Introduction

Context and objectives

Extensive scientific literature has been produced to provide assessment of the environmental performance of waste management systems and strategies (e.g. Arafat et al., 2015; Christensen et al., 2009; Pressley et al., 2014). At the same time, research initiatives also focused on specific waste streams, as opposed to focusing on, for example the overall municipal solid waste (MSW). For instance, a number of studies were undertaken to identify key drivers of food wastage and to quantify food waste generation (e.g. Bräutigam et al., 2014; European Commission, 2010a, 2014a, 2014b, 2016; Food and Agriculture Organization, 2011a, 2011b; Priefer et al., 2016).

While these initiatives provided fairly accurate information over European food waste generation quantities and management routes, 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 focus on one specific sustainability dimension, for example only the environmental (Nakakubo et al., 2012) or the economic (Kim et al., 2011) dimension.

This is currently changing owing to increasingly challenging sustainability targets and requirements enforced by recent policy documents – such as the Communication ‘A resource-efficient Europe’ under the EU 2020 Strategy (European Commission, 2011), the recommendations on environmental footprint of products (European Commission, 2013a) and the recent Communication on Circular Economy (European Commission, 2015a and 2015b) – which have boosted research also on the methodological side, for example towards developing methods to evaluate resource efficiency and sustainability. Room for improvements exists, especially to harmonise current assessment approaches and adapt them to the specific context of food waste management. This article is an attempt to reduce such gaps.
Towards this goal and also aiming at providing a valuable tool to support science-based policy making and decision making, the work presented in this article provides a life-cycle based framework methodology to simultaneously quantify the environmental and economic performance of European food waste management, which is indeed a novelty. This methodology makes use of a comprehensive set of indicators that provide comprehensive assessment of environmental aspects (12 indicators) and economic aspects (three indicators). A numerical case study is developed in parallel, as an example of application of the proposed framework methodology.

The methodology implicitly assumes that prevention of food waste generation is the most sustainable option. However, the methodology does not deal directly with food waste prevention and, instead, focuses on the food waste flow that needs to be managed/treated.

The authors are currently committed to expand the methodology presented in this article to include the third dimension of sustainability, that is, the social dimension. This was, however, excluded from the scope of this article because, at the time of writing, existing approaches to quantify social impacts were considered by the authors not sufficiently robust and broadly accepted.

**Food waste generation and sustainability concerns**

Worldwide, over 1.3bt of food for human consumption is wasted or lost annually (Food and Agriculture Organization, 2011a) throughout the food supply chain (FSC) (i.e. from agriculture production, transport, processing, distribution and consumption), which is equivalent to about one-third of the total global food production. In Europe, estimates from 2014 show up to 100mt of food waste generated every year (European Commission, 2015c), corresponding to approximately 200kg per capita (considering a population of 503 million people in the EU27 in year 2014, as from Eurostat (European Commission, 2015d)).

Current levels of food waste generations lead to significant environmental impacts, as well as to economic and social costs. For instance, 2007 worldwide figures provided by the Food and Agriculture Organization (FAO) (Food and Agriculture Organization, 2011b) on the consequences of food produced for human consumption that had been lost or wasted include: 3.3 Gtonnes of CO₂ eq. emitted to the atmosphere, 250 km² of surface and groundwater consumption and an occupation of nearly 28% of the world’s agricultural land. In Europe, food waste is responsible for about 170mt of CO₂ eq. per year (European Commission, 2010a). Worldwide, the economic costs of food waste were estimated to be 1055 billion USD annually (Food and Agriculture Organization, 2011b).

**Legislative background**

Existing European legislation does not provide a definition of food waste, leaving room for different interpretations (European Commission, 2014a). Food waste is regarded as a main constituent of the broader group bio-waste, which is defined in the European Waste Framework Directive (WFD) 2008/98/EC (European Commission, 2008) as ‘[…] biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants’. The Landfill Directive (LD) 1999/31/EC clarified that an even broader group exists called biodegradable waste, which includes ‘any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and cardboard’ (European Commission, 1999).

