), corresponding to around 17% of the global food waste production. Figure 2 shows the estimations breakdown for the FSC stages. According to the definition of food waste (see Box 1), FAO considers allocation factors to determine the part diverted to human consumption and conversion factors to determine the edible

[10] http://www.eu-fusions.org/

[11] Again, FAO (2011) considers Europe as "Europe+Russia"

part. This fact results in around 1.3 billion tonnes of food waste generated globally, approximately 15% of which are generated in Europe (195 million tonnes). The breakdown is again shown in Figure 2.

BOX 3 – Potentiality in quantifying food waste generation

Given the high uncertainty affecting estimations of food waste generation, different "potentialities" are often considered in literature. As defined in (EC, 2010a) they are ranked from bigger to lower potential size:

- theoretical which is the total food waste that could be generated;
- technical which is the part of the theoretical food waste that could be properly separated and collected;
- economic which is the part of the technical food waste that is economically viable to use after collection and separation;
- sustainable which considers the waste hierarchy in which a significant portion of food waste could be avoided.

A report of the European Commission (EC, 2010a) based on data of food waste generation on EUROSTAT estimates around 90 million tonnes of food waste for EU-27 in 2006. In this study the food waste generated during agricultural activities is not included in the reported data, which are shown with split by sector involved in the FSC instead of by FSC stage (e.g. the manufacturing sector is considered to include the post-harvesting and processing stages of the FSC). Other estimations from EUROSTAT data using a different method[12] reports 116 million tonnes of total food waste for EU-27 in 2006 (Kretschmer et al., 2013), which is closer to the FAO value. Per capita values of all the estimation methods are shown in Table 1.

[12] In this case estimations are calculated as tonnes of wastes reported in 'total animal and vegetal wastes' minus the ones reported in 'animal faeces, urine and manure'. There is some uncertainty in the estimations since green wastes could be accounted for.

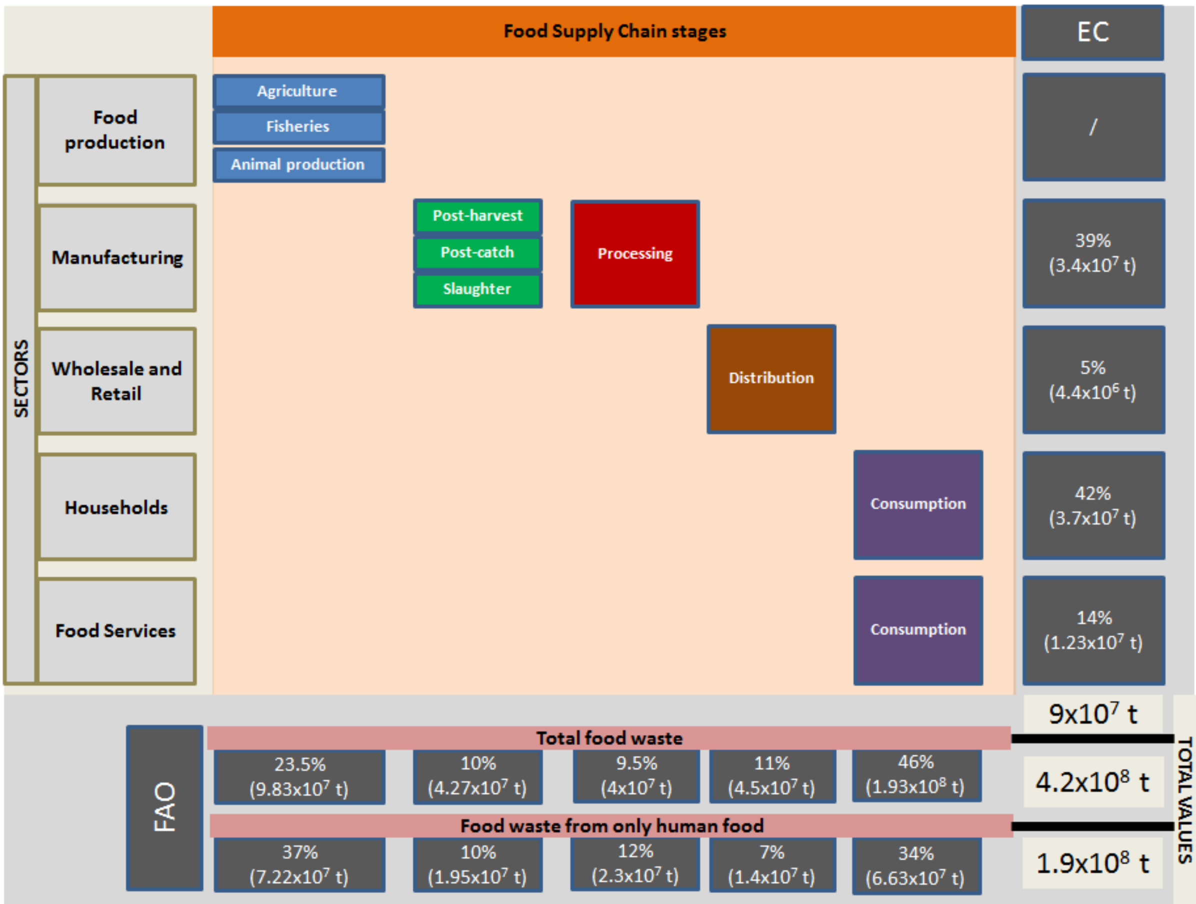


Figure 2: Total food waste estimation for Europe (based on FAO, 2011; EC, 2010a)

As shown in Figure 2, consumption is the FSC stage that contributes the most to the food waste generation (i.e. from 34% to 56% depending on the study). According to the EC (2010a), around 80% of the overall food waste is generated in the manufacturing and household consumption which points out that these are the sectors having the biggest potential for reduction of food waste generation.

Table 1: food waste estimations: absolute values and per capita

REFERENCE STUDY	Year	Geographical scope	Total (t/year)	Per capita (kg/year/capita)
Total food waste along FSC (FAO)	2006	EU27+Russia	$42x10^7$	659
Food waste (human consumption only) along FSC (FAO)	2006	EU27+Russia	$19x10^7$	298
Food waste (human consumption only) along FSC excluding production stage	2007	EU27	$9x10^7$	182
Food waste (human consumption only) along FSC excluding production	2007	EU27	$11.6x10^7$	234

stage (different estimation method from EUROSTAT[13])				
Food waste (human consumption only) along FSC excluding production stage (EUROSTAT[14])	2014	14 MS in EU27	-	116

*Note: Population EU-27 = 495090300 hab. Population EU-27+Russia=637311300 hab. (based on data from EUROSTAT, 2007).

All these estimations are based on weight but generation of food waste can also be understood in terms of "loss of calories" for human consumption. This approach is known as "field to fork" and considers the losses occurring in the supply chain from the harvest stage till the consumption stage measuring them in Kcal per person per day. In this sense, again, different figures can be found in literature ranging from 50% (Kummu et al., 2012) to 57% (Papargyropoulou et al., 2014). Kummu et al., (2012) considered that from the total production of food crops, being 3938 kcal/cap/day, in the end, only 50% is actually used by humans and the other 50% is used for animal feed, wasted within the FSC and directed to seed or other use. Papargyropoulou et al., (2014) start from the value of 4600 kcal/cap/day and conclude that the final loss is close to 57%.

2.2 Food waste management: overview of technological options

This section provides a brief overview of the main existing technological options to manage food waste, regardless of their environmental performance and/or the position they occupy in the "waste hierarchy" established in the Waste Framework Directive 2008/98/EC. A management system that combines several of the presented options is also possible (Vandermeersch et al., 2014). A summary description of each option is included below.

Other processes are also needed throughout the food supply chain, such as collection and transport (see Annex 1 for further info on food waste collection and transport); these are equally not dealt with in this section. Methodological guidance on how to identify the most sustainable management options for food waste is instead provided in Chapter 4.

2.2.1 Anaerobic digestion

Anaerobic digestion is a biochemical pathway able to convert almost all sources of biomass (including wet materials such as organic wastes and animal manure) to a highly energetic energy carrier referred as biogas. The anaerobic digestion process consists of four phases (hydrolysis, acidogenesis, acetogenesis, methanogenesis) where microorganisms transform sequentially the different complex organic substrates to biogas composed by methane (CH_4) (around 50-70%), carbon dioxide (CO_2) (around 30-50%) and traces of hydrogen sulfide (H_2S) and water vapor.

There are multiple configurations and designs for digesters depending on:

[13] In this case estimations are calculated as tonnes of wastes reported in 'total animal and vegetal wastes' minus the ones reported in 'animal faeces, urine and manure'. There is some uncertainty in the estimations since green wastes could be accounted for.
[14] This is based on preliminary results (from 14 MS) from the "PLUG IN EXERCISE" on food waste coordinated by EUROSTAT in 2014/2015.

- Loading rate in total solids content: wet digesters that operates with less than 15% total solids in the reactor and dry digesters that operates with around 25-30% total solids.
- Operating temperature: thermophilic digesters that operate in a temperature range of 50 – 65°C and mesophilic digesters that operate at around 35 - 40°C.
- Number of reactors used: two-phase digesters that separate in two reactors the production: acid phase (production of organic acids) and methane phase (production of methane), allowing for a better optimization of operation conditions in each stage. On the other hand, one-phase digesters that present only one reactor where all reactions take place with operating conditions that suit all of them.
- Feeding method: batch digesters that are loaded and reaction take place for a certain period and continuous flow digesters that are fed and discharged in continuous manner.

A common practice is to operate digesters with co-digestion of two or more types of feedstock (i.e. animal manure as the primary feedstock adding digestible organic waste to increase gas production).

The biogas produced can be either combusted for heat and power (CHP) generation or upgraded to produce a substitute of Natural Gas (SNG), known as biomethane, which can be injected in the grid (to be combusted) or utilized as vehicle fuel:

- For CHP generation biogas can be combusted in internal combustion engines (either spark ignition or compress ignition with dual fuel configuration), achieving total efficiencies up to 77-80%. It is also common its use in gas and micro-gas turbines (with total efficiencies between 63-71%). Its use in fuel cells (overall efficiency up to 80%) is in early stages of research and development (mainly solid oxide fuel cell (SOFC) and molten carbonate fuel cells (MCFC)) (US EPA, 2014).
- Concerning the upgrading technologies, there exist different types: absorption (either physical or chemical), pressure swing adsorption (PSA), cryogenic technology and membrane separation (Petersson and Wellinger, 2009). The most common method used to decrease the CO_2 content in biogas is the physical absorption using water scrubbing due to the larger solubility of CO_2 and H_2S in water compared to the solubility of CH_4.

A by-product of the anaerobic digestion is the digestate that can be used directly as a fertiliser or composted to enhance its characteristics. Figure 3 shows a scheme of this pathway.

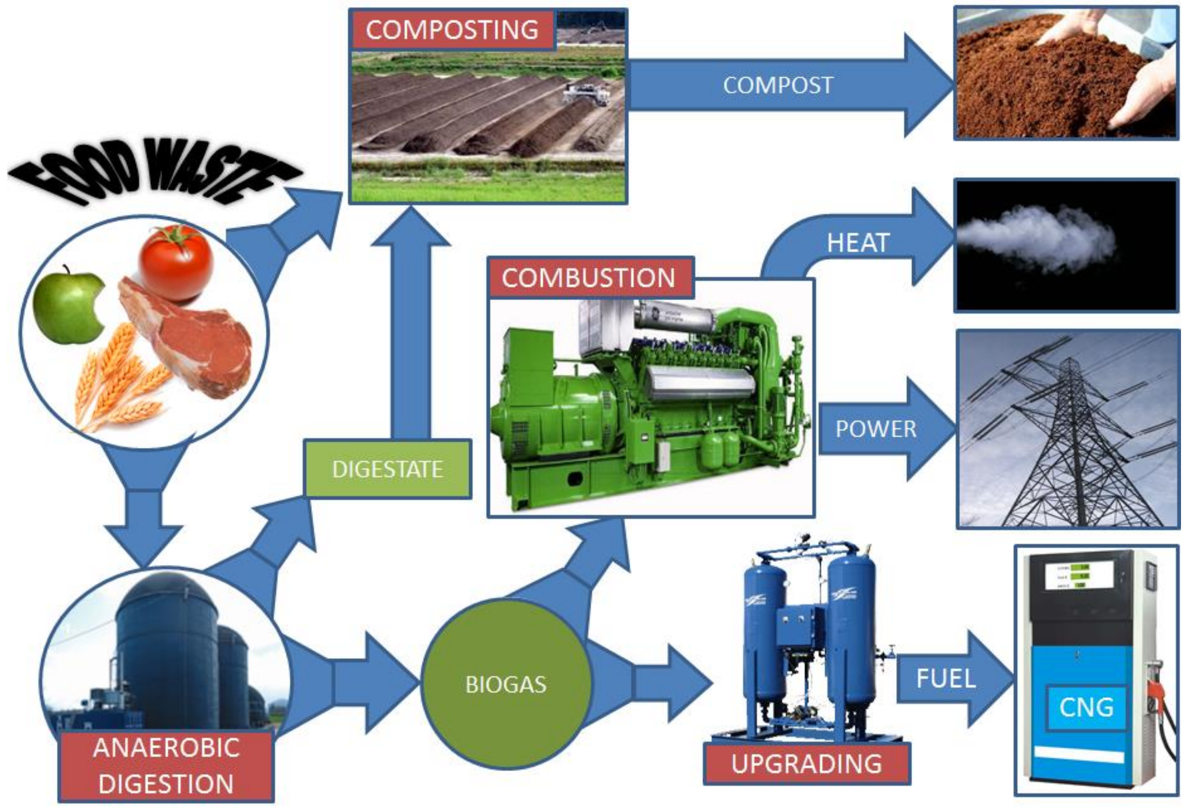


Figure 3: Anaerobic digestion and composting pathway

2.2.2 Composting

Composting is a natural process where microorganisms and fungus decompose in aerobic conditions organic material into a humus rich soil amendment known as compost. The process is exothermic and the heat is maintained at thermophilic conditions (around 50-65°C) and for a sufficient time in order to destroy harmful microorganisms and stabilize the compost. Compost can have an added value on fields as it can improve soil structure and quality by adding organic matter, nutrients and diversified biologic microorganisms. There are mainly two configurations for centralised composting:

- Windrow composting is done by pilling the food waste in long rows which are turned regularly to homogenize the temperature of the pile, mix in and improve oxygen content and porosity. It can be done in open air facilities that present the risk of producing odours and generating uncontrolled emissions of greenhouse gases such as methane, or in enclosed buildings that allows the better control of odours and emissions;
- In-vessel composting is done confined in containers or vessels in which air flow and temperature can be carefully monitored and adjusted. The exhaust gases produced during the process are collected and treated normally through a biofilter.

2.2.3 Incineration

Incineration is the controlled combustion of the waste material with a surplus of air. It is normally used for treating heterogeneous waste such as mixed food and packaging waste with the objective of reducing volume, weight and hazard of the wastes. The high content of moisture in food waste (i.e. around 70% on a mass basis) can be an issue in incineration processes since they require less than 30% of moisture for being efficient. Therefore, a pre-processing phase of drying is usually required.

There are different combustion configurations and technologies:

- The fixed-bed combustion is the simplest and most used technology, with grate furnaces or underfeed stokers. Biomass is burned in a fix bed in the presence of the primary air and the gases produced are burned usually in a separated zone with added secondary air.
- A more recent technology is the fluidized bed combustion in which the biomass is combusted in a solid-bed material that is fluidised passing through it the primary combustion air.

The steam produced in the incineration can be used for heat and/or electricity generation. Combined heat and power (CHP) facilities present higher efficiencies compared to separate heat and electricity systems. The most used technology for CHP is the steam turbine system. The biomass is burned in the combustion chamber to produce high-pressure water steam through a boiler that causes the rotation of the steam turbine. The power created in the turbine is converted to electricity with the generator, while steam is extracted from the turbine and used in homes or industrial processes.

One of the problems of incineration is the possible generation of a broad range of gaseous pollutants, including NO_2, micro-pollutants and some toxic and persistent organics such as dioxins and furans. Strict emission standards must be met when incinerating wastes (EC, 2000).

2.2.4 Landfilling with landfill gas collection

Provided that the requirements set by the Landfill Directive 1999/31/EC (EC, 1999) in terms of reduction of landfilling of biodegradable matter are met, food waste can be disposed in landfills. Similar to the biogas generated in anaerobic digestors, the generation of landfill gas (a mixture of CH_4, CO_2, and a number of trace gases) is the result of anaerobic degradation of organic matter. Such gas is typically collected and either flared in-situ, or recovered for energy generation (electricity and/or heat). Different technologies exist that aim at reducing the duration of the degradation processes, optimising the production and collection of the landfill gas, as well as reducing gaseous and liquid emissions to the surrounding environment through implementation of pollution control measures. These are often referred to as active landfill technologies and include e.g. bioreactor landfills and flushing landfills (e.g. Manfredi & Christensen, 2009).

Landfilling without collection of landfill gas is no longer allowed and a progressive outphasing of biodegradable waste landfilling is established by the Landfill Directive (EC, 1999), as anticipated above. Even when landfill gas is used for CHP generation, upgraded to natural gas or flared to reduce the GHG emissions[15], landfilling may lead to a variety of environmental impacts caused by fugitive gaseous emissions from the landfill surface, un-optimal flaring[16], emission of leachate to groundwater and surface water bodies, littering, dust, odours and noise. Typical environmental impacts from sub-

[15] When landfill gas is flared (i.e. combusted) most of the CH_4 in the gas is converted to CO_2. This CO_2 is typically considered of biogenic origin, thus (typically) it is considered neutral to climate change.

[16] If the combustion is not "complete", part of the CH_4 is released to the atmosphere contributing to climate change (CH_4 contributes to climate change 23 times as much as CO_2 in a 100-year time horizon)

optimal landfilling include climate change, ozone depletion, depletion of water resources, ecosystem toxicity, and human toxicity.

2.2.5 Dry/wet feeding

In some cases, the production of feeding for livestock animals is another alternative for valorising the food waste (FAO, 2004). Some food waste requires heat treatment to reduce the risk of foreign animal diseases and to eliminate possible harmful pathogens. This can be achieved in two ways: dry and wet feeding. The most common one is the dry feeding where food waste is dewatered and then injected into a dryer for sterilizing and dewatering at the same time. Conventional dehydration is done applying heat but it can also be done by fermentation or fry cooking – food waste is cooked in waste vegetable oil under reduced pressure at relatively low temperatures, e.g. 110°C). The resulting feed can be used directly or it can be used as high quality ingredients for commercial concentrate feeds.

Wet feeding is typically used with high moisture food waste. It does not require dehydration and so lower cost and little protein is lost during the process. It can also involve a fermentation step to decrease the pH, which results in a prolonged durability of the feed product. The major concerns related with this option are handling, storing and transportation of the feed product, due to investment and logistics.

2.2.6 Pyrolysis and gasification

A wide range of thermal treatments, fairly well established for coal, exist and are promising for food waste, though still at an early stage of implementation. Among these, pyrolysis and gasification present a good potential with respect to power generation from biomass (Ahmed & Gupta, 2010), though they present technical and economic challenges.

Pyrolysis is a thermochemical process that, by heating in the absence of oxygen, converts organic material to solid, liquid and gaseous fractions referred as charcoal, bio-oil and biogas, respectively. The yield of the different products is dependent on the process variables and the properties of the feedstock. Among the products, bio-oils are the preferred one being charcoal and biogas by-products of the process. The normal use of bio-oils is to produce heat and power. They can be also upgraded to biodiesel via hydrogenation or gasified together with the char slurries to produce syngas. Depending on the operating conditions used, pyrolysis processes can be divided in conventional, fast and flash. Conventional pyrolysis has been largely used for production of charcoal. The temperature is lower (i.e. around 400°C) and the vapor residence is high boosting the formation of solid char and gases. In fast pyrolysis temperature is higher (i.e. about 500°C), the vapor residence is lower and cooling is faster, giving high liquid yields (50%). Flash pyrolysis is equal to fast pyrolysis with lower vapour residence time enhancing the yield of the liquid fraction (75%).

Gasification, as the pyrolysis, is a thermochemical process where carbonaceous materials are converted into a gaseous fuel (either called syngas or producer gas) consisting of carbon monoxide (CO), Hydrogen (H_2) and traces of methane (CH_4) and carbon dioxide (CO_2). The reaction takes place at high temperature, without combustion, and using media such as air, oxygen and steam. Depending on the feedstock and the gasification technology used, a drying phase should be done to reduce moisture since high levels (i.e. 45 – 55%) can obstruct gasification. The main application is for combined heat and power generation, where the producer gas is used as fuel and therefore, the most important characteristic of the gas is its calorific value (the higher, the better).

3. Methodological foundation and approach

The core part of this report is the provision of a framework methodology to quantitatively assess the environmental and economic sustainability performance of food waste management options (Section 4.3).

The conceptual and methodological foundation of the framework methodology is Life Cycle Thinking (LCT). Among the wide range of existing LCT-based methods and standards, Life Cycle Assessment (LCA) is used to provide evaluation of the environmental dimension. Regarding the economic dimension, different LCT-based indicators are considered, such as the "actual costs" of waste management and the "gate-fees" (representing the charge paid by the authorities to a waste treatment operator). Multi-objective optimization and Pareto optimality techniques are also used to help identify most sustainable management options for food waste. Sections 3.1 to 3.4 provide more details on these methodological aspects.

3.1 Evaluation of food waste management in a life-cycle perspective

Life Cycle Thinking (LCT) can be intended as a conceptual approach that aims at identifying improvements and lowering impacts of any goods/services at all stages of the life cycles associated with such goods/services, i.e. from extraction of raw materials to end-of-life. Using an LCT-based approach helps to avoid the situation of resolving one problem while creating another, the so-called "shifting of burdens".

In the field of waste management, using LCT to help identify the options that delivers most environmentally sound outcomes is supported by several legislative documents and, in particular, by the Waste Framework Directive 2008/98/EC (EC, 2008). This directive also allows for deviations from the general "waste hierarchy" principle, provided that LCT-based evidence shows that these deviations lead to a better overall environmental outcome[17] (Figure 4).

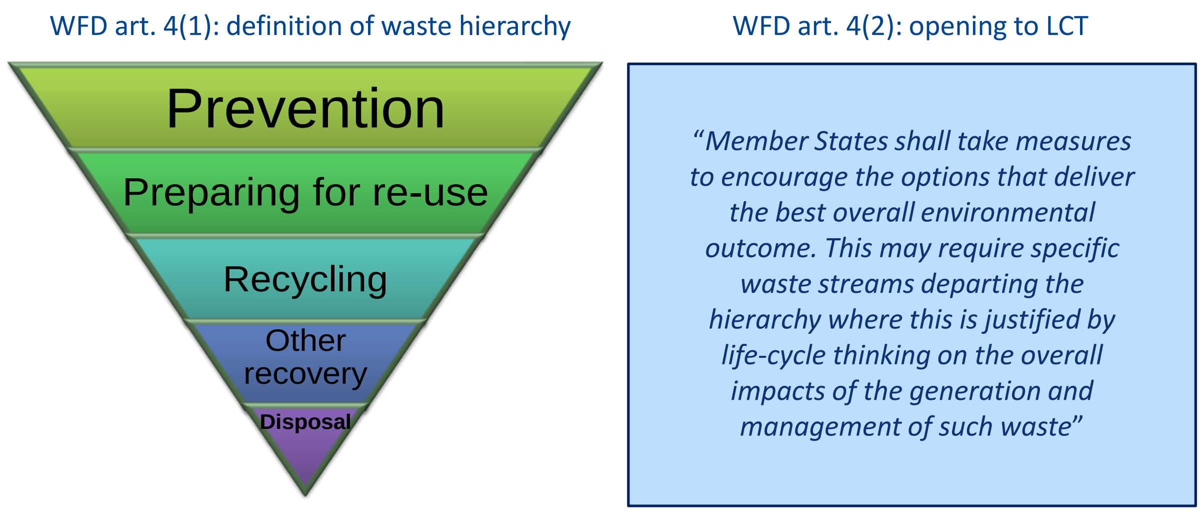


Figure 4: Waste hierarchy and Life Cycle Thinking (LCT) as introduced by the Waste Framework Directive 2008/98/EC

[17] For instance, if LCT based evidence exist that in a certain waste management context it is environmentally preferable to co-incinerate food waste rather than treating them in a composting plant, then the WFD allows for such deviation from the waste hierarchy.

Waste prevention and reuse (the top two priorities indicated by the waste hierarchy) are not explicitly addressed in the framework methodology. The area of focus in fact includes the subsequent steps of the waste hierarchy, thus specifically addresses the flow of food waste that could neither be prevented nor reused, but needs to be managed/treated.

Life Cycle assessment (LCA) can be intended as a transposition of LCT into quantitative terms. LCA – as defined by the ISO 14044 (ISO, 2006) and further specified in the International Life Cycle Data System (ILCD) Handbook (EC, 2010) – is a decision support tool widely used to evaluate the environmental impacts arising from any goods/services. LCA is also extensively applied to evaluate waste management systems, scenarios and strategies, with a view of identifying key life cycle stages, key flows of material/energy from an environmental perspective, as well as improvement possibilities (e.g. Manfredi & Pant, 2013; Manfredi & Goralczyk, 2013).

The 2011 JRC guidelines "Supporting Environmentally Sound Decisions for Waste Management" (EC, 2011d) provide detailed guidance on how to apply LCT and LCA to support environmentally sound decisions for waste management (Figure 5). This also includes guidance at the level of specific waste streams, including bio-waste (see BOX 1 – definitions).

Figure 5: 2011 JRC Guidelines on waste management, life cycle thinking and assessment

It should be clarified that beyond LCA, several other decision support tools – whose objectives, target audiences and levels of standardization vary considerably – find their conceptual foundation in LCT, virtually allowing comprehensive evaluation of the three pillars of sustainability (environmental, economic and social). These include e.g. Life Cycle Costing (LCC), Cost-Benefit Analysis (CBA), Hybrid LCA (H-LCA), Social Life Cycle Assessment (S-LCA), Ecological Footprint, Greenhouse Gas Protocol, and the European Commission Environmental Footprint (EF) method. The use of the latter has been recommended by the Commission to undertake studies aiming at quantifying the environmental performance of products and organizations (EC, 2013c and 2013d).

As anticipated, this report includes examples of LCA-based modelling of selected food waste management scenarios (Chapter 5). Such modelling has been conducted with the software EASETECH (Environmental Assessment System for Environmental TECHnologies). EASETECH (Clavreul et al., 2014) is an LCA-based model for assessment of environmental technologies developed by the Department of Environmental Engineering of the Technical University of Denmark (DTU-Environment). EASETECH was released in 2014 as a follow up of the software EASEWASTE, also developed by DTU-Environment. Its primary aim is to perform LCA of complex systems handling heterogeneous material flows. EASETECH models resource use and recovery as well as environmental emissions associated with environmental management in a life-cycle context. The main novelties compared to other LCA software are as follows. First, the focus is put on material flow modelling, as each flow is characterised as a mix of material fractions with different properties and flow compositions are computed as a basis for the LCA calculations. Second, the tool has been designed to allow for the easy set-up of scenarios by using a toolbox, the processes within which can handle heterogeneous material flows in different ways and have different emission calculations. Finally, tools for uncertainty analysis are included, enabling the user to parameterise systems fully and propagate probability distributions through Monte Carlo analysis.

3.2 LCA-based modelling of waste management options: overview of key assumptions

Conducting an LCA involves making a number of decisions, assumptions and choices that exert an influence on the final results. As a consequence, substantial differences are found in LCAs even when their areas of focus (e.g. food waste management) are similar or the same. The most common choices in LCA are related to the following aspects:

- The decision context (i.e. the type of decisions that the results of the LCA can or cannot support), which has a direct influence on whether the attributional or the consequential modelling is used. The choice between these two LCA modelling approaches is fundamental, as it influences the way the modelling is conducted (e.g. the type of input-data used, and the way of accounting for the benefits of energy recovery) and the meaning and usability of the LCA results (in particular the type of decisions that results can support)[18].
- If consequential modelling approach is used, choices and assumptions have to be made to identify any "displaced technologies[19]" and/or products (e.g. what exact type of electricity is assumed to be displaced by the electricity produced within the system boundary).
- The exact definition of the functional unit (FU) of the assessment (i.e. the function or service that the system being analysed is assumed to provide), the way the boundary of the evaluation is defined (i.e. what is accounted for, and what is not), the choice of the reference flow associated to the FU (e.g. 1 tonne of food waste).
- The choices made at the level of the Life Cycle Impact Assessment (LCIA) phase, in particular the choice of the impact assessment models and methods, which influences – among others – the comprehensiveness of the environmental assessment (e.g. in terms of the number of environmental impact categories that is accounted for) and the indicators associated to each impact category (thus the units used for expressing a certain impact category).
- The way of accounting for emissions and sequestration of carbon; if a distinction is made between fossil and biogenic carbon; if and how delayed emissions are considered.

In the context of LCA applications to waste systems and strategies, the evaluation of the environmental consequences arising from management of food waste is becoming increasingly common. The factors and methodological choices that mostly influence the results of LCAs involving food waste are in principle the same of those of any LCA of waste management systems (as from the above list). However, LCAs that include food waste management may become particularly complex as, in addition to technical

[18] These aspects are presented and analysed in details in the ILCD Handbook – General Guidance (EC, 2010c)
[19] In LCA terms, these are referred to as "marginal technologies"

processes, also biological processes take place during the waste management chain. These biological processes – which are highly dependent on local and interlinked factors such as soil profile, rainfall, and temperature – should in principle be carefully modelled and accounted for. In fact, they can lead to unwanted emissions and may reduce the otherwise good potential for recovery of energy and nutrients that can be achieved from proper management of food waste. While such factors are often included in LCA-based assessment of food waste management, the specific way of accounting for them can vary considerably as it depends on several non-obvious choices and assumptions that the LCA practitioners should make (Bernstad and la Cour Jensen, 2012).

Such complexity leads considerably different estimations of environmental impacts reported in reviews of LCA studies on food waste. Corrado et al. (2016) analyse the different assumptions made in LCA studies when modelling food losses. Bernstad and la Cour Jensen (2011 and 2012), reported large variations of the estimated impacts on Climate Change, ranging from net impacts to net avoided impacts (Table 2). These variations are shown to be determined mostly by different definition of system boundaries, differences in the input data, and different ways of accounting for the benefits from energy recovery. This in turn indicates that comparisons among LCAs are often biased, if at all possible. In other words, such differences mainly indicate the lack of homogeneity at the level of key factors and assumptions rather than actual differences in the environmental performance. Reported results of sensitivity analysis confirm that the assumptions that mostly influence results of LCAs on food waste are those made for: (1) emissions and storage of carbon and other nutrients; (2) emissions from composting and use on land of compost; (3) energy generation (e.g. from anaerobic digestion) and substitution.

Table 2: estimated impacts on Climate Change from treatment of 1 tonne of food waste, based on review of 25 studies (adapted from Bernstad and la Cour Jensen, 2012)

Impact range for Climate Change	Incineration	Landfilling	Anaerobic digestion	Composting
Min (kg CO_2-eq)	-250	400	-400	-50
Max (kg CO_2-eq)	600	1200	400	850

3.3 Evaluation of costs associated with food waste management

In a life-cycle perspective, the total cost for the management of a certain mass (e.g. 1 tonne) of food waste is given by the sum of the collection/transportation costs and the treatment costs. However, it should be pointed out that in practice the introduction of food waste collection along with other fractions reduces the frequencies of collection of the residual waste. In order to reflect this aspect in the estimation of costs, a suitable indicator is e.g. Euro/capita/year/tonne of residual waste. Anyway, willing to assess costs relative to just one tonne of food waste (as in the case study presented in Chapter 5), then collection costs can simply be evaluated in Euro/tonne of food waste.

Evaluations of costs of waste management options can be conducted in different ways. However, these are typically based on one of the following two approaches, or a combination of them:

- Approach based on the "actual costs", which are calculated considering the summation of costs associated with all processes included in the system boundary and necessary to provide the chosen LCA Functional Unit (FU).

- Approach based on the "gate-fee", which is the charge paid by the authority (e.g. a local authority such as a municipality) to a waste treatment operator to provide a service (in this case the management of wastes). It usually covers the capital costs, operation, maintenance, labour, along with profits and final disposal of any residue created in the process.

The use of one approach over another firstly depends on the decision context. For instance, if a municipality is seeking to identify the best performing management options, then using the gate-fee approach is particularly meaningful and straightforward as gate-fees express the cost that a municipality has to pay to the company treating the waste. However, to apply this approach, the practitioner of the study should in principle estimate and predict such gate-fees, which entails (possibly complex) analyses of local market dynamics to determine the relationship between such gate-fees and prices. Gate-fees have in fact higher chance to change over time than the costs of specific facilities, in the sense that they are likely to change even if the actual cost remain the same (EC, 2001). However, in order to have the complete estimation of the costs supported by e.g. a municipality for food waste management, other elements should be accounted for, such as collection and transport costs as well as any existing tax or subsidy. Further explanation is given in Section 4.4.

If the analysis is conducted to help identify best performing options at a MS or European level, the costs-based approach is perhaps more appropriate, as costs better reflect the resources that are needed to implement a given waste management system or strategy. Such estimations are typically conducted in an LCT-perspective using Life Cycle Costing (LCC) (e.g. Martinez-Sanchez et al., 2015).

BOX 4 – Examples of innovative indicators for economic evaluation:

Economic viability is a key aspect of intensive source separation schemes. It is important to consider that food waste is usually collected together with other waste fractions, and it must be noted that the evaluation of management costs of a single fraction like food waste is not enough to have a comprehensive view, as all the other fractions contribute not only to treatment costs but also with revenues, like those related to the selling of recyclables such as paper or plastics.

An important survey was completed by Lombardy Region in March 2010[20] focusing on the economics of collection schemes in the whole Region, comparing 1546 municipalities for a total of 10,000,000 inhabitants. In Lombardy, the delta cost of mixed MSW treatment (using incineration or MBT - Mechanical Biological Treatment) is very low, around 20 €/t (i.e. food waste treatment in composting or anaerobic digestion plants is 80 €/t, whilst the treatment of residual waste was around 100 €/t at the time of the study, now decreasing). In Lombardy, garden waste is not collected commingled with food waste, as typical in other regions of Italy, and it is delivered to municipal collection points and then composted in low tech plants with a gate fee of 30 €/t.

The key advantage of performing this cost assessment study in Lombardy is the fact that there is the simultaneous presence of all possible situations both in terms of recycling rate, ranging from 20% to more than 70%, and of geographical features, from very small rural villages to high population density in very large municipalities including Milan. It was possible to compare these very different cases by building a new indicator, i.e. overall costs per equivalent inhabitant, which flattens the differences related to high tourism or presence of many commercial activities generating urban waste, and

[20] Valutazione statistico economica dei modelli di gestione RU in Lombardia, http://www.reti.regione.lombardia.it/shared/ccurl/613/648/Valutazione_modelli_GestioneRU.pdf , then updated in the new Regional Waste Management Plan.

removing street sweeping costs which are not related to the waste collection model.

The main outcome of this study is that with the above mentioned boundary conditions, increasing the recycling rate up to more than 70% (achievable only with food waste collection), the normalised overall costs do not increase. This is due to the fact that collection costs increase slightly, as well as common costs (such as costs connected with the civic amenity sites), but treatment costs diminish due to the higher revenues from recyclables as shown in Figure 6. Moreover, increasing recycling rate, doesn't increase the costs even in very rural or low population municipalities.

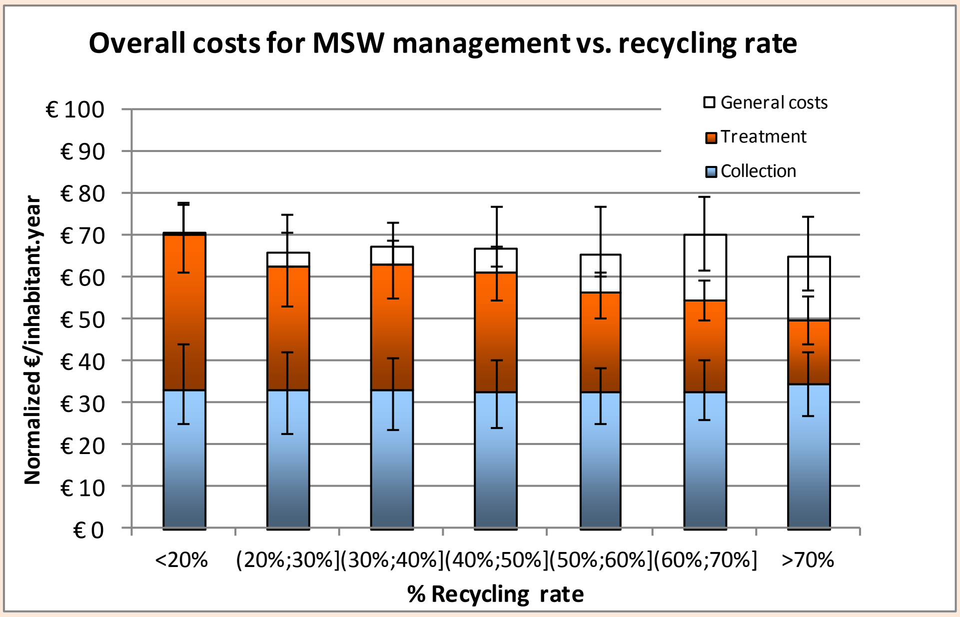


Figure 6: Comparison of about 1200 municipalities in Lombardy: average of normalized overall costs for subsets of municipalities having the same range of recycling rate. Costs are split into collection (blue), treatment (orange), general costs (white).

3.4 Evaluation of the social dimension: an overview

Social Life Cycle Assessment (S-LCA) focuses on identifying and assessing social impacts associated with product life cycles (UNEP/SETAC, 2009). S-LCA tries to quantify social aspects that may affect stakeholders, such as workers or communities, either negatively or positively. It attempts to shed light on the social dimensions of product supply chain stages and the possible impacts they have on social conditions. Together with environmental LCA and Life cycle costing (LCC), the three strive towards a holistic assessment of sustainability impacts in supply chains. As these are complementary to each other, S-LCA has an important role in sustainability assessment.

As mentioned in the previous section, food waste produced during the consumption stages accounts for a relatively high share within the supply chain. Prevention of this food loss can be tackled through communication and behavioural change aimed at consumers and retailers. When food waste is generated, its treatment has social impacts. Environmental LCA has been widely used and recognized for its utility in assessing waste treatments options. LCA studies of waste management are present in literature and they often cite the need of assessing social aspects (Cherubini et al., 2009; Del Borghi et al., 2009; Ekvall et al., 2007; Manfredi et al., 2011). Studies in the broader social literature mainly focus on waste prevention campaigns (Bartl, 2014; Cox et al., 2010; Dururu et al., 2015; Wilson et al., 2012) and the attitude of people towards a specific treatment (Bernad-Beltrán et al., 2014; del Cimmuto, 2014; Spies, 1998).

However, only few studies address waste treatment using S-LCA and make use of indicators (e.g. Rybaczewska-Blazejowska, 2013). These studies highlight the importance of encompassing social impacts into assessments of various waste treatment options. Moreover, more studies are needed in order to increase the robustness of indicators and social impact assessments of waste in different contexts. These studies also indicate the necessity for more indicators to be developed and tested, although they converge on some main issues/indicators.

3.5 Optimization and Pareto front

The proposed methodology also makes use of multi-objective optimization and Pareto optimality techniques to help identify most sustainable scenarios. Multi-objective optimisation involves two or more objective functions to be optimized simultaneously. When these objective functions are incomparable (i.e., are expressed in different units), or at least partially conflicting (i.e., there is a trade-off between them and the achievement of a better value in one objective worsens the other (Messac et al., 2013)), then a single solution that simultaneously optimises each objective does not exist, but rather several optimal solutions can be found. The solutions to the multi-objective problem are known as Pareto optimal (also known as non-dominated or Pareto efficient). They can be graphically represented in a Pareto front (see Box 5).

BOX 5 – Pareto front

There Pareto front consist of all solutions that are not dominated by any other solution (Pareto optimal) i.e. solutions in which none of the objective functions can be improved in value without degrading some of the other objective values. In bi-objective optimization is also known as trade-off curve. In Fig. 5 is shown the objective space representing the two objective functions in the axes (f1 versus f2). The Ideal point (Z^I) is defined with the values of both objectives when minimised independently without taking into account the other objective. This point is just ideal since no feasible solution exists with these values. On the other hand, the Nadir point (Z^N) represents the most undesirable solution and it can be reachable or unreachable depending on the constraints. It is calculated with the value of one objective when the other one is minimised independently (the point Z_1^N is the value of the f1 when f2 is minimised; in the same way, the point Z_2^N is the value of the f2 when f1 is minimized).