The LD had set mandatory targets to progressively reduce the share of biodegradable municipal waste put into landfills. However, it does not indicate how biodegradable waste diverted from landfills should be managed. The WFD – with respect to the overall MSW – does not fill this gap; nonetheless, it prescribes that measures shall be taken by Member States (MS) to achieve environmentally sound waste management by following the so-called ‘waste hierarchy’ (art 4(1)). Such hierarchy considers waste prevention as the most environmentally sound option, while landfilling is considered the worst option. The WFD allows for deviations from the ‘waste hierarchy’ only if evidence based on life cycle thinking (LCT) shows that these deviations lead to a better overall environmental outcome.

In addition to designing strategies to reduce generation of food waste and ensure security of food supply (e.g. European Commission, 2013b), the European Commission (EC) is also committed to improving the sustainability of the management of the food waste that could not be prevented (e.g. European Commission, 2010b). In 2011, food wastage was identified as one of the main problems that needed to be addressed to increase resource efficiency (European Commission, 2011) and the EC invited all MS to explicitly consider food waste in their national waste prevention programmes (European Commission, 2008). On 2 December 2015, the EC’s circular economy package (European Commission, 2015a) was launched. In its action plan (European Commission, 2015b), it establishes a 50% reduction target of food waste generation by 2030, in line with the target set by the United Nation General Assembly as part of the 2030 Sustainable Development Goals.

**Food waste management options**

A number of well-established alternatives exist for the treatment of food waste as a part of the larger biodegradable waste stream. For instance, in their review, Bernstad and la Cour Jansen (2012) show that four technologies are the most common options: Landfilling, composting, anaerobic digestion and incineration. Other options are also covered in literature and seem capable of delivering an improved environmental performance, for example feed production from a specific food waste fraction (e.g. bread) (Vandermeersch et al., 2014), feed production using housefly larvae (van Zanten et al., 2015), gasification and pyrolysis (Ahmed and Gupta, 2010). These and other emerging options, however, are not considered in the numerical case study presented in this
article, since data availability and quality do not always compare with those of the well-established technologies.

**Materials and methods**

This section provides information on the approaches, tools and key assumptions upon which the methodology for sustainability assessment of food waste management options is built. It also presents key information regarding the numerical case study developed to provide an example of application of the proposed methodological framework.

**Overview**

In line with the prescription of the WFD (European Commission, 2008), the starting assumption of the framework methodology presented in this article is that the ‘waste hierarchy’ is an environmentally sound decision support principle. Deviations from such principles can, however, be accepted if justified by LCT.

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. Such methodology provides a structured decision support framework to help identify which alternatives for food waste management are optimal in the sense that they minimise environmental and economic impacts. The evaluation of environmental impacts is conducted based on life cycle assessment (LCA). The evaluation of economic impacts is conducted based on a life-cycle approach and also making use of the so-called ‘gate-fee’ for a straightforward estimation of the costs for waste treatment to be paid by, for example a municipality. Since elements of the ‘optimisation theory’ are also utilised, environmental impacts and economic impacts (i.e. costs) will be considered and referred to as ‘objective functions’.

**Evaluation of the environmental performance with LCA: Overview and key assumptions**

Among the existing LCT-based methods and standards, LCA is considered in the proposed methodological framework and associated numerical case study to evaluate the environmental performance. LCA – as defined by the ISO 14044 (2006) and further specified in the International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010c) – is a decision support tool widely used to evaluate the environmental impacts arising from any goods and service. LCA is also extensively applied to evaluate waste management systems, scenarios and strategies (e.g. Manfredi et al., 2009, 2010, 2011).

Evaluation of the environmental consequences arising from management of food waste is becoming increasingly common (e.g. Eriksson et al., 2015; Kim and Kim, 2010; Lundie and Peters, 2005). The factors and methodological choices that mostly influence the results of LCAs involving food waste are, in principle, the same as those of any LCA of waste management systems, for example choices made in the definition of the system boundary, choice of the impact assessment models and methods, choice of how to account for benefits of, for example energy recovery, recycling, carbon sequestration and many others. Such complexity leads to considerably different estimations of environmental impacts reported in reviews of LCA studies on food waste. As an example, the reviews conducted by Bernstad and la Cour Jansen (2011, 2012), report large variations of the estimated impacts on climate change arising from treatment of food waste, ranging from net impacts to net avoided impacts. For instance, anaerobic digestion of 1 t of food waste is reported in the range from −400 to +400 kg CO₂ eq.