The Pareto front also gives information on the objective trade-offs, that is, how one objective improvement is related to the deterioration of the second one while moving along the curve.

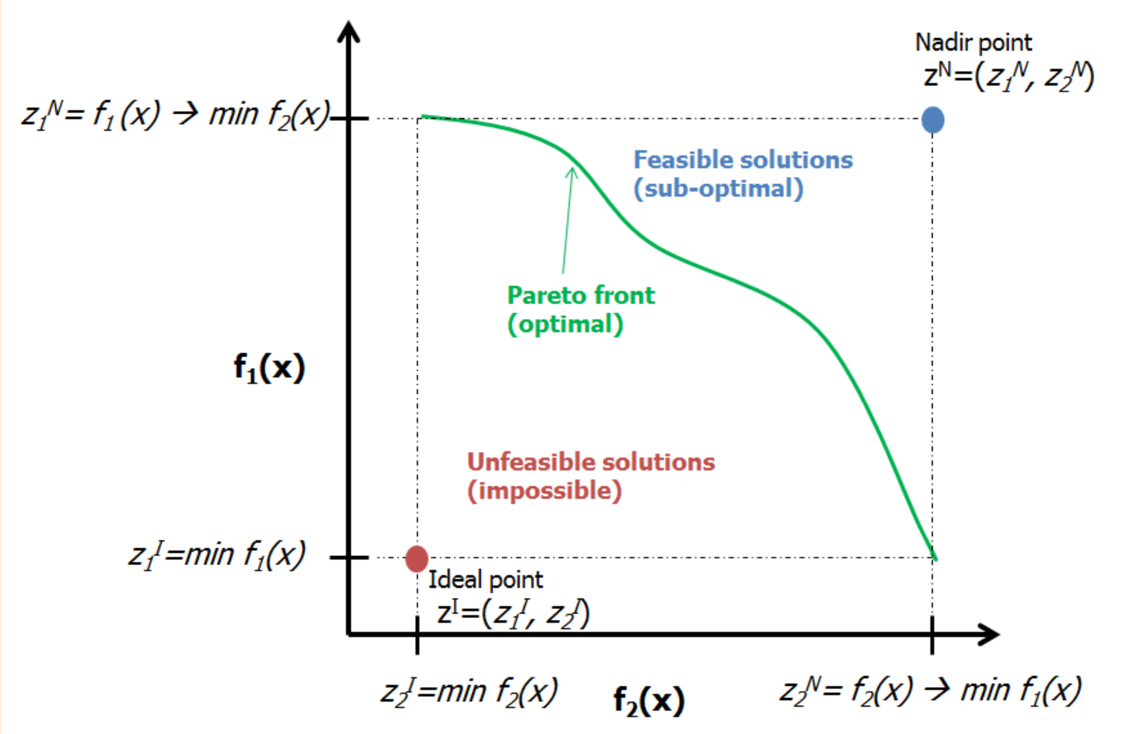


Figure 7: representation and explanation of the Pareto front

While all the Pareto optimal solutions are – by definition – equally good, in practice only one of them is selected by decision/policy makers. The final choice will in fact depend on a number of additional factors that are completely beyond the scope of this report and also depends on the preferences of the decision maker. This methodology is in fact intended to only provide relevant information that can help and support a decision maker in taking the final decision. There are different methods for solving multi-objective problems depending on the moment the decision maker enters the decision making process (see Box 6). The so-called "a posteriori" method is used in this methodology, according to which the representative set of (Pareto optimal) solutions is found and graphically presented to the decision maker to select one final solution.

BOX 6 – Methods for solving multi-objective problems

Based on the phase in which the decision maker (DM) is involved in the analysis of the multi-objective problem, there are two different types of methods (Rangaiah, 2009):

- Generating methods - that generates the pareto optimal solutions with no input from the DM. Among these solutions, the DM then selects the preferred/preferable one. These methods usually require more computational effort. They are divided into no-preference (do not require the relative priority of objectives, e.g. Global Criterion and Neutral Compromise Solution) and "a posteriori" methods (the DM express his/her preferences with all the solutions already generated, e.g. the scalarization approach).
- Preference-based methods - that generates the optimal solutions taking into account the preferences of the DM in the solving process. They are divided in "a priori" (the DM expresses his/her preferences before the solution process, e.g. setting goals or weighting the objectives) and "interactive methods" (consisting in iteration of phases of dialogue and calculation).

4. Towards a LC-based framework to evaluate and improve the sustainability of food waste management

4.1 Overview

This chapter aims at advancing existing understanding on how to assess the sustainability performance of food waste management. The ultimate goal is to provide relevant inputs towards establishing a coherent, lifecycle based sustainability assessment framework for food waste management from which:

- Most sustainable management options can be identified;
- Opportunities for further improving the sustainability of such options can be derived.

In the next sections, this task is approached in the following ways:

- First, an application of the waste hierarchy to the food waste context is proposed;
- Afterwards, a lifecycle based framework for simultaneous assessment of the environmental and economic performance is presented. Such framework could be expanded to include also additional dimensions, e.g. the evaluation of social/societal aspects. For sake of simplicity, additional dimensions are not integrated for the time being and the evaluation of social/societal aspects is addressed separately (see next bullet);
- At last, a simplified approach for the consideration of the social dimension of sustainability is provided.

4.2 From the waste hierarchy to the food waste hierarchy

As anticipated in Section 3.1, the so-called "waste hierarchy" defined by the Waste Framework Directive (WFD) 2008/98/EC (EC, 2008) provides the general, legally binding principle upon which waste management decisions shall be based. According to such principle, the priority order for waste management is: prevention, preparing waste for re-use, recycling, other recovery, and disposal. The WFD implicitly assumes that following the waste hierarchy will result – in most cases – in the waste being dealt with in the most environmentally sound way. However, the WFD opens to deviations from the waste hierarchy to ensure that the environmentally preferable option can be systematically identified. More specifically, such deviations are allowed only if LCT-based evidence shows that deviating from the hierarchy leads to a better overall environmental outcome (Figure 4).

The waste hierarchy is being extensively used by waste decision makers (at European, national and local levels) as a straightforward and cost-effective principle to inform environmentally sound decisions. While the waste hierarchy is applicable to virtually any type of waste, it is convenient to "refine" its scope so that it better fits the specific waste stream(s) included in any given waste management system to be analysed, e.g. food waste. For instance, Papargyropoulou et al. (2014) propose an interpretation of the waste hierarchy for food waste that is intended to address all dimensions of sustainability: environmental, economic, and social (Figure 8). Thus, following such hierarchy is expected to provide useful insight to identify sustainable options for food waste management.

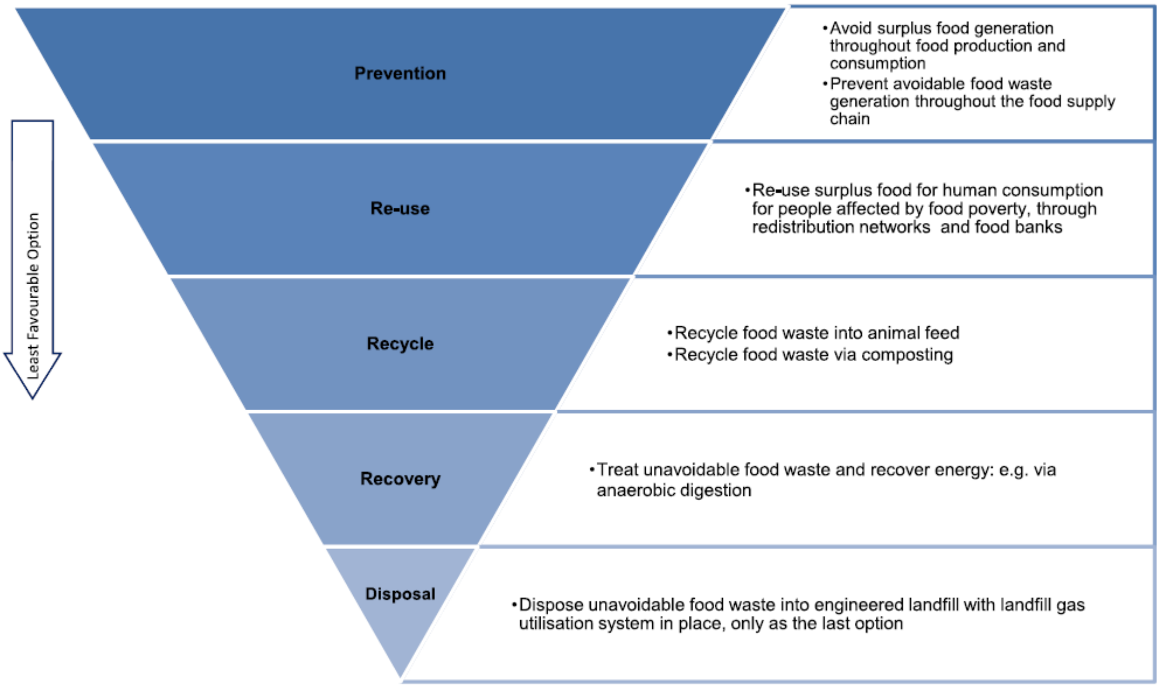


Figure 8: Interpretation of the waste hierarchy for food waste (from Papargyropoulou et al. (2014))

Prevention of food waste is the most preferable options according to such hierarchy. Preventing food waste entails both reducing generation of surplus food (i.e. avoid food production beyond human needs along the entire FSC) and prevent generation of avoidable food waste. While pursuing food waste prevention actions requires substantial re-consideration of the entire food waste production and consumption practices, extensive evidence exists that it can provide the highest benefits in all dimensions of sustainability.

4.3 Towards sustainability assessment: combined environmental-economic assessment

This section is intended to provide a step-by-step methodology (Figure 9) for combined evaluation of environmental and economic dimensions of food waste management systems. It could be expanded to include additional dimensions (e.g. social aspects could be the 3rd dimension), however, for the time being this is intentionally excluded. The selection of sustainable options makes also use of multi-objective optimisation and Pareto optimality.

The methodology here presented implicitly assumes that the first two priorities indicated by that waste hierarchy – prevention and reuse – remains the preferable options, i.e. are the most sustainable. However, it does not deal directly with them and instead focuses on the subsequent steps of the waste hierarchy, i.e. on the food waste flow that could neither be prevented nor reused, but needs to be managed/treated. In addition, it also accounts for any necessary waste collection and transport process.

Such methodology provides a structured decision support framework to help identify which alternatives for food waste management options are optimal in the sense that they minimise both objective functions, i.e. environmental and economic impacts. This can be useful to decision-makers and policy-makers in the field of waste management. However, the final decision will always depend on a number of aspects related to the decision context (e.g. policy and political context) that are intentionally not considered in this report and, in general, also depends on the preferences of the decision maker.

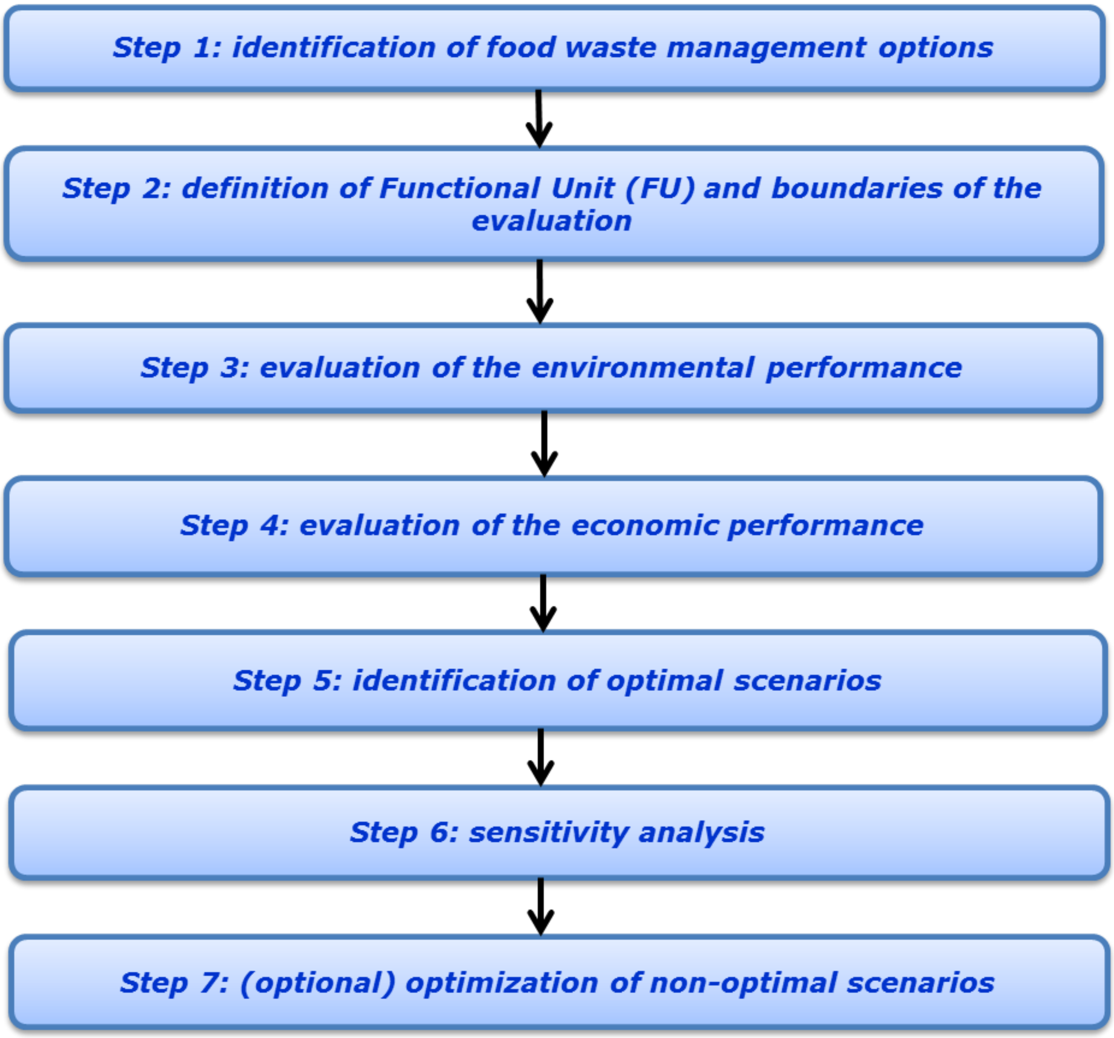


Figure 9: overview of the proposed methodological framework for sustainability assessment of food waste management

4.3.1 Step 1: identification of food waste management options

A thorough analysis of the waste management context under consideration should be conducted in order to identify all treatment options for food waste management that are going to be evaluated, e.g. composting (C), anaerobic digestion (AD), incineration (I) and landfilling (L). This includes both the options that are already in place and those that could be installed. Based on this, the modelling scenarios can be named after the core treatment technology (or combination of technologies[21]) for food waste considered in that scenario, e.g. AD_1, AD_2, …AD_n, C_1, C_2, …C_n, I_1, I_2, …I_n, L_1, L_2, …L_n, etc. Within the same type of management scenarios (e.g. the I = Incineration scenarios), the alternatives "1, 2, …n" stand for the different technological options (e.g. I_1 = incineration with electricity generation; I_2 = incineration with CHP generation; and so on).

[21] For instance, the scenario I (incineration), also landfilling may have to be included for the final disposal of the material outputs from the incineration process (e.g. fly ash and bottom ash). Or, in the scenario AD (anaerobic digestion), also composting may have to be included in order to improve the quality of the digestate.

4.3.2 Step 2: definition of Functional Unit (FU) and boundaries of the evaluation

Based on the identified options (step 1) and on the decision-context (i.e. the specific way the results of the assessment are intended to be used to support decision-making) an appropriate functional unit (FU) should be defined to described quantitatively and qualitatively the exact functions and services provided by the scenarios considered. Definition of the FU also includes specification of the composition of the waste input (e.g. chemical composition). Based on the selected FU, the system boundaries can be defined and should include all relevant foreground and background processes that are needed to provide the functions and services included in the FU.

The system boundary of each scenario, in addition to the core management technology(ies), should include the appropriate processes representing waste collection and transport. In general, these are not the same for all scenarios, and in some cases they are directly associated with specific treatment technology(ies) being considered (e.g. collection schemes for composting are likely to be different from collection scheme for incineration). Additionally, in this stage all the functions/outputs provided by the analysed systems should be identified and a choice should be made on how to account for systems that provide more than one function/output. This is further explained in Box 7.

BOX 7 – Definition of system boundaries

It should be noted that although the FU is defined in an unambiguous manner (e.g. based on management of 1 tonne of food waste of specified composition), each management scenarios may provide a number of outputs that do not appear in all considered scenarios (e.g. compost, electricity, heat). In order to make it possible from a LC point of view to compare the performance of scenarios providing different outputs, two alternative (but mathematically equivalent) techniques can be applied: "system expansion" or "substitution".

System expansion entails expanding the boundary of the system considered to include additional functions/product-outputs provided by the other systems compared. For instance, when comparing incineration with composting: the incineration scenario will have to include production of the same quantity/quality of compost produced in the compost scenario, and the compost scenario will have to account for the same amount of electricity (and/or heat) produced within the incineration scenario.

Substitution – which is the approach used in the modelling examples presented in this paper – consists in accounting for the benefits arising from the displacement of a certain function/product with an equivalent function/product generated within the system modelled. As a typical example, the electricity produced by waste incineration displaces the electricity that would have been produced elsewhere. The choice of the type of function/product that is displaced is crucial for the results of the LCA and depends – among other factors – on whether LCA is conducted following an attributional or a consequential modelling approach. For an exhaustive explanation of these two techniques and on attributional/consequential LCA, reference should be made to the ILCD Handbook – General Guide for LCA (EC, 2010d). The following simplified example, however, can also be considered to understand the equivalency of these two techniques.

Let us consider a certain "system A" that produces a certain quantity of electricity causing an environmental impact of D_A, and we want to compare it with another "system B" that produces a certain quantity of compost causing an environmental impact of D_B. Let us also assume that electricity and compost could otherwise be produced in conventional systems (e.g. fossil based) that cause an environmental impact equal to C_A and C_B, respectively.

Using system expansion, the total impact of system A would be the impacts of producing electricity (D_A) plus the impacts of the conventional system producing compost (C_B). For system B, the total impacts would be the impacts of producing compost (D_B) plus the impacts of the conventional system producing electricity (C_A). In the example, the difference in the total impacts is equal to X.

Using substitution, the total impacts of system A would be the direct impacts of producing electricity (D_A) minus the credited impacts for avoiding the production of the same quantity of electricity with the conventional system (C_A). For system B, the total impacts would be the direct impacts of producing compost (D_B) minus the credited impacts for avoiding the production of the same quantity of compost with the conventional system (C_B). In the example, the difference is again X (but this time in the negative axis – emissions avoided).

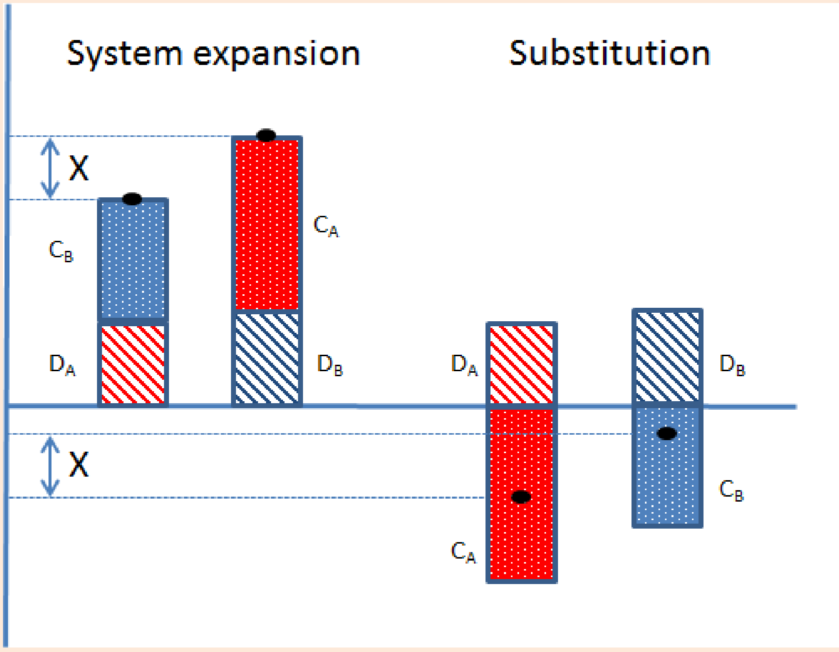


Figure 10: visual explanation of the equivalence between substitution and system expansion

4.3.3 Step 3: evaluation of the environmental performance

Based on the identified waste management options (step 1), the chosen FU and all the processes included in the system boundaries (step 2), the evaluation of the environmental performance should be conducted using LCA modelling, e.g. via an LCA-based software.

While this report does not aim at providing detailed guidance on how to conduct an LCA of waste management systems, it is important to notice that the level of detail/accuracy of such LCA heavily depends on what basis such LCA is performed in practice, e.g. what LCA software is used. Ideally, such software will allow for a high degree of freedom and flexibility to the user, but will also include default datasets and processes to be used whenever the LCA practitioner does not have specific data available. This aspect is particularly important in view of the fact that conducting an LCA entails – ideally – gathering enormous number of data to e.g. populate the Life Cycle Inventory (LCI) with all necessary inputs/outputs flows covering a high number of foreground and background processes.

Based on the results from the LCA (e.g. impact scores for each of the considered impact categories), one or more variable(s) should be chosen as representative of the environmental performance of the waste management scenarios identified in "step 1".

It should be noted that the choice of such variable(s) has a key importance in the sense that it can directly influence the identification of the most sustainable management option(s). Possible choices include:

a. (RECOMMENDED CHOICE) To consider as many variables as the number of impact categories considered in the LCA. This means that in the comparison with the indicator that will be chosen to represent the economic performance (step 4), each environmental impact category is considered individually (i.e. in isolation from the others).
b. To consider the total environmental impact (intended as the sum of the individual impact scores after normalization/weighting[22]).
c. To consider the sum of the normalized/weighted impact scores of a sub-group of impact categories (potentially down to one individual impact category, e.g. climate change).

Choosing (a) is the most general and impartial approach, as all impact categories included in the LCA are considered (i.e. no exclusions are made) and will be subsequently compared with the indicator representing the economic performance.

Choosing (b), i.e. calculating a single impact scores based on the impacts scores estimated for all impact categories, implies that the "importance/relevance" of each impact category is proportional to the weighting factor used to perform the weighting phase of the LCA[23]. If all impacts categories were assigned a weighting factor of "1" (one), then all impact categories would be considered equally important to support decision making.

Choosing (c) is equivalent to choosing (b), with the difference that the weighting factor assigned to one or more impact categories is set to "0" (zero). This is thus equivalent to excluding one or more impact categories from the evaluation, e.g. because they are considered not relevant for decision making.

BOX 8 – Accounting for collection and transport in LCA

As food waste is a fraction of overall Municipal Solid Waste (MSW), typically accounting for 30-40% of it, the environmental impacts related to the primary collection stage of 1 tonne of food waste, i.e. the routes covering all households, should be carefully assessed.

In literature, many studies reporting ranges of km/t for waste collection can be found related do different waste collection schemes. Actually, the most optimized MSW management schemes including food waste collection are not "additional" schemes (e.g. adding a new collection route for this fraction, leaving the rest of the scheme unchanged). These schemes strongly rely on the reduction of the collection frequency of residual (i.e. mixed) waste, as shown in Table 3 referring to the changes in a municipality with kerbside (door to door) collection scheme when introducing food waste separate collection.

[22] Normalisation and weighting of LCA results are two optional phases in LCA (ISO, 2006). However, when summing impact scores among different impact categories towards estimating a single, overall score representing the environmental performance, then normalization and weighting become mandatory.
[23] For an in-depth understanding to the meaning of weighting in LCA, reference can be made to ISO 14044 (ISO, 2006) and the ILCD Handbook – General Guide for LCA (EC, 2010c).

Table 3: Typical examples of how food waste collection integrates with residual waste collection by reducing its frequencies (examples of Kerbside schemes in IT, UK, and some areas of ES)

Examples	**Kerbside, without food waste collection**	**Kerbside, with food waste collection**
Residual waste collection frequency	2/week	1/week or even every two weeks with PAYT schemes
Food waste collection frequency	--	2/week
Residual waste collection vehicles	Large trucks	Small vehicles; often the same truck (with two compartments) is used for both food waste and residual waste collection
Food waste collection vehicles	--	

An extended analysis, broadening the scope to the other waste fractions, can lead to more appropriate results; however, when the functional unit is 1 tonne of food waste, the following points should be addressed:

- How to select the appropriate value of km/t when integrated collection schemes are in place, for example when food waste is collected in two-compartment vehicles together with mixed waste;
- Carefully select the appropriate vehicles for food waste collection.

A simplified assumption, for scenarios in which food waste is not separately collected (i.e. is sent to landfill or incineration), is to consider the same vehicles used for mixed waste collection in less advanced schemes such as large diesel trucks. A more refined analysis should be based on real mileage data taken from a case study where the optimized collection scheme is in place, also considering the distance covered by the vehicles when not collecting waste, such as moving from the city to the transfer points or to the headquarters. These are typically those that can be optimized when shifting to reduced collection frequencies.

BOX 9 – Allocation of environmental impacts to the flow of food waste

While food waste can be treated in isolation in technologies such as composting and anaerobic digestion, in other treatment options where it is not source separated, food waste is typically treated together with other waste streams. This is for instance the case of incineration (where food waste is co-incinerated with other waste types, e.g as part of MSW) and landfilling (where the amount biodegradable waste landfilled should anyway stay below the limit set by Landfill Directive 1999/31/EC). In all these cases, the allocation of environmental impacts to the chosen FU (e.g. 1 tonne of food waste) is not straightforward and may require making assumptions and/or simplifications. A thorough analysis of available allocation options in LCA is provided by Allacker et al. (2014).

If, for instance, incineration is considered than the 1 tonne of food waste considered (as from the chosen FU) will be incinerated together with a certain mass of different waste types (e.g. X tonnes). Such share of food waste in the overall waste incinerated varies depending on the food waste recycling rate in that area at a certain time. Waste incinerators are, however, not designed to handle waste with very low calorific value, so a first assumption can be to set the maximum share of food waste in the overall waste incinerated (e.g. 50%), and an average share where food waste separate collection is in place with medium results (e.g. 25%).

As a second assumption, one could thus consider a mass-based allocation of impacts and avoided impacts, i.e. that co-incineration of such 1 tonne of food waste is to be calculated assessing the impacts of 4 tonnes of overall incinerated waste and then assigning 25% of the environmental impacts (from net emissions to the environment) and 25% of the environmental benefits (from displacement of energy with the energy produced from waste incineration) arising from the incineration of the overall waste. While this assumption may be accepted as a first, rough simplification, attention should be paid to the following (non-exhaustive) list of aspects that undermine its validity and add to the complexity of the modelling:

- Emissions from incinerators are typically classified into process-specific and input-specific. Process-specific emissions (e.g. CO_2) are those that mostly depends on the characteristics and the efficiency of the combustion process and not (or marginally) on the composition of the waste incinerated. For this type of emissions, thus, the mass-based allocation of impacts mentioned above may be a reasonable assumption. Input-specific emissions (e.g. heavy metals), instead, depends on the quality/composition of the waste incinerated. A mass-based allocation of this type of emissions is thus in principle not meaningful. In this case, allocation of emissions should rather be based on knowledge of the transfer of chemicals/pollutants from the input waste to the output emissions (e.g. how the content of mercury in the waste incinerated distributes among the outputs of bottom ash, fly ash, gaseous emissions, etc.)
- The amount of energy that can be recovered from waste incineration is proportional to the (average) calorific value of the waste input. In the LCA modelling, as this recovered energy is assumed to displace for energy produced elsewhere, it is accounted for as an environmental benefit. Thus, the criteria for allocating environmental benefits arising from energy recovery should be based on the calorific value (e.g. share of the overall calorific value that is brought by food waste). It should also be noted that the magnitude of the environmental benefits arising from displacement of each unit of energy (e.g. 1kWh) depends on the type of energy that is assumed to be displaced. The choice of such energy depends on many factors, including whether the LCA is conducted using an attributional or a consequential modelling approach[24].

The considerations presented in this box (taking waste incineration as an example) indicate the potential complexity of allocation of emissions and impacts in case of co-treatment of food waste. While addressing such complexity may even require to use different allocation criteria for different emissions, it also highlight the fact that in many case assumptions will have to be taken to simplify the system being evaluated and reduce modelling efforts and costs.

4.3.4 Step 4: evaluation of the economic performance

This step focuses on the estimation of the costs necessary to provide the functions/services defined in the functional unit within the identified waste management scenarios. Different indicators can be used to express the economic performance of each scenario. These mainly differ on which parts of the food waste management chain are included and on whom is supposed to pay such costs. For instance:

- The **cost for the authority** (e.g. a municipality) **for waste treatment**. Adopting the approach from other studies (e.g. EC, 2001; EC, 2004), the

[24] For an exhaustive explanation of these two LCA modelling approaches, reference should be made to the ILCD Handbook – General Guide for LCA (EC, 2010c)

treatment costs can be represented using as a proxy the **gate-fee**, i.e. the cost that the authority pays for delivering waste to a specific treatment plant. Gate-fees account for both capital costs (CapEx) and operational costs (OpEx), which thus do not need to be directly estimated. It should be noted, however, that a given gate-fee is not necessarily equal to the true welfare economic costs nor is it equal to the financial costs. For example, the gate fee could be lower than the financial costs due to temporary market condition (e.g. overcapacity for incineration), length of contracts, or direct or indirect subsidies.

- The **cost for society for waste treatment**. This is obtained by removing the effect of any existing subsidies or taxes from the previously identified gate-fees. This allows to estimate a sort of "raw gate-fee", which is closer to the sum of capital expenditures and operational costs. In other words, in case of subsidies, the society is paying them so they have to be added to the cost for the authority (gate fee).
- The **cost for society for waste management.** This is estimated by adding to the previously estimated "cost for society for waste treatment" the costs of waste collection and transport.

After Steps 1 to 4, it is thus possible to associate two "coordinates" to each option. These coordinates are: EI (environmental impact, e.g. the total environmental impact) and C (cost). Thus a chart EI/C can be plotted including all scenarios considered.

BOX 10 – Influence of size capacity and efficiency of the treatment plant on the costs

The gate fees assumed as a reference in this study have been selected considering average plant size and efficiency. However, it must be highlighted that there is indeed a large variability in gate fees - which reflects CapEx and OpEx - primarily due to the plant size. This is particularly true for incinerators where the gate fee for a small plant, (e.g. < 200,000 t/year) can be significantly higher than that of larger plants.

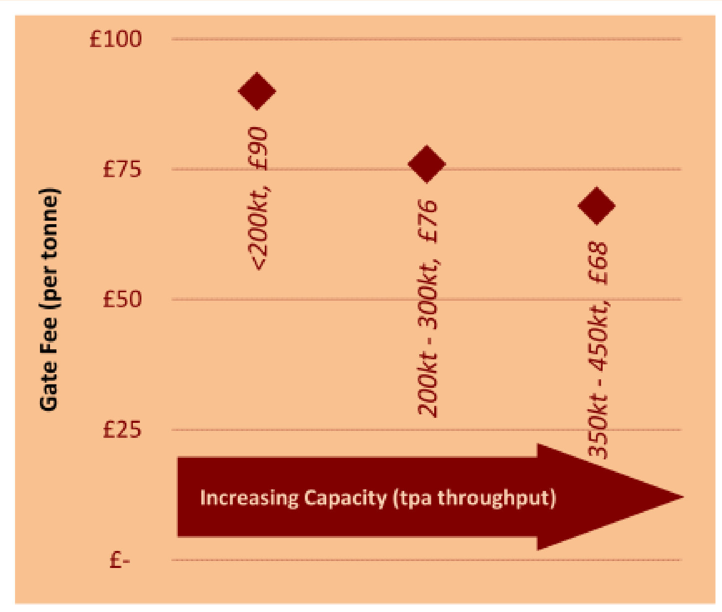


Figure 11:Incineration gate fees in UK according to the plant size (WRAP, 2012)

It is also true that thinking in terms of intrinsic optimization (i.e. pushing to the bottom - left of the Pareto chart), those plants that are suboptimal in terms of applied gate fee are sooner or later forced either to exit the market or to undergo a renovation in order to compete with all other existing similar plants which due to better management and technology can treat food waste with a lower gate fee.

A particular remark should be made for very small scale decentralized treatment options; those technologies (e.g. composting plants treating < 1,000 t/year) are actually promising and can compete with larger scale ones only if they are part of a network in which some synergies exist, such as in the Austrian model integrating food waste treatment in existing farms.

Besides that, plant efficiency affects overall costs per tonne basically in all technologies, especially those that besides the gate fee rely much on external revenues as part of the annual budget such as those using anaerobic digestion and incineration with energy recovery.

4.3.5 Step 5: identification of optimal scenarios

Based on the results of the environmental and economic assessment (steps 3 and 4) and on the indicator(s) chosen to represent the environmental and economic performance, the environmental impacts versus economic impacts can be plotted. This allows to visualize the objective space and plot the so-called "Pareto-front" and thus identify the optimal scenarios, i.e. those that minimise at the same time both objective functions.

If targets/limits (i.e. constraints) on any of the objective functions exist or if the user wants to define such constraints (e.g. the maximum accepted environmental impact arising from binding emission limits; and a maximum acceptable cost), these can be included graphically in the chart. If any optimal scenario falls outside the region delimited by such constraints it will have to be discarded.

4.3.6 Step 6: sensitivity analysis

In decision making processes, the consideration of uncertainty is crucial since apparent differences in impacts may be misleading if the uncertainty is large enough to overwhelm any relative differences between the compared alternatives (Baker and Lepech, 2007). To base decision making upon a more solid ground, thus, the influence of possible uncertainties on the results of the evaluation must be systematically evaluated. Such uncertainties may arise due to a variety of aspects such as data variability, erroneous measurements, wrong estimations, unrepresentative or missing data and modelling assumptions (Clavreul et al., 2012).

There are several procedures to account for uncertainty and one of the simplest ones is the sensitivity analysis. It consists of systematically varying input parameters in order to determine how sensitive the outputs are to each input. Further analysis can be done in order to understand how uncertain our results are by means of uncertainty propagation techniques.

4.3.7 Step 7: (optional) optimization of non-optimal scenarios

It is possible to expand the proposed methodological framework to make it capable of analysing the identified non-optimal solutions towards finding the key parameters that may improve their sustainability performance. For instance, the so-called eco-efficiency analysis can be performed via Data Envelopment Analysis (DEA) in order to set improvement targets for the non-optimal management options. Ideally, if technically/economically viable, achieving targets would make the identified non-optimal solutions become optimal. While this step is purposely not further expanded in this report, Cristobal et al. (2016) provide analysis and guidance on how to use eco-efficiency analyses towards optimising food waste management. More information is available in other published literature (Dyckhoff and Allen, 2001; Zhou et al., 2008).

4.4 Elements of social assessment: a simplified framework

In literature, a framework and a set of indicators, in order to rate and compare different food waste treatment options by means of their social impacts, are still missing. To fill such gaps, this section aims at:

1. Developing a set of indicators covering several scales: from collection, to treatment, taking into account a wider socio-economic context.
2. Assessing potential social impacts associated to the food waste treatment under different treatments options.

The theoretical framework is the result of a joint effort aiming at analysing current literature and adapting existing S-LCA frameworks.

4.4.1 Selection of relevant indicators for the social impacts of food waste

The criteria used for the choice of indicators revolves around the inherent dissimilarity between them, the stakeholder specific relevance and the usefulness in comparing social impacts in the different waste treatment options. Focus is placed on indicators and drivers that have an impact on an ex-post basis to evaluate social impacts of current treatment options already in place. This means that emphasis is placed on treatment facilities already built. Whereas, if considering an ex-ante basis, it will mean that those social aspects can influence the choice of building one treatment option rather than another (landfill over incineration for example). As this is outside the boundary of our scope, we will focus on ex-post.

For the selection of the indicators, we took as a starting point the stakeholders and indicators list from the UNEP Guidelines on Social LCA (UNEP/SETAC, 2009) and extracted and adapted the most relevant for this study. Furthermore, some more specific indicators on social impacts of waste treatment options were found in the literature (e.g. Rybaczewska-Blazejowska, 2013). A collection of the indicators is summarized in Table 4 below.

Table 4: List of chosen indicators for social impacts in waste treatment options

Stakeholder	**Subcategory**	**Indicators**	**Example of indicator to be collected**
Workers (collection + plant)	*Health and Safety*	Accident rate at workplace	Number of accidents / tonnes of waste collected or treated
		Occupational risks	
		Labour intensity	Number of workers / tonnes of waste collected or treated
Local community	*Safe and healthy living conditions*	Contribution of the proposed scenario to environmental load	LCA indicators

		Odour	Odour concentration at residential areas, as modeled with air dispersion official models
		Noise	Modeled noise around the plant
		Traffic	Increase in traffic with respect to a baseline
		Visual and landscape impact	Qualitative-quantitative impact matrix
	Community engagement	Time and space required by waste management at home	Customer satisfaction survey: degree of satisfaction of citizens[25]
	Local employment	% of Work force hired locally	%
Consumers	*Feedback mechanism*	Number of customer satisfaction surveys carried out per year	Number of queries and complains received at call center/other follow up instrument
Society	*Technology development*	Innovation in food waste treatment technology over time	Number of plants complying with BREF (Best Available Technologies guidance)
	Commitments to sustainability issues	Number of green procurement:	Purchase of compost, heat etc. Market share of compost vs. Mineral fertilizers
	Contribution to economic development	Creation of green jobs per € invested	Number of jobs/tonnes of food waste managed, or/€ invested
Value chain actors not including consumers	*Promoting social responsibility*	Number of stakeholders engaged in the promotion of the model	

The final selection of indicators has been included in the framework in Table 6. These indicators should assist in understanding the degree of social impacts at different levels that the various food waste treatment options might have. However, attention should be placed on the fact that, ideally, a set of indicators should reflect the local conditions where the food waste is produced and treated. Social impacts of the waste treatment stage will depend on the local or regional choice of waste management companies, the

[25] E.g. http://www.ecodallecitta.it/docs/news/EDC_dnws3540.pdf

technologies used and the way in which these companies interact with their employees, the local community and other relevant stakeholders (Dreyer et al., 2006)

4.4.2 Framework

Social aspects are present throughout the various stages of the food waste production chain. A first important step is the communication for prevention as well as the redistribution of unsold food which may have substantial social implications in terms of positive/negative impacts. However, as mentioned before, the study will have as focus the subsequent stage, from the moment the food is disposed of as waste by consumers or retailers and the available options for treating it.

The prevention stage is of utmost importance in order to diminish the amount of food that consequently becomes waste. We acknowledge this issue in our framework through the Prevention box in Table 6, which summarizes the role of communication towards changing behaviors for food waste reduction. Food waste prevention can be tackled by sensitizing consumers to buy less (only what is really needed) and to efficiently store it and consume it. Also, retailers can support the prevention of food waste by collaborating with food banks and charities in order to allow their edible unsold food to reach less privileged people, instead of landfills. Furthermore, they can contribute their unsold food to farms in order for it to become feed. Another way is to entice consumers to buy near-expiry date food by discounting the final price. These are just some of the behaviours that can reduce food waste and a strong communication campaign can play a strong role. Similarly, at the end of life stage, communication plays an important role concerning consumers as typically a lack of awareness about the implications of the final recycling is one of the strongest drivers that paves the way to a failure of the waste treatment system. This can be tackled by increasing citizen's awareness through campaigns about the final destination of food waste, recycling, environmental impacts and benefits. The implementation of this communication can be monitored by indicators such as the number of municipal/wide area reports disclosing information about the final destination of all waste fractions

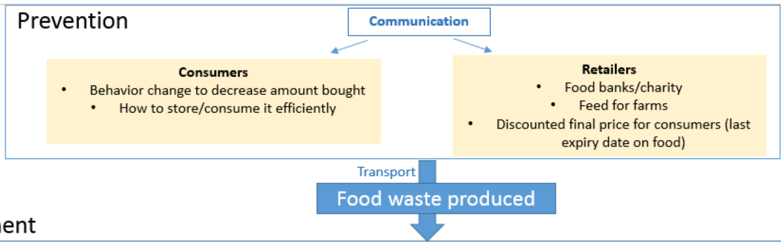
Once the prevention stage has done its course and food is discarded, there are various waste treatment options, some of which are highlighted in the *Waste treatment box* in Table 6: Landfill, Incineration, Composting and Anaerobic Digestion. The social impacts of these options can be analyzed from different levels which can be summarized in 1) the collection/transport of waste; 2) the activities performed in the waste treatment plant; 3) the treatment plant's surrounding; and 4) the wider external context. Each level is encompassed within the higher level, to show the interrelations between the different levels. The framework also includes relevant indicators for the assessment of social impacts at different levels to compare various food waste treatment options. Some scores can be applied to each qualitative or semi-qualitative indicator in a rating of A-F as described in Table 5 (Ciroth and Franze, 2011). Table 5 shows fictional scores purely to display how the table would look like.