Evaluation of the environmental impacts in the numerical case study presented in this article along with the methodological framework 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. The database version that was used in the numerical case study is the one made available in August 2015.

A number of alternative scenarios for food waste management are compared in the numerical case study based on management of 1 t of food waste, coherently with the selected LCA functional unit: Landfilling (L), composting (C), anaerobic digestion (AD) and incineration (I). In addition to the function(s) defined by the functional unit, some management scenarios provide a number of additional outputs/functions (e.g. compost, electricity, heat) that do not appear in all considered scenarios. In these cases, when comparing their performance, two alternative techniques can be applied: ‘system expansion’ or ‘substitution’ (European Commission, 2010c). Substitution – which is the approach used in the modelling example presented in this article – consists of accounting for the benefits arising from the displacement of a certain function/product with an equivalent function/product generated within the system modelled.

Also, it should be noted that while food waste can be treated separately in technologies, such as AD, in other treatment options food waste is typically treated together with different waste streams. This is for instance the case of incineration (where food waste is co-incinerated with other waste types), composting and landfilling. In all these cases, the allocation of environmental impacts to the chosen functional unit (e.g. 1 t of food waste) is not straightforward and require making assumptions and/or simplifications (see e.g. Allacker et al., 2014). Whether or not allocation choices have to be made also depends on how the LCA is conducted. EASETECH allows the modelling of a given technology assuming a single type of waste input (in this article, food waste). Therefore, in the numerical case study, emissions and impacts refer only to the chosen waste input, i.e. 1 t of food waste.
As emissions from landfills are not ‘virtually instantaneous’, but spread over long time periods, a time horizon of 100 years was set in EASETECH when modelling landfilling of food waste. This means that all emissions, from the moment food waste is landfilled (time ‘zero’) until 100 years into the future, are accounted for. Another aspect that needs to be addressed when landfilling is considered in LCA, is the way storage of carbon is accounted for. A certain amount of carbon will in fact remain stored in the landfill 100 years after disposal, mostly of biogenic origin in the case of food waste. In line with the recommendations provided by Christensen et al. (2009), in the numerical case study storage of biogenic carbon was accounted for as a net saving with respect to ‘climate change’, while storage of fossil carbon was accounted for as neutral.

The life cycle impact assessment calculations conducted for the case study are based on the EC Product Environmental Footprint method (European Commission, 2013a). Environmental impacts are estimated for 12 out of the 15 Product Environmental Footprint-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 calculated in a different way.

**Evaluation of the costs associated with food waste management**

As the LCA functional unit considered in the study is related to 1 t of food waste, the proposed methodological framework and the related numerical case study also assesses costs (both collection/transportation costs and treatment costs) relative to 1 t of food waste (i.e. Euro per tonne of food waste).

Evaluations of costs of any type of waste management options can be conducted in different ways.

- **Approach based on the consideration of the ‘actual costs’,** intended as a summation of costs relative to all processes included in the system boundary and necessary to provide the chosen functional unit. This approach is used in the numerical case study to assess the costs of food waste collection and transport.

- **Approach based on the consideration of the ‘gate-fee’,** intended as the charge paid by the authority (e.g. a local authority, such as a municipality) to a waste treatment operator to generate a stream of revenue (in this case the management/treatment of wastes). It usually offsets the capital costs, operation, maintenance, labour, along with profits and final disposal of any residue created in the process. This approach is used in the numerical case study to assess the costs of food waste treatment in the different scenarios.

The use of one approach over another first depends on the decision context. For instance, if a municipality is seeking to identify best performing management options, then using the gate-fee approach is particularly straightforward as gate-fees express the cost that a municipality has to pay to the company managing the waste. However, to apply this approach at a larger scale, e.g. at a European level, dedicated studies should, in principle, be undertaken to estimate and predict such gate-fees, which entails analyses of local market dynamics to determine the relationship between such gate-fees and prices. Gate-fees have a 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 costs remain the same (European Commission, 2001). Costs arising from legal requirements and obligations may also add to the overall cost of waste management; these are however not considered in this study.

**Optimisation and Pareto