Table 5: Score assessment

Performance assessment	Colour	Impact assessment
Very good performance	A	Positive effect
Good performance	B	Lightly positive effect
Satisfactory performance	C	Indifferent effect
Inadequate performance	D	Lightly negative effect
Poor performance	E	Negative effect
Very poor performance	F	Very negative effect

Regarding the **collection** phase, there exist different types of collection systems, according to local contexts and conditions. As explained in the next sections, the two most common schemes are the so called kerbside scheme (door to door), or the bring scheme with large road containers (WRAP, 2007). As the kerbside scheme allows for higher quantity and quality of food waste captured, following the approach used for the environmental assessment, this collection method has been applied to the scenarios "composting" and "anaerobic digestion", whereas traditional road container scheme with large trucks and mechanical loading has been considered for the scenarios "landfilling" and "incineration" which actually refer to an option that is either only theoretical (food waste should not be sent to incineration because of its low calorific value) or prohibited by the EU legislation (landfilling of biodegradable waste). These two collection methods involve different work risks and citizen's involvement in terms of Social LCA.

Table 6: Framework to study social impacts in various food waste treatment options with fictional indicator scores



Waste treatment

	Collection (Worker)	Plant (Worker)		Plant's surrounding (Community)							External Context (Consumers, Society)				
Indicators	Accident rate at workplace	Occupational risks	Labour intensity	Contribution of the proposed scenario to environmental load	Odor	Noise	Traffic	Visual and landscape impact	Time/space for home waste	% work force hired locally	# customer surveys per year	Tech innovation in treatment over time	# of green procurement	Green jobs per € invested	# of stakeholders engaged in social responsibility
Landfill	2 per ton	C	B	F	F	C	E	D	C	75%	0	E	4	D	7
Incineration	4 per ton	B	B	E	A	C	D	C	B	90%	1	A	15	C	5
Composting	1 per ton	E	D	B	C	B	C	E	B	80%	2	B	20	E	9
Anaerobic Digestion	3 per ton	D	C	A	B	C	E	D	B	80%	1	F	18	D	8

The above framework is a proposal in order to try and capture social impacts at different levels for each food waste treatment option. The idea is to ultimately be able to fill Table 6 with quantitative, qualitative and semi-qualitative data in order to compare the options and their social impacts. Data will need to be gathered and analysed locally, due to the differences in local conditions.

5. Case study

The following case study should be intended as an example of simplified application of the proposed methodological framework. Therefore, its results (i.e. the identification of the most sustainable options for food waste management) have a limited validity and should not be used to support waste management decisions in specific contexts, i.e. they should not be generalized nor transferred. The modelling examples, in fact, are fictitious scenarios symbolising an average, typical European situation in which support is needed to help decide how to manage food waste. Most data used refer to the time period 2010 to 2015.

The FU chosen in the modelling application is "management of 1 tonne of food waste (from collection to final disposal), including any impurity that comes along these fractions. Table 7 and Table 8 shows some key components of the waste composition used in the modelling.

Table 7: Waste composition (TS = total solids)

Fraction	Water (%)	TS (%)	C bio (%TS)	C fossil (%TS)	K (%TS)	N (%TS)	P (%TS)
Animal food waste	57.14	42.9	55.4	1.13	0.53	7	1
Vegetable food waste	76.99	23	47.5	0.24	1.27	1.9	0.23

Table 8 (continuation): Waste composition (TS = total solids)

Fraction	Cd (%TS)	Cr (%TS)	Cu (%TS)	Hg (%TS)	Pb (%TS)	Zn (%TS)
Animal food waste	$1.1x10^{-5}$	$1.2x10^{-4}$	$6x10^{-4}$	$2x10^{-6}$	$7x10^{-6}$	$4.9x10^{-3}$
Vegetable food waste	$9.5x10^{-6}$	$4.5x10^{-4}$	$1.2x10^{-3}$	$2x10^{-6}$	$1x10^{-4}$	$2.5x10^{-3}$

With respect to the modelling application herein, six scenarios are considered and presented in Table 9 under the assumptions that all treatment plants are already established and fully operative. Table 9 also presents key set up elements for each treatment technology as well as important modelling assumptions.

Table 9: Modelling scenarios considered, key technological choices and assumptions

Acronym	Full name	Short description and key modelling assumptions
I	Incineration	• Incineration with energy recovery and combined heat and power (CHP) generation. • Incineration technology: grate furnace with wet flue gas cleaning • Energy recovery efficiency of heat and electricity are set to 14% and 46% of the lower heating value (LHV) of the input waste, respectively. • The heat produced is assumed to be 100% used for

		local district heating, displacing heat from the average technology in Europe (year 2012). • The electricity produced is assumed to be 100% sent to the grid, displacing electricity from the average electricity mix in Europe (year 2010). • Bottom ash output is assumed to be sent to a mineral waste landfill. • Fly ash output is assumed to be stabilized and used as backfilling material, displacing gravel.
AD	Anaerobic digestion	• Technology: one stage, wet and thermophilic conditions. • Gas yield is assumed as 138 m^3 of biogas / ton of waste. • The methane content in the biogas is 63%. • The main output is biogas which is combusted in an engine to produce electricity with an efficiency of 25% of the LHV of the biogas. • The electricity produced is assumed to be 100% sent to the grid, displacing electricity from the average electricity mix in Europe (year 2010). • The digestate is used directly on land as a fertilizer and substitutes a chemical fertilizer with substitution ratio for N,P,K equal to 0.4, 1 and 1, respectively.
C	Composting	• Technology: enclosed windrow composting with ventilation system and off-gases directed to a biofilter for odour control. • The Volatile Solid, Carbon and Nitrogen degradation in the composting phase is equal to 74.56%, 74.56% and 65%, respectively. • Electricity consumption in the composting phase is 0.035 kWh / tonne wet waste. • Compost is used directly on land as a fertilizer and substitutes a chemical fertilizer with substitution ratio for N,P,K equal to 0.4, 1 and 1, respectively.
AD+C	Anaerobic digestion + Composting	• Same technologies and assumptions as in AD and C scenarios. • Digestate is composted for quality improvement and stabilization. • Electricity consumption in the composting phase is 0.02 kWh / tonne wet waste.
Le	Landfilling with electricity production	• Landfill with bottom liner, top soil cover, gas and leachate collection and treatment / utilization. • Decay rate (k rate) of Carbon for moderate moisture conditions equals 0.137 per year. • Management of landfill gas is structured in time periods and the collection rate is set to 35% during the first 5 years, 65% during the next 10 years and 75% during the following 40 years. After year 55, landfill operations stop. • Uncollected gas is partially oxidized in the soil top cover. Oxidation rates are assumed compound-specific and change over time. • A leaking rate from the gas collection pipes of 2% of the landfill gas produced is considered. • Collected landfill gas is combusted in an engine to produce electricity with an efficiency of 25% of the

		LHV of the landfill gas. • The electricity produced is assumed to be 100% sent to the grid, displacing electricity from the average electricity mix in Europe (year 2010). • Leachate generation is set to 300 mm/year. Leachate is collected and treated in a waste water treatment plant (WWTP), which is modelled using primary sedimentation and secondary treatment process utilizing a biological N removal process. • Sludges from the WWTP are used in AD and the biogas produced utilized for electricity consumption on site (3.48 kWh/kgCOD). Uncollected leachate is assumed to reach surface water bodies.
Lf	Landfilling with flares	• Same technology and assumptions as in Le scenario. • Collected landfill gas is burnt in flares during the first 55 years (i.e. till landfill operation stops). Oxidation rates (due to combustion in flares) are assumed compound-specific and change over time.

The processes included in the evaluation for each scenario are detailed in Figure 12 to Figure 17, which include all the processes mentioned in Table 9, all necessary input of electricity, fuels, materials, as well as waste transport and collection steps.

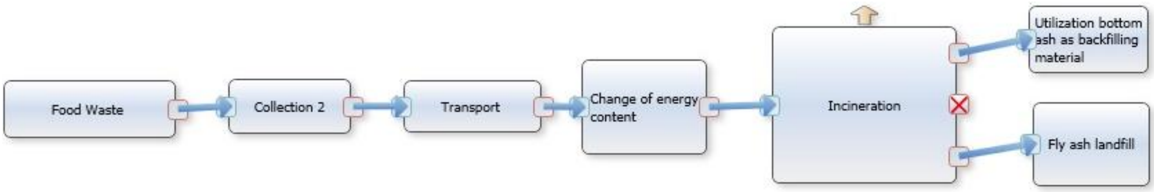


Figure 12: Key processes included in scenario 1 – Incineration

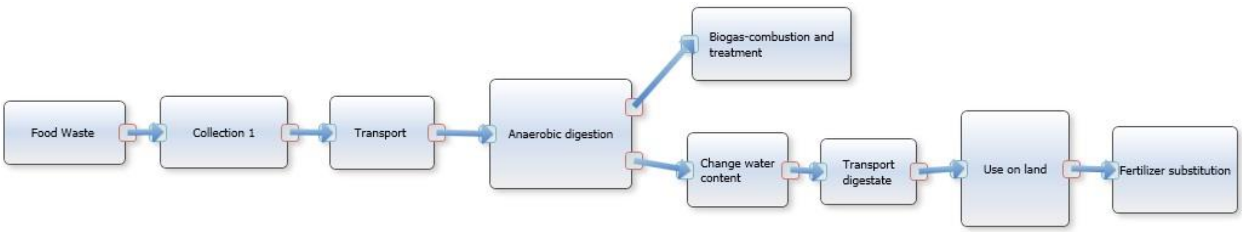


Figure 13: Key processes included in scenario 2 – Anaerobic digestion

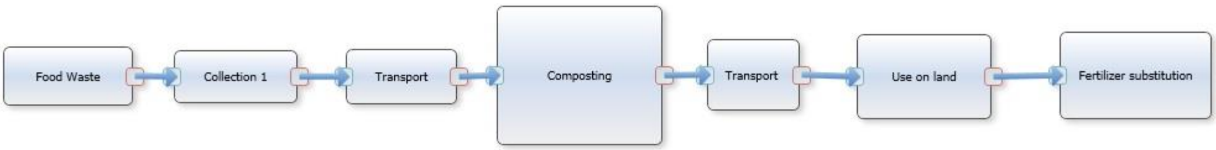


Figure 14: Key processes included in scenario 3 –Composting

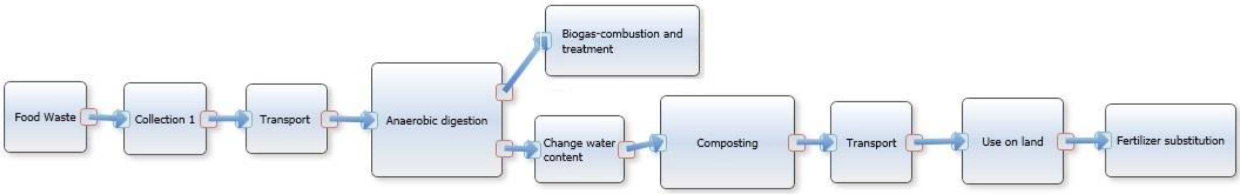


Figure 15: Key processes included in scenario 4 – Anaerobic digestion + Composting

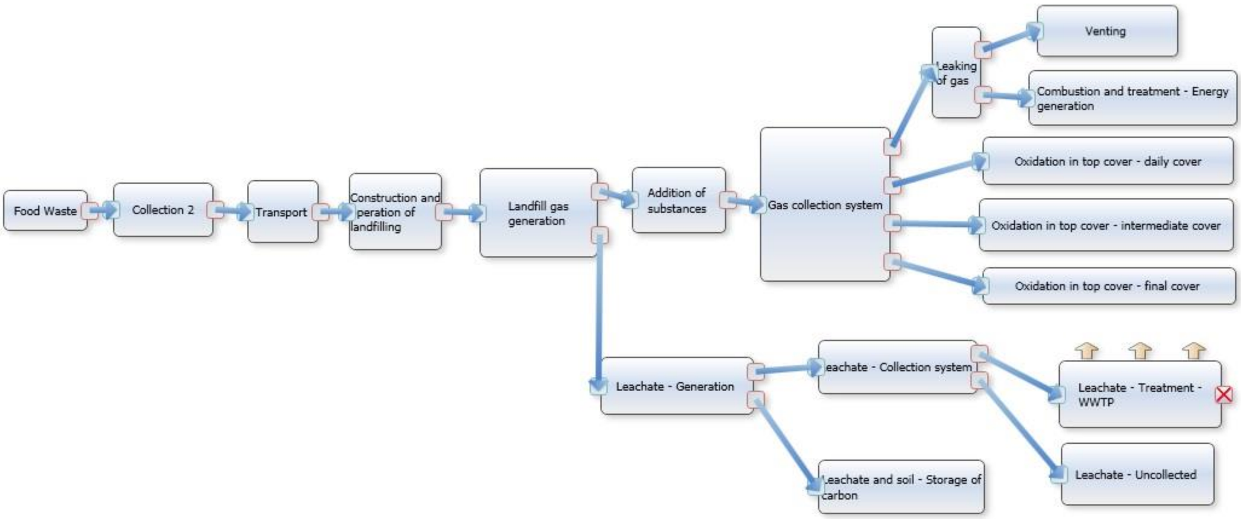


Figure 16: Key processes included in scenario 5 – Landfill with electricity production

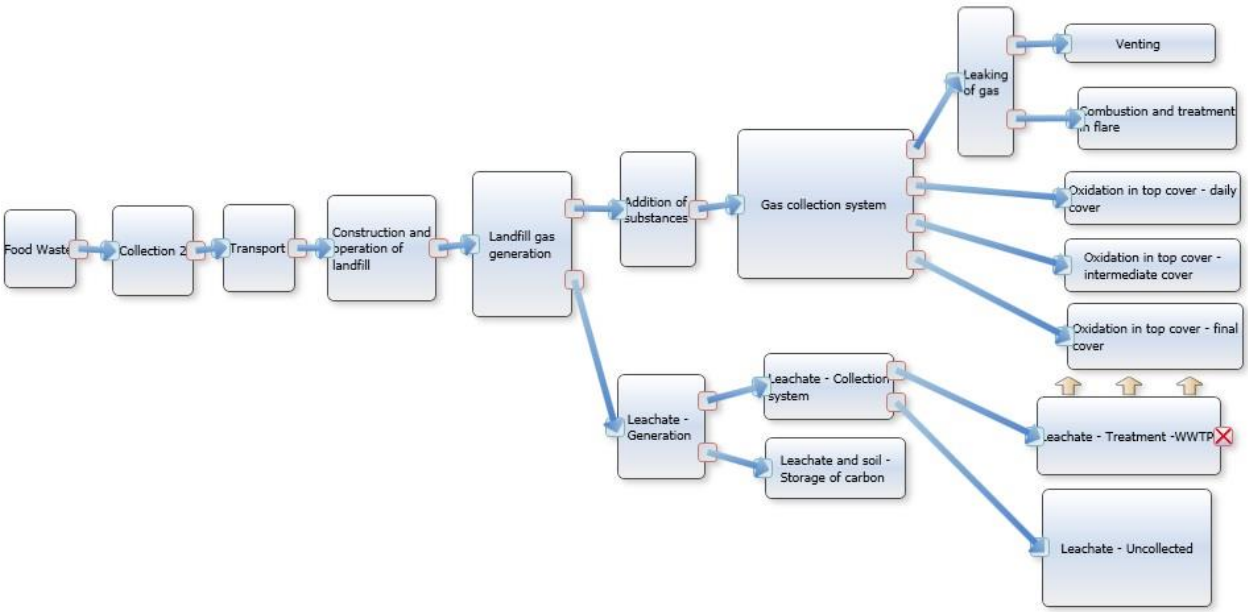


Figure 17: Key processes included in scenario 6 – Landfill with flare

Concerning transport, for all scenarios it was assumed that the waste management plant is 50 Km far from the collection point and that vehicles must also come back to such point. The collection step is considered as technology-specific (Table 10), e.g. the type of collection assumed for the treatment technologies that can treat food waste only is, in general, different from that assumed for the technologies that handle municipal solid waste. This is reflected in the costs for collection and transport assumed in the modelling (Table 12).

Table 10: key data for the different collection schemes

Collection Scheme	Scenarios	Type of truck	Emission standard	Fuel consumption rate (l diesel / tonne of waste)
Only food waste	AD, C, AD+C	10 t	Euro3	6
Municipal solid waste	I, Le, Lf	25 t	Euro3	7.5

The Life Cycle Impact Assessment (LCIA) calculations of the modelling example, conducted with EASETECH, are based on the EC Product Environmental Footprint (PEF) method (EC, 2013). Environmental impacts are estimated for 12 out of the 15 PEF recommended impact categories, as EASETECH does not include two impact categories (i.e. land use and water resource depletion) and another one (i.e. resource depletion – mineral, fossil and renewable) is not calculated as recommended by the PEF method. Results are shown in Table 11.

As emissions from landfills are not "virtually instantaneous" but spread over long time periods (up to thousands of years for some compounds, e.g. leaching of metals), a time horizon of 100 years was set in EASETECH. This means that all emissions from the moment when 1 tonne of food waste is landfilled (time "zero") till 100 years into the future are accounted for. Another aspect that needs to be addressed when landfilling is considered in LCA, it is the way storage of carbon is accounted for, which is a highly debated topic in LCAs of waste management systems (e.g. Christensen et al., 2009; Manfredi et al., 2009). A certain amount of carbon will in fact remain stored in the landfilled waste after 100 years, which is mostly of biogenic origin given that landfilling of food waste is here considered. In line with the recommendations provided by Christensen et al., (2009), with respect to "climate change" storage of biogenic C was accounted for as a net saving, while storage of fossil carbon was accounted for as neutral.

*Table 11: Environmental impacts from management of 1 tonne of food waste in the considered management scenarios (*Comparative Toxic Unit for humans; **Comparative Toxic Unit for ecosystems)*

Impact category	AD	C	AD+C	I	Le	Lf
Climate Change (CC) kg CO_2 eq.	-7.6×10^{-2}	1.03×10^{2}	0.61	-4.07×10^{2}	8.80×10^{2}	9.94×10^{2}
Ozone depletion (OD) CFC-11 eq.	-7.47×10^{-7}	1.64×10^{-7}	-6.93×10^{-7}	-2.55×10^{-6}	1.80×10^{-4}	1.72×10^{-4}
Human toxicity, cancer effects (HH,ce) CTUh*	-1.70×10^{-7}	1.86×10^{-8}	-1.43×10^{-7}	-5.61×10^{-7}	-8.47×10^{-8}	6.55×10^{-8}
Human toxicity, non-cancer effects (HH,nce) CTUh*	4.30×10^{-4}	4.61×10^{-4}	4.58×10^{-4}	-6.95×10^{-6}	1.69×10^{-6}	3.68×10^{-6}

Ionising radiations, human health effects (IR,hh) kg U235 eq. (to air)	-0.34	0.15	-0.31	-0.83	-0.32	8.17×10^{-2}
Photochemical ozone formation (POF) kg NMVOC eq.	0.76	0.21	0.88	0.47	1.22	0.80
Freshwater eutrophication (FE) kg P eq.	-9.06×10^{-2}	-0.11	-0.11	-5.45×10^{-3}	-1.17×10^{-3}	2.75×10^{-4}
Marine eutrophication (ME) kg N eq.	7.07	1.89	2.15	0.21	0.41	0.25
Ecotoxicity for aquatic freshwater (Fecotox) CTUe**	2.50×10^{2}	2.71×10^{2}	2.67×10^{2}	-12.36	59.12	62.90
Acidification (A) mol H+ eq.	4.92	0.46	0.89	0.13	0.56	0.31
Terrestrial eutrophication (TE) mol N eq.	23.54	2.58	5.36	2.33	3.03	1.26
Particulate matter (PM) kg PM2.5 eq.	0.11	2.22×10^{-2}	2.34×10^{-2}	-1.96×10^{-2}	1.14×10^{-2}	1.41×10^{-2}

All relevant parameters considered in the economic evaluation of the modelling example are reported in Table 12. In line with the geographical, technological and time-related representativeness of such modelling example, these parameters are meant to represent a hypothetical average European waste management context. The gate-fees have also been selected considering average plant size and efficiency. However, it must be highlighted that there is large variability in gate fees; this is particularly true for incinerators, where the gate fee for a small plant (e.g. < 200,000 t/year) can be significantly higher than that of larger plants (see Box 10).

Table 12: Estimation of the economic performance, as euros/tonne of food waste (based on data taken from WRAP, 2012 & 2015; CEWEP, 2008; EC, 2010d; EC, 2004)

Scenarios	Cost for authority for waste treatment (Gate-fees)	Subsidies	Taxes	Cost for society of waste treatment	Cost of collection and transport	Cost for society of waste management
AD	55	33	0	88	105	193
C	65	0	0	65	105	170

AD+C	80	33	0	113	105	218
I	100	45	0	144	64	208
Le	80	12	30	62	64	126
Lf	80	0	30	50	64	114

Below, results are provided for some of the impact categories as examples, since showing all the impact categories will be too long. They should be intended as exemplary attempts of presenting results, not as attempts to make recommendations on what should be done. The first three figures (Figure 18 to Figure 20) provide examples of the climate change impact category.

Figure 18 shows results with respect to climate change vs the cost for the authority for waste treatment (i.e. the gate-fees). Under this choice of environmental and economic indicators, only incineration and anaerobic digestion appear as optimal in comparison with the results obtained for the other options. On the other hand, when considering climate change in combination with the cost for society for waste treatment and for waste management (Figure 19 and Figure 20, respectively), all technologies appear as optimal except AD+C. These results highlight the considerable influence of any existing subsidies (and taxes) on the overall sustainability performance.

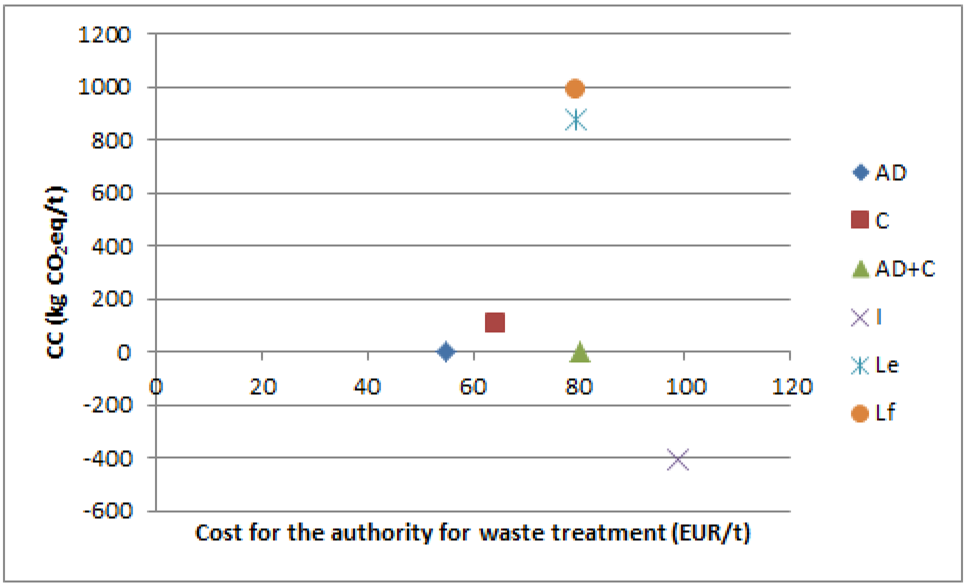


Figure 18: Climate change impact vs Cost for the authority for waste treatment

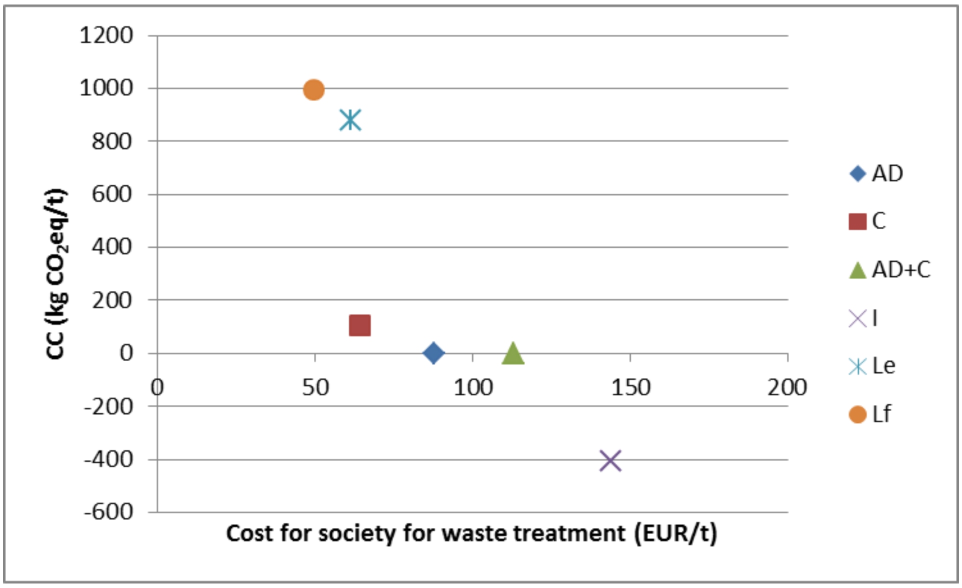


Figure 19: Climate change impact vs Cost for society for waste treatment

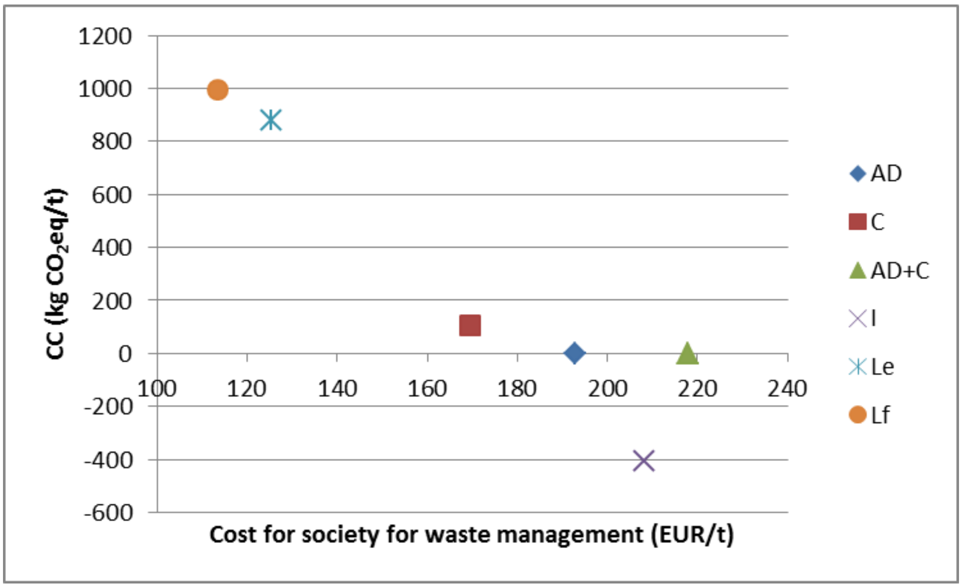


Figure 20: Climate change impact vs Cost for society for waste management

Further examples shown in Figure 21 to Figure 23 represent samples of the freshwater eutrophication impact category. In this case, Figure 21 shows results with respect to freshwater eutrophication vs the cost for the authority for waste treatment. Under this choice of environmental and economic indicators, three technologies appear as optimal: AD, C and AD+C. On the other hand, when considering freshwater eutrophication in combination with the cost for society for waste treatment and for waste management (Figure 22 and Figure 23, respectively), all technologies appear as optimal except I and AD.

In the specific case of the cost for society for waste treatment (Figure 22), the trade-off front show a big slope between C and Le which could make the decision maker reflect that C or AD+C is better option than Le or Lf since a little increase in cost (i.e. 3 EUR/t) can significantly reduce the impact by two orders of magnitude. This difference cannot be as well appreciated in the cost for society for waste management (Figure 23) since this same environmental trade-off is at the cost of 44 EUR/t.

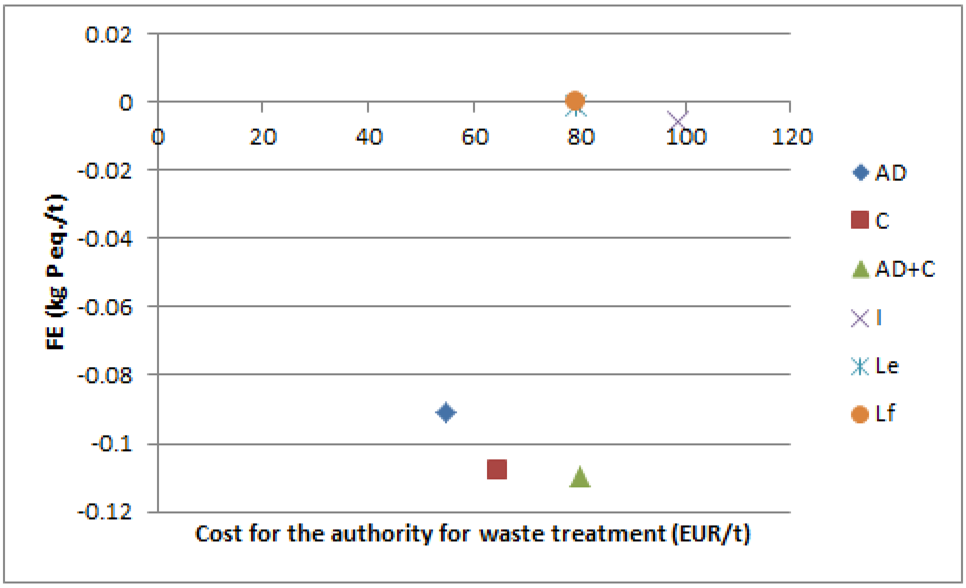


Figure 21: Freshwater eutrophication impact vs Cost for the authority for waste treatment

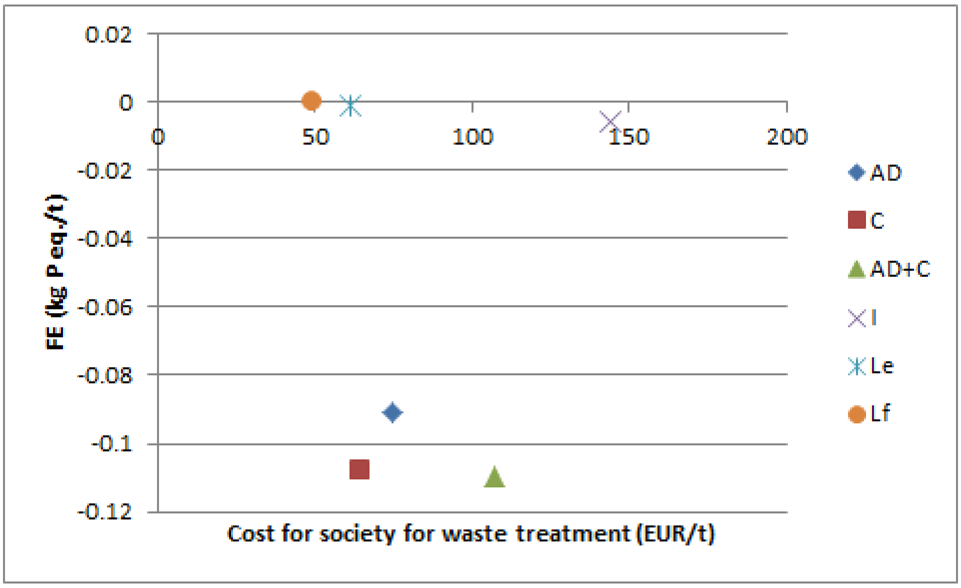


Figure 22: Freshwater eutrophication impact vs Cost for society for waste treatment

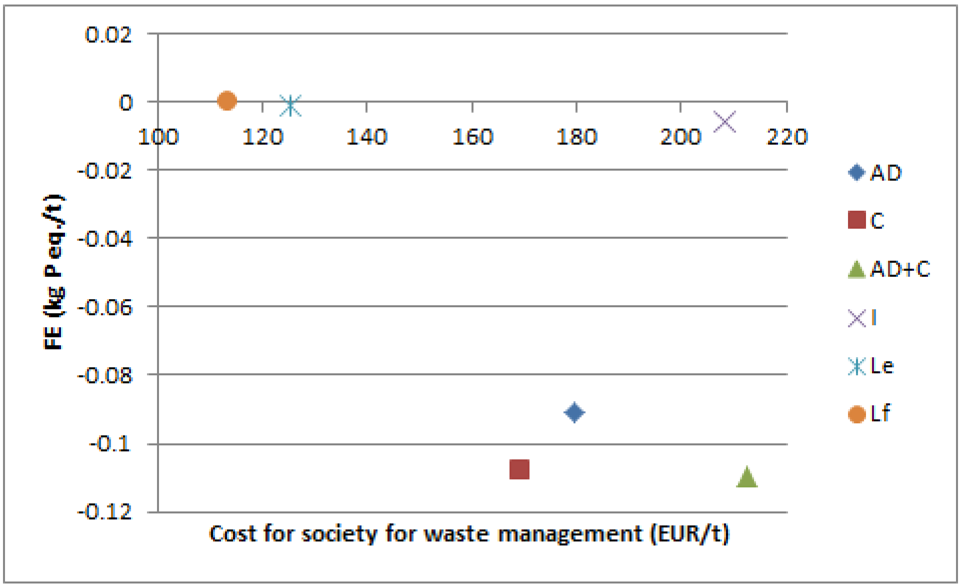


Figure 23: Freshwater eutrophication impact vs Cost for society for waste management

The EASETECH software allows both sensitivity and uncertainty propagation analysis. In this case study for matter of simplicity, only sensitivity analysis is performed. The most common technique is scenario analysis where assumptions are changed one-at-a-time. In Table 13, results are shown for the AD scenario, which was taken as example. First of all, a contribution analysis was performed to see which processes are the most contributing to the impact category climate change (chosen as example). Then four parameters were selected for the sensitivity analysis.

Table 13: Sensitivity analysis for the AD scenario (new values of parameters are chosen based on expert judgment)

Scenario / sub-scenario	**Process**	**Parameter**	**Default value**	**New value**	**Impact value (kg CO_2eq.)**
AD	Base scenario	-	-	-	-7.6×10^{-2}
AD/1	Electricity substitution	Efficiency	0.25 %	0.4 %	-88.4
AD/2	Use on land of the digestate	Distribution of carbon – C (soil storage)	13.2 %	9 %	14.22
AD/3	Use on land of the digestate	Distribution of carbon – C (soil storage)	13.2 %	14 %	-2.776
AD/4	Use on land of the digestate	Distribution of nitrogen – N_2O (air)	2.78 %	1 %	-1.43×10^{2}
AD/5	Fertilizer substitution	Average N fertilizer	0.4 %	0.2 %	52.7
AD/6	Fertilizer substitution	Average N fertilizer	0.4 %	0.6 %	-52.78

In order to show how these uncertain values can affect the selection of optimal technologies in the Pareto front analysis, Figure 24 shows the example of Climate Change vs Cost for society for waste treatment. As shown, AD will always be an optimal solution in any of the sub-scenarios, but depending on the sub-scenario, AD+C can either be optimal or not. For the base scenario of AD, AD+C was the only non-optimal technology. When selecting AD/2 and AD/5, AD+C becomes optimal.

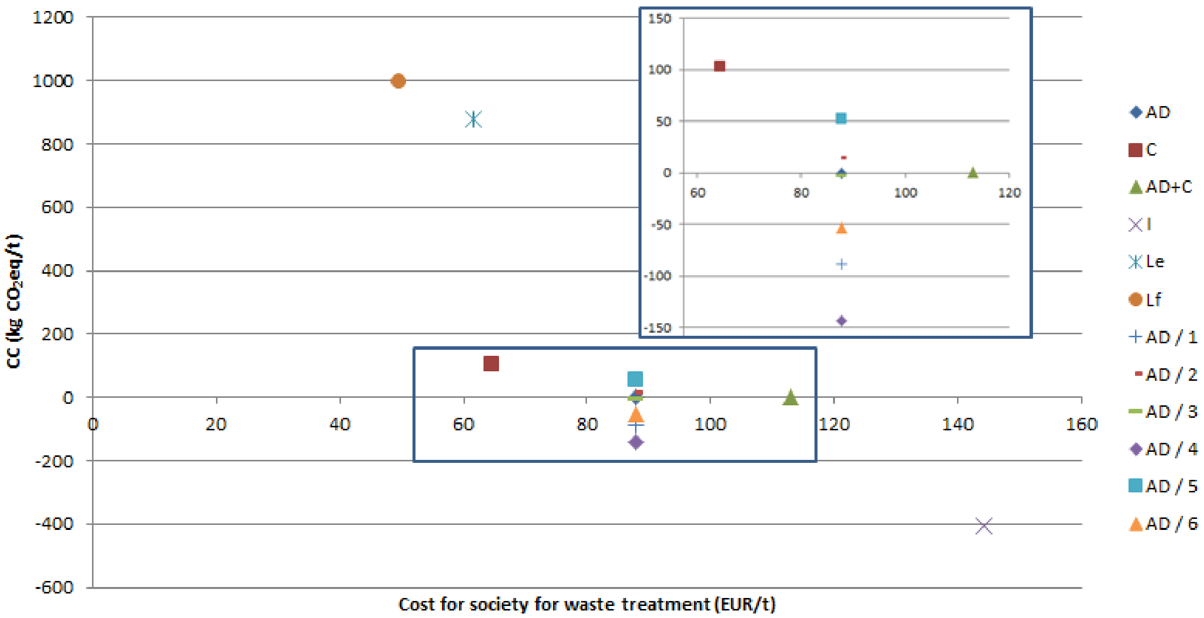


Figure 24: Example of sensitivity analysis for the modelled scenarios

6. Conclusions and Perspectives

While available information on European food waste generation and management routes is comprehensive and generally robust, knowledge on the sustainability of food waste management is still somewhat fragmented and not exhaustive. In addition, available tools to assess of the performance of waste management options are typically only focused on one of the three dimensions of sustainability (environmental, economic, social) and/or do not specifically focus on food waste.

Towards filling these gaps, the work presented in this report provides a life-cycle based methodology to assess environmental and economic sustainability of food waste management, as well as a simplified numerical case study. The proposed methodology is also able to accommodate additional assessment criteria, e.g. the social dimension of sustainability, thus gets closer to the development of a comprehensive sustainability assessment framework.

While application of the proposed methodology is in principle relatively straightforward (as from the numerical case study presented in Chapter 5), in reality the number of aspects to be carefully addressed, the variety of assumptions to be made and the amount of data needed may markedly increase the required efforts. In particular, the following aspects were found to be crucial:

- With respect to the identification of alternative food waste management options (step 1), while some may be already available in the waste management context selected, others may not be available but could equally be considered. This will influence all subsequent step, in that e.g. the emissions and costs related to building any non-existing installation for treating food waste will have to be carefully evaluated and accounted for. The time perspective thus may also change, as for instance the sustainability performance of non-available options may become attractive only after a certain time.
- LCAs that include food waste management may become particularly complex as, in addition to technical processes, also biological processes take place during the waste management chain. Other aspects that for instance are likely to exert a strong influence on the LCA results include the way of accounting for energy recovery and displacement, carbon storage and delayed emissions.
- The choice of the environmental and economic indicators to express the sustainability performance (step 3 and 4, respectively) is critical. Such choice, in fact, determines the way results are displaced, thus may influence the identification on the most sustainable option for food waste management. For instance, with respect to the environmental performance a broad range of impact categories should be accounted for and all of them should in principle be considered when displaying the results of the assessment. Any exclusion should be systematically justified.
- In the evaluation of the economic performance (step 4), the choice of the economic indicator strictly depends on the decision context, e.g. on the geographical scope and on who will pay the costs for food waste management. Using gate-fees can help provide straightforward estimations of costs. However, it should be stressed that, in principle, thorough LCC-based studies should be undertaken to estimate costs, as gate-fees may not provide accurate figures.
- The consideration of uncertainty (step 6) is crucial since apparent differences in the estimated impacts may be misleading if the uncertainty is large enough to overwhelm any relative differences between the compared alternatives.

The numerical case study presented along the methodology demonstrate that its application is relatively straightforward and the required efforts do not differ from those of typical life-cycle based evaluations. However, as mentioned, it should be stressed that due to the specific nature of food waste, biological processes take place during the waste management chain. These biological processes – which are highly dependent on local and interlinked factors such as soil profile, rainfall, and temperature – should in principle be carefully modelled in order to be accounted for in the evaluation. For the time being, the proposed methodology only partially accounts for such factors, thus room for improvements exist.

The work developed by JRC on food waste does not end here. Forthcoming work will be focused on the development of methodologies for setting up the prevention targets for food waste and the evaluation of different food waste prevention options in terms of their environment and economic performance.

ANNEX 1 - Source separation, collection and transport of food waste: overview and examples

The existing collection schemes in Europe

Different collection schemes for biowaste have been put in place in many EU Member States in order to comply with the waste management hierarchy and foster the quality composting option. As this report is focused on food waste, it is important to carefully address how this specific fraction is currently collected in those schemes and the main differences.

The existent schemes can be mainly divided into bring systems, relying on large road containers on the road serving a large number of buildings, and kerbside (door to door) schemes where a smaller bin is dedicated to a single building. Then the main dichotomy is into single stream collection of food waste only and commingled collection where it is allowed to deliver also garden waste in the same container.

How to maximise food waste capture

- ***Increase comfort and reduce nuisance at the source: tools for the kitchen***

The first stage of food waste collection obviously takes place in the kitchen, but in many collection schemes the key factors that are needed to enhance citizens' participation are neglected, not considering that citizens must be convinced about the easiness of this collection. For instance, the issue of the odour impact in food waste collection is very relevant. The so called "Yuck" factor (i.e. the effect of nuisances like disgusting smell), plays a major role. The idea that food scraps that were once attractive and edible products become a disgusting item once they are transferred from the plate to the collection bin is a common problem in food waste collection programs.

An effective tool to improve in this respect has been the use of vented caddy, coupled with compostable bags. The effect of this specific item is to assure air uptake inside it, leaving food waste drier and less smelly than using closed caddies. This item is now widely used where kerbside (door to door) food waste collection programmes are in place, e.g. in Italy, Catalonia, UK.

Dynamic olfactometry (sniffing tests) have been carried out in Italy using the European standard methodology EN 13725:2003[26] . Moreover, the determination of the hedonic has been performed: the hedonic tone is the relative pleasantness / unpleasantness of odor, according to the German standard VDI 3882[27]. The results of a specific study[28] highlight how the vented caddy can significantly reduce the odor impact, up to 10 times less.

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[26] EN 13725:2003 - European Standard for the Determination of odor concentration by dynamic dilution olfactometry for threshold measurement

[27] VDI 3882 - German Standard for the determination of Odour Intensity

[28] ARS ambiente, 2014, Olfactometry study on food waste caddies performed for Novamont Spa. Available at http://goo.gl/D1H4a3

Figure 25: Vented caddy for food waste collection, with compostable bags. Odour test with dynamic olfactometry comparing closed and vented caddies[29]

- ***Outside your kitchen: how to deliver easily***

Many studies[30] have shown that the capture rate of waste fractions to be collected separately for recycling present a negative correlation with the distance of the container to your household (Figure 26). The main advantage of kerbside food waste collection schemes, compared to "bring systems" based on large containers, is indeed the fact that the citizens can deliver easily the most important fraction that is generated in your house, emptying as frequently as needed the kitchen caddy into a temporary bin that is almost always placed in an internal part of the building. This bin is then put on the kerbside only for the collection day (typically twice per week).

Kerbside schemes, besides convenience and easiness, offer other advantages such as the possibility to monitor the individual production of every building in order to apply "Pay as You Throw" penalties and incentives or to leave dedicated warnings and advices related to the quality of the recyclable fraction. Other systems can be applied where there is absolutely no space in the internal common areas of the building, such as the so called "proximity schemes" where the bin is left permanently on the street, to be used by two or three buildings, with a specific lock and key.

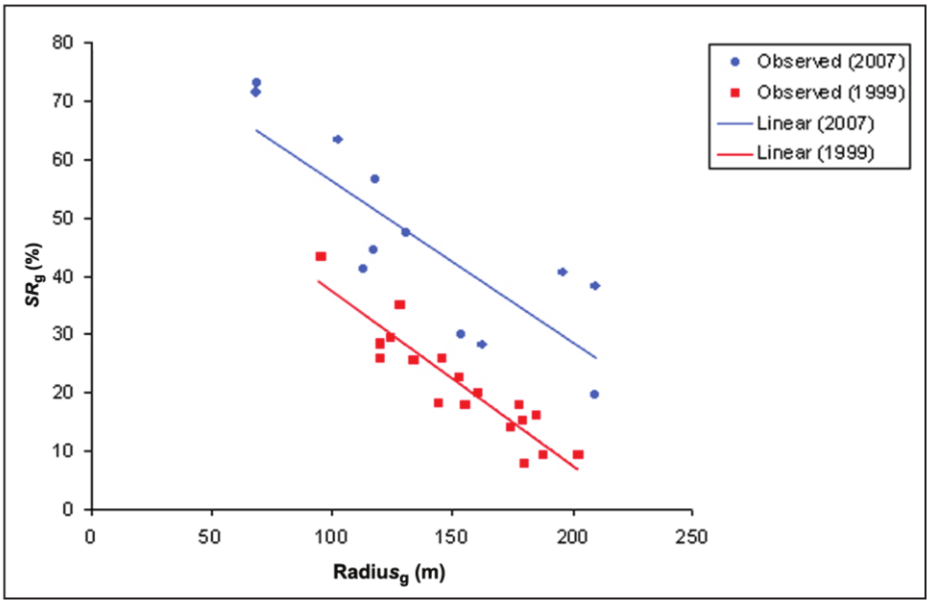

Figure 26: Separation Rate (Gross amount of waste collected in container / Total amount of waste generated) vs. distance between the container and the household. From Gallardo et al. 2012.

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[29] ARS Ambiente (2014), Olfactometry study on food waste caddies performed for Novamont Spa. Available at http://goo.gl/D1H4a3

[30] e.g. Evolution of Sorted Waste Collection: a Case Study of Spanish Cities

BOX 11 – Incentive schemes for food waste separation: examples

Across the EU, some local authorities like Sardinia (Italy) and Catalonia (Spain) experimented some interesting incentive schemes in order to foster the implementation of source separation of food waste.

In 2003, the Sardinia Region of Italy experienced a very low separate collection rate, averaging 3.8%. In the beginning of 2004, the Region decided to implement a new regulation offering a combination of economic incentives and penalties to municipalities in order to stimulate their adoption of separate food waste collection.

This regulation introduced in the initial phase a 30% surcharge on the standard landfill tipping fee for the municipalities not implementing food waste source separation, and a 30% reward for the ones beginning the new collection scheme, linked to a specific target: percentage of food waste collected, 10% minimum, and its quality, less than 5% impurities. In the following years these values were adjusted, raising the penalties and lowering the rewards, as the regional system evolved towards high participation by the municipalities.

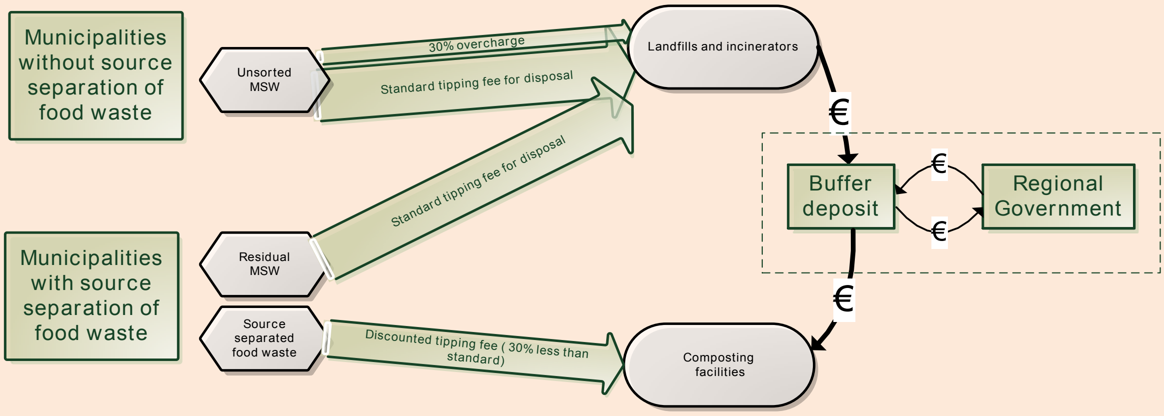

Figure 27: The penalty/reward scheme put in place by Sardinia Region after 2004[31]

Table 14: Details of the penalty, incentive and its access criteria during the most intense period of the scheme (2006-2008). After 2010, a new access criterion was added: the use of certified compostable bags. Source Sardinia Region (2011), annual report on penalty / reward scheme

Penalty	Discounted fee	Access criteria (min. % of food waste collected)	Access criteria (max impurities in food waste collected)	Access criteria (population involved in the collection)	Access criteria (min. % total source separation)
40%	30%	15%	5%	Only with total implementation	-

The effects of this system on the regional average in separate collection rate were impressive; in four years almost every municipality in Sardinia had implemented food waste collection. The regulation then introduced new targets to be achieved in terms of food waste capture: 15% of total municipal waste, and overall recycling rate up to

[31] M. Giavini (2012) - Successful Policies Supporting Residential Food Waste Collection: the Case Study of Sardinia, Italy. ISWA World Congress, 2012.

60%. Particular issues tackled were the quality of food waste collected, in terms of compostable materials, and the funding set up in order to provide citizens with compostable bags for collection.

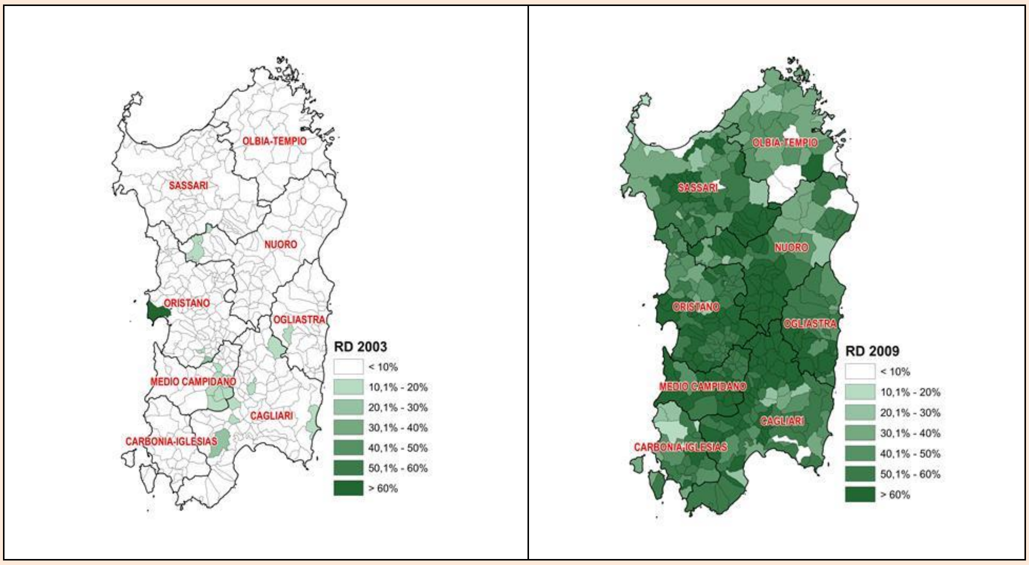

Figure 28: effect of the penalty/reward scheme on single municipalities in Sardinia from 2003 to 2009 (RD = recycling rate). M. Giavini (2012)

The region of Catalonia introduced a landfill tax in 2004 through the Catalan Law 16/2003 on financing waste treatment infrastructure and waste management taxes. The initial rate of the tax was 10.00 € /t of municipal waste, and since 2008 an incineration tax of 5.00 €/t. Since 2010 municipalities that have not initiated separate collection schemes for biowaste, are charged an incremented rate of 20.00 €/t for landfilling and 15.00 €/t for incineration, and the new regional Waste Management Plan (PreCat20) foresees a rapid increase up to € 47.10 in 2020.

The interesting feature of this system is that at least 50% of the revenue generated by the disposal tax has to be destined to biological treatment of biowaste and the mechanical-biological treatment (MBT) of residual waste, aiming at reducing the organic content of the residual waste finally going to landfill of incineration.

The remaining revenue is refunded to the local authorities according to their performance regarding separate collection of biowaste, which is incentivized by refunds that vary depending on the quantity and the quality of the biowaste delivered to the biological treatment plants. The quality in terms of purity/contamination rate is assessed by periodical composition analyses of the waste.

- ***What to collect, and when***

Two important issues that affect the participation of citizens to food waste collection are the collection frequency and the choice between collection of food waste only or commingled collection with garden waste.

At a national level, different approaches are followed regarding what kind of food waste to collect.

In the Netherlands (NL), Belgium (BE) and to some extent Germany (DE), Austria (A) only “VGF” (Vegetable, Garden, Fruit or “GFT”) is collected, which means that in this stream only uncooked food residues are allowed to be collected. Another possibility is to include also cooked food waste items but excluding meat and fish, this is already implemented in D and A, called the "Biotonne" system where it is collected commingled with yard waste.

The most comprehensive food waste collection where all food residues are allowed including cooked ones and meat or fish is in place in Italy (IT), UE, United Kingdom (UK), Sweden (SW), and some parts of Spain (Catalonia). This last system relies on the use of compostable bags to keep the receptacles clean. These systems show different results in terms of capture rate of food waste, as listed in the following table.

Table 15: Average food and yard waste capture rates with the most common collection schemes in place in Europe [32]

	Kg/capita/year of food waste captured	**Kg/capita/year of yard waste captured**
VGF excluding meat fish, and cooked	30-50	40-80
Biotonne incl. cooked but without meat and fish	40-50	40-80
Intensive food waste collection with compostable bags	70-120	0

The collection frequency is key when thinking about enhancing citizen's participation and optimising costs.

Addressing best practices, collection schemes dedicated to food waste are based on a twice per week frequency, up to three times a week in Mediterranean areas with hot weather during summer, while central-european commingled schemes like the Biotonne rely on a weekly or even fortnightly collection. Actually, there are some optimisations that generally lead to important savings in collection costs without affecting citizens' participation, such as the following:

- **Reduce the collection frequency for residual waste**: this is the real key for the success of recycling programmes in general, even more effective if combined with a Pay-As-You-Throw (PAYT) variable waste tax;
- **Use of a two-compartment light weight vehicles** for collecting both food waste and another fraction such as residual waste at the same time, taking advantage of the high density of food waste and the low density of all the others;

Keep food waste collection frequency at the minimum needed, taking into account the climate condition and the possible, use of innovative tools such as the vented caddy with compostable bags, by which food scraps can be kept in the kitchen without particular nuisance and then stored in a temporary bin (30-40 liters for detached houses, 120 liters for multifamily buildings).In some successful experiences (e.g. Province of Bergamo - Italy, or many districts in UK), in many municipalities food waste collection takes place weekly, as residual waste, with high capture rates (80-90 kg/capita/year).

[32] M. Giavini (2013). Capture Rates of Source Separated Organics: a Comparison Across EU, with a Focus on Metropolitan Areas. ISWA World Congress, 2013.

Figure 29: Two-compartment light vehicle for the combined collection of food waste and residual waste to optimize routes and costs

- ***Biowaste or food waste? How to manage Garden waste***

There are some clear advantages by keeping food waste separate from garden waste, the most important being the fact that garden waste can be treated in simple and low cost composting facilities.

If the collection scheme is commingled then a significant amount of the mixture ("biowaste") will be garden waste and most importantly the gate fee for composting or anaerobic digestion (AD), will be significantly higher than the option in which garden waste (collected only at household waste recycling centres) is mainly composted by itself in open air windrows, and food waste (source separated) is digested anaerobically or composted along with only some of the garden waste collected. This is because open air windrow is a much cheaper way of treating garden waste than AD or food waste composting.[33]

When simple composting plants dedicated to garden waste only are operating, gate fees fall in the range of 25-35 €/t, while those for food waste in complex composting plants are around 70-80 €/t.

Another option for managing garden waste minimizing costs for the municipality should be home composting, favourable on environmental grounds and effective when properly implemented and managed with appropriate training courses.

The importance of food waste quality

When implementing a wide strategy aimed at increasing the capture rate of food waste, one of the main concerns is related to the possible risk of having more impurities in the collected food waste, as a result of a possible citizen's low awareness when addressing waste fractions in general. That's why it's important to constantly perform waste composition analyses on the food waste collected, in order to better understand when and with which collection scheme there are more impurities.

In Italy, a good experience is carried out by the Italian Composting Association (CIC) which routinely carries out 500-700 analyses every year, on behalf of the composting plants. Those are performed because the plants want to check the quality of input biowaste, sometimes to be able to apply different gate fees according to the quality (and this is indeed a good strategy to improve quality at the source).

[33] Eunomia, Dealing with Food Waste in the UK, 2007.

Figure 30: Waste composition analysis performed in Italy by the Italian Composting Association

A good strategy has been implemented in Catalonia (Spain) where a different gate fee is applied for composting according to the level of impurities.

Table 16: Modulation of gate fees for composting according to the level of impurities of biowaste treated (source: ARC Catalunya - 2010)

Impurity level in food waste	Gate fee
< 5%	€ 41,96
5 - 10%	€ 44,90
10-15%	€ 50,13
15-20%	€ 50,54
20-25%	€ 58,91
25-30%	€ 67,53
30-35%	€ 70,20
35-40%	€ 75,96

A recent statistical review of all 3700 analyses performed in Italy from 2008 to 2014 (source: CIC, 2014) shows that:

- 27% of the analyses have <2,5% non-compostable materials (excellence)
- 62% of the analyses have <5% non-compostable materials (good)
- 89% of the analyses have <10% non-compostable materials (accettable)

It's worth to point out that most of the analyses have been performed where the typical Italian door to door scheme is in place. Indeed, a further distinction could be applied showing the significantly higher impurities in bring schemes based on large road containers (see the following figure).

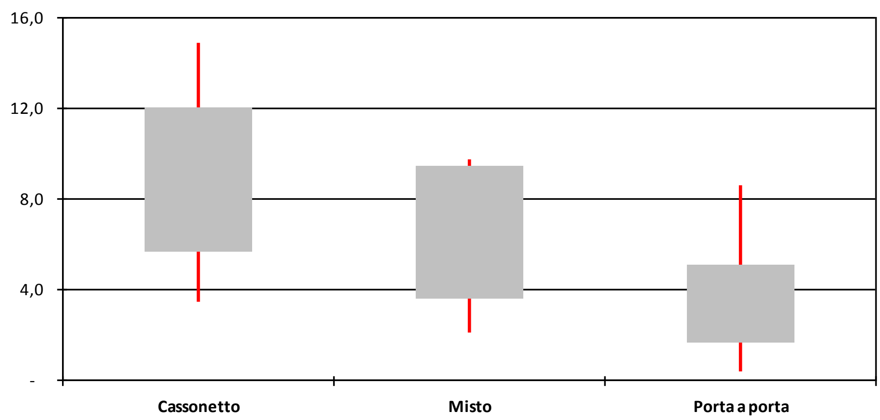

Figure 31: Average % impurities (non-compostable materials) in the three main collection schemes in place in Italy (source: CIC, 2013) Cassonetto = large road containers, Misto = mixed, Porta a porta = door to door.

The role of compostable bags in food waste collection

The use of compostable bags is itself a comprehensive shift towards a new approach in dealing with food waste management. Although they are more expensive than poliethylene bags, their use is effective in increasing capture rate as a result of increased comfort for citizens, and keeping impurities at a low amount both in the input and in the final compost thus facilitating the treatment.

In many Member States new regulations are coming into force, banning the use of polyethylene shopper bags, in some cases (e.g. Italy) allowing only thick reusable bags or compostable. So now citizens simply use for food waste collection the same shopper they find in supermarkets, with a view to more sustainability and reduction of environmental harms such as marine littering caused by polyethylene bags.

Compostable bags are an effective solution used in IT, UK, ES, NO, IE also to keep receptacles clean and to maintain collection frequencies at the minimum needed, typically twice a week.

How to optimize food waste transportation?

Collection costs are strictly related to the vehicles used for the collection of food waste. While for residual waste large compacting trucks are typically used, in order to keep collection costs low for intensive food waste collection maintaining at the same time a high capture rate, the use of cheap and light vehicles can generate important savings with respect to large trucks.

While in densely populated cities large trucks could still be used, in municipalities with lower population the most effective system is based on the use of "satellite" small vehicles (5-7 cu.m tanks), with single driver and no helper, that collect up to 1-2.5 tonnes of food waste and then deliver either directly to the composting plant, to a transfer station or to single large truck by coupling with it on the street. This large truck also performs some collection during spare time between the coupling transfers with the satellites.

Some centralized collection points may serve as a temporary parking for large trucks which serve as main vehicle in which small-tank vehicles deliver the food waste collected door to door.

Figure 32: On the most effective and cost saving scheme for food waste collection involves the use of light satellites, a large truck and a limited number of coupling on the street for emptying the satellites.

Food waste collected in a municipality is usually delivered to a transfer point from which it is moved to the final treatment via large trucks (secondary transportation). It must be pointed out that the proximity principle may not apply so strictly when talking about food waste. Sometimes concerns may be raised related to the fact that composting facilities are not available in a certain radius around the municipality performing food waste collection, but actually, reviewing existing LCA studies[34], it comes out that the benefits of composting with respect to landfilling are so huge that overcome by far the environmental impact contribution due to the final transportation stage. This is important to take into account when considering the option of centralized medium-large scale facility *vis-a-vis* decentralized small ones, that although may allow some saving in terms of transport, may not be so easily managed as they require a much higher quality of the collected food waste.

[34] e.g. Progetto Gerla: a LCA assessment of the Waste Management Plan of Lombardy. Regione Lombardia, Italy, 2014.

References

Ahmed, I.I., and Gupta, A.K., (2010). Pyrolysis and gasification of food waste: Syngas characteristics and char gasification kinetics. Applied energy, 87: 101 – 108.

Allacker K., Mathieux F., Manfredi S., Pellettier N., de Camillis C., Ardente F., Pant R., (2014). Allocation solutions for secondary material production and end of life recovery: proposals for policy initiatives. *Resources, Conservation & Recycling,* 88: 1-12

Arafat, H.A., Jijakli, K., Ahsan, A., (2015). Environmental performance and energy recovery potential of five processes for municipal solid waste treatment. *Journal of Cleaner Production*, 105: 233 – 240.

Baker, J.W., and Lepech, M.D., (2007). Treatment of uncertainties in life cycle assessment. Standfor University, Stanford, USA.

Bartl, A., (2014). Moving from recycling to waste prevention: A review of barriers and enables. *Waste Management Research*, 32: 3 – 18.

Bernad-Beltrán, D., Simó, A., Bovea, M.D., (2014). Attitude towards the incorporation of the selective collection of biowaste in a municipal solid waste management system. A case study. *Waste Management*, 34(12): 2434 – 2444.

Bernstad, A., and la Cour Jansen, J., (2011). A life cycle approach to the management of household food waste - A Swedish full-scale case study. *Waste management*, 31(8): 1879 – 96.

Bernstad, A., and la Cour Jansen, J., (2012). Review of comparative LCAs of food waste management systems – current status and potential improvements. *Waste management*, 32: 2439 – 2455.

CEWEP - Confederation of European Waste-to-Energy Plants - (2008) Report on incineration gate fees, in: Analisi prezzi medi impianti Anno 2009, Regione Emilia Romagna (IT).

Cherubini, F., Bargigli, S., Ulgiati, S., (2009). Life cycle assessment (LCA) of waste management strategies: Landfilling, sorting plant and incineration. Energy, 34(12): 2116 – 2123.

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P., Hauschild, M., (2009). C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. *Waste management and Research* 27(8): 707 – 715.

Ciroth, A., and Franze, J., (2011). Presentation on conducting a Social LCA, Berlin, GreenDelta-2011. Available on line: http://www.greendelta.com/uploads/media/GreenDeltaTC_Franze_Ciroth.pdf

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., (2014). An environmental assessment system for environmental technologies. *Environ. Model. Softw*., 60: 18 – 30.

Clavreul, J., Guyonnet, D., Christensen, T.,H., (2012). Quantifying uncertainty in LCA-modelling of waste management systems. *Waste management*, 32: 2482 – 2495.

Corrado, S., Ardente, F., Sala, S., Saouter, E., (2016). Modelling of food loss within life cycle assessment: towards a methodological framework. Submitted to *Journal of cleaner production*.

Cox, J., Giorgi, S., Sharp, V., Strange, K., Wilson, D., Blakey, N., (2010). Household waste prevention — a review of evidence, *Waste Management and Research*, 28: 193 – 219.

Cristobal, J., Limleamthong, P., Manfredi, S. & Guillen-Gosalbez, G. (2016). Methodology for assessment and retrofit of food waste management options according to environmnetal criteria via data envelopment analysis and life cycle assessment. Submitted to *Journal of Cleaner Production*.

Del Borghi, A., Gallo, M., Del Borghi, M., (2009). A survey of life cycle approaches in waste management. The *International Journal of Life Cycle Assessment*, 14(7): 597 – 610.

Del Cimmuto, A., Mannocci, A., Ribatti, D., Boccia, A., La Torre, G., (2014). Impact on knowledge and behaviour of the general population of two different methods of solid waste management: An explorative cross-sectional study. *Waste Management and Research*, 32: 556 – 561.

Dreyer, L., Hauschild, M.Z., Schierbeck, J., (2006). A framework for Social Life Cycle Impact Assessment. *The International Journal of Life Cycle Assessment*, 11(2): 88 – 97.

Dururu, J., Anderson, C., Bates, M., Montasser, W., Tudor, T., (2015). Enhancing engagement with community sector organisations working in sustainable waste management: A case study, *Waste Management and Research*, 33: 284 – 290.

Dyckhoff, H., Allen, K., (2001). Measuring ecological efficiency with data envelopment analysis (DEA). *European Journal of Operational Research*, 132: 312 – 325.

EC – European Commission, (1999). Directive 1999/31/EC of the Council of the European Union of 26 April 1999 on the landfill of waste – *Official Journal of the European Communities* L 182/1. Available online at http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:31999L0031

EC – European Commission, (2000). Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste – Official Journal of the European Communities L 332/91. Available online at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2000:332:0091:0111:EN:PDF

EC – European Commission, Directorate General Environment (DG ENV), (2001). EUNOMIA report: Costs for Municipal Waste Management in the EU. Available online at http://ec.europa.eu/environment/waste/studies/pdf/eucostwaste.pdf

EC – European Commission, (2004). COWI report: Preliminary Impact Assessment for an Initiative on the Biological Treatment of Biodegradable Waste. Available online at http://ec.europa.eu/environment/waste/compost/pdf/ia_biowaste_directive_report.pdf

EC – European Commission, (2008). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives – *Official Journal of the European Union* L 312/3. Available online at http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0098&from=EN

EC – European Commission, (2009). Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32009L0028&from=en

EC – European Commission, (2010a). Preparatory study on food waste across EU 27. Technical Report. Available online at http://ec.europa.eu/environment/eussd/pdf/bio_foodwaste_report.pdf

EC – European Commission, (2010b). Communication COM(2010)235 on future steps in bio-waste management in the European Union – *Official Journal of the European*

Union. Available online at http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52010DC0235&from=EN

EC - European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), (2010c). International Reference Life Cycle Data System (ILCD) Handbook: "General guide for Life Cycle Assessment - Detailed guidance", "General guide for Life Cycle Assessment – Provisions and action steps", "Specific guide for Life Cycle Inventory data sets", "Recommendations for Life Cycle Impact Assessment in the European context", "Framework and requirements for Life Cycle Impact Assessment (LCIA) models and indicators", "Review Schemes for Life Cycle Assessment (LCA)", "Reviewer qualification for Life Cycle Inventory (LCI) data sets", "Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment", "Nomenclature and other conventions". Publications Office of the European Union. Available online at http://eplca.jrc.ec.europa.eu/?page_id=86

EC – European Commission, (2010d). Assessment of the options to improve the management of biowaste in the European Union. Available online at: http://ec.europa.eu/environment/waste/compost/pdf/ia_biowaste%20-%20final%20report.pdf

EC - European Commission, (2011a). A resource-efficient Europe – Flagship initiative under the Europe 2020 Strategy. COM(2011b) 21. Available online at http://ec.europa.eu/resource-efficient-europe/pdf/resource_efficient_europe_en.pdf

EC – European Commission, (2011b). Communication COM(2011a) 571 Roadmap to a Resource Efficient Europe – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52011DC0571&from=EN

EC – European Commission (2011c): DG ENV report, Guidelines On The Preparation Of Food Waste Prevention Programmes. Available online at http://ec.europa.eu/food/safety/food_waste/library/index_en.htm

EC - European Commission, (2011d). Supporting environmentally sound decisions for waste management - Technical guidelines (3 reports). Available online at http://sa.jrc.ec.europa.eu/?page_id=777

EC – European Commission, (2012a). Communication COM(2012) 60 Innovating for Sustainable Growth: A Bioeconomy for Europe – *Official Journal of the European Union*. Available online at http://ec.europa.eu/research/bioeconomy/pdf/official-strategy_en.pdf

EC – European Commission, (2013a). Communication COM(2013) 36 Setting up a European Retail Action Plan – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2013:0036:FIN:EN:PDF

EC – European Commission, (2013b). Communication COM(2013) 196 Building the Single Market for Green Products – Facilitating better information on the environmental performance of products and organisations – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2013:0196:FIN:EN:PDF

EC - European Commission, (2013c). RECOMMENDATION 2013/179/EU on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations, Annex II (Product Environmental Footprint (PEF) Guide) – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2013:124:0001:0210:en:PDF

EC - European Commission, (2013d). RECOMMENDATION 2013/179/EU on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations, Annex III (Organisational Environmental Footprint (OEF) Guide) – *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2013:124:0001:0210:en:PDF

EC – European Commission, (2014a). Communication COM(2014) 397 proposal for a Directive of the European Parliament and of the Council amending Directives 2008/98/EC on waste, 94/62/EC on packaging and packaging waste, 1999/31/EC on the landfill of waste, 2000/53/EC on end-of-life vehicles, 2006/66/EC on batteries and accumulators and waste batteries and accumulators, and 2012/19/EU on waste electrical and electronic equipment *Official Journal of the European Union*. Available online at http://eur-lex.europa.eu/resource.html?uri=cellar:e669092f-01e1-11e4-831f-01aa75ed71a1.0001.01/DOC_1&format=PDF. This COM has been withdrawn in February 2015, after which new consultations have started towards developing a new proposal for a COM by the end of 2015.

EC - European Commission, (2014b). Towards a circular economy: A zero waste programme for Europe. COM(2014) 398. Available online at http://eur-lex.europa.eu/resource.html?uri=cellar:50edd1fd-01ec-11e4-831f-01aa75ed71a1.0001.01/DOC_1&format=PDF. This COM has been withdrawn in February 2015, after which new consultations have started towards developing a new proposal for a COM by the end of 2015.

EC – European Commission, (2015a). Towards a circular economy. New package released on December 2nd, 2015. All documents included in the package are available online at http://ec.europa.eu/priorities/jobs-growth-investment/circular-economy/index_en.htm

EC – European Commission, (2015b). Communication COM(2015) 614/2: Closing the loop – An EU action plan for the Circular Economy. Available online at http://ec.europa.eu/priorities/jobs-growth-investment/circular-economy/docs/communication-action-plan-for-circular-economy_en.pdf

EC – European Commission, (2015b). Communication COM(2015) 595: Proposal for a Directive of the European Parliament and of the Council amending Directive 2008/98/EC on waste. Available online at http://ec.europa.eu/priorities/jobs-growth-investment/circular-economy/docs/amending-directive-waste_en.pdf

EU FUSIONS - FP7 project, (2015). Food Use for Social Innovation by Optimising Waste Prevention Strategies (FUSIONS), http://www.eu-fusions.org/

Ekvall, T., Assefa, G., Björklund, A., Eriksson, O., Finnveden, G., (2007). What life-cycle assessment does and does not do in assessments of waste management. *Waste Management*, 27(8): 989 – 996.

European Food SCP Roundtable, (2010). Voluntary environmental assessment and communication of environmental information along the food chain, including to consumers, Guiding Principles. Available online at http://www.food-scp.eu/sites/default/files/Guiding_Principles.pdf

European Food SCP Roundtable, (2011). Communicating environmental performance along the food chain. Available online at http://www.food-scp.eu/files/ReportEnvComm_8Dec2011.pdf

European Food SCP Roundtable, (2012). Continuous Environmental Improvement (http://www.food-scp.eu/files/Continuous_Environmental_Improvement_FINAL_21_November_2012.pdf

European Food SCP Roundtable, (2013). ENVIFOOD Protocol, Environmental Assessment of Food and Drink Protocol, European Food Sustainable Consumption and Production

Round Table (SCP RT), Working Group 1, Brussels, Belgium. Available online at (http://www.food-scp.eu/files/ENVIFOOD_Protocol_Vers_1.0.pdf

EUROSTAT, (2007). Data referring to population on 2007. http://ec.europa.eu/eurostat

FAO – Food and agriculture organization of the United Nations, (2004). Protein sources for the animal feed industry. Rome. Available on line: http://www.fao.org/docrep/007/y5019e/y5019e00.htm

FAO – Food and agriculture organization of the United Nations, (2011). Global food losses and food waste – Extent, causes and prevention. Rome. Available on line: http://www.fao.org/docrep/014/mb060e/mb060e.pdf

FAO – Food and agriculture organization of the United Nations, (2013). Food wastage footprint - Impacts on natural resources. Rome. Available on line: http://www.fao.org/docrep/018/i3347e/i3347e.pdf

FAO – Food and agriculture organization of the United Nations, IFAD and WFP, (2015). The Sate of Food Insecurity in the World 2015 - Meeting the 2015 international hunger targets: taking stock of uneven progress. Rome. Available on line: http://www.fao.org/3/a-i4646e.pdf

Gallardo A, et al. Evolution of sorted waste collection: a case study of Spanish cities. *Waste Management & Research*, 30(8):859-63.

ISO 14040:2006 (2006b). International Standard – Environmental management – Life cycle assessment – Principles and framework. International Organization for Standardization. Geneva, Switzerland.

Kim, M.H., and Kim, J.W., (2010). Comparison through a LCA evaluation analysis of food waste disposal options from the perspective of global warming and resource recovery. *Science of total environment*, 408: 3998 – 4006.

Kim, M.H., Song, Y.E., Song, H.B., Kim, J.W., Hwang, S.J., (2011). Evaluation of food waste disposal options by LCC analysis from the perspective of global warming: Jungnang case, South Korea. *Waste Management*, 31: 2112 – 2120.

Kretschmer, B., Buckwell, A., Smith, C., Watkins, E., Allen, B., (2013). Technology options for recycling agricultural, forestry & food wastes and residues for sustainable bioenergy and biomaterials. Report for the European Parliament, STOA, as part of the study "Technology Options for Feeding 10 Billion People". IEEP: London.

Kummu, M., de Moel, H., Porkka, M., Siebert, S., Varis, O., Ward, P.J., (2012). Lost food, wasted resources: global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. *Science of the total environment*, 438: 477 – 489.

Manfredi, S. & Christensen, T.H. (2009). Environmental assessment of solid waste landfilling technologies by means of LCA-modelling. Waste Management, 29: 32 – 43.

Manfredi, S. & Cristobal, J. (2016). Towards more sustainable management of European food waste: methodological approach and numerical application. Submitted to *Journal of Cleaner Production*.

Manfredi, S., & Pant, R., (2013). Improving the environmental performance of bio-waste management with life cycle thinking (LCT) and life cycle assessment (LCA). *The International Journal of Life Cycle Assessment*, 18(1): 285 – 291.

Manfredi, S., & Goralczyk, M., (2013). Life cycle indicators for monitoring the environmental performance of European waste management. *Resources, Conservation and Recycling,* 81: 8 – 16.

Manfredi, S., Pant, R., Pennington, D., Versmann, A., (2011). Supporting environmentally sound decisions for waste management with LCT & LCA. *The International Journal of Life Cycle Assessment,* 16(9): 937 – 939.

Manfredi, S., Scharff, H., Tonini, D., Christensen, T.H., (2009). Landfilling of waste: accounting of greenhouse gases and global warming contributions. *Waste management & Research,* 27(8): 825 – 836.

Martinez-Sanchez, V., Kromann, M.A. & Astrup, T.F., (2015). Life cycle costing of waste management systems: Overview, calculation principles and case studies. *Waste management*, 36: 343 – 355.

Messac, A., Ismail-Yahaya, A., Mattson C.A., (2003). The normalized normal constraint method for generating the Pareto frontier. *Structural and multidisciplinary optimization*, 25 (2): 86 – 98.

Nakakubo, T., Tokai, A., Ohno, K., (2012). Comparative assessment of technological systems for recycling sludge and food waste aimed at greenhouse gas emissions reduction and phosphorus recovery. *Journal of Cleaner Production*, 32: 157 – 172.

Papargyropoulou, E., Lozan, R., Steinberger, J.K., Wright, N., (2014). The food waste hierarchy as a framework for the management of food surplus and food waste. *Journal of Cleaner Production*, 76: 106 – 115.

Petersson, A., Wellinger, A., (2009). Biogas upgrading technologies – developments and innovations. IEA Bioenergy. Task 37 – Energy from biogas and landfill gas. Available on line: http://www.iea-biogas.net/files/daten-redaktion/download/publi-task37/upgrading_rz_low_final.pdf

Pressley, P.N., Aziz, T.N., DeCarolis, J.F., Barlaz, M.A., He, F., Li, F., Damgaard, A., (2014). Municipal solid waste conversion to transportation fuels: a life-cycle estimation of global warming potential and energy consumption. *Journal of Cleaner Production*, 70: 145 – 153.

Rangaiah, G.P., (2009). Multi-objective optimization: Techniques and applications in Chemical Engineering. Advances in Process Systems Engineering – Vol. 1. World Scientific Publishing. Singapore.

Regione Lombardia (2014). Progetto Gerla: a LCA assessment of the Waste Management Plan of Lombardy. In: Piano Regionale di Gestione dei Rifiuti, Lombardy Region.

Rybaczewska-Blazejowska, M., (2013). Economic, environmental and social aspects of waste management- the LCA analysis. *Prace Naukowe Akademi i im. Jana Dlugosza w Czestochowie,* 7: 239 – 250.

Spies, S., Murdock, S., White, S., Krannich, R., Wulfhorst, J.D., Wrigley, K., Leistritz, L., Sell, R., Thompson, J., (1998). Waste facility experience and perceptions of waste-related health and safety risks. *Society and Natural Resources*, 11(8): 719 – 741.

UK – House of Lords, European Union Committee (2014): Counting the Cost of Food Waste: EU Food Waste Prevention – Authority of the House of Lords, London.

UNEP – United Nations Environment Programme (2014): Prevention and reduction of food and drink waste in businesses and households - Guidance for governments, local authorities, businesses and other organisations, Version 1.0.

UNEP/SETAC (2009) Guidelines for social life cycle assessment of products. The UNEP/SETAC Life Cycle Initiative. Available online at http://www.lifecycleinitiative.org/starting-life-cycle-thinking/life-cycle-approaches/social-lca/

US EPA – United States Environmental Protection Agency (2014). Catalog of CHP Technologies. Available on-line: http://www.epa.gov/chp/documents/catalog_chptech_full.pdf

Vandermeersch, T., Alvarenga, R.A.F., Ragaert, P., Dewulf, J., (2014). Environmental sustainability assessment of food waste valorization options. *Resources, Conservation and Recycling*, 87: 57 – 64.

Wilson, D.C., Parker, D., Cox, J., Strange, K., Willis, P., Blakey, N., Raw, L., (2012). Business waste prevention: a review of the evidence. *Waste Management and Research*, 30: 17 – 28.

WRAP – Waste and Resource Action Programme – UK, (2007). Annual report prepared by Eunomia: Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis. Available online: http://www.wrap.org.uk/sites/files/wrap/Biowaste_CBA_Final_Report_May_2007.pdf

WRAP – Waste and Resource Action Programme – UK, (2009). Household Food and Drink Waste in the UK. Available on line: http://www.wrap.org.uk/sites/files/wrap/Household_food_and_drink_waste_in_the_UK_-_report.pdf

WRAP – Waste and Resource Action Programme – UK, (2012). Comparing the cost of alternative waste treatment options – Gate fees report. Availabel on line: http://www.wrap.org.uk/sites/files/wrap/Gate%20Fees%20Report%202012.pdf

WRAP – Waste and Resource Action Programme – UK, (2015) Gate fees report: Comparing the cost of alternative waste treatment options in the UK.

Zhou, P., Ang, B.W., Poh, K.L., (2008). A survey of data envelopment analysis in energy and environmental studies. *European Journal of Operational Research*, 189: 1 – 18.

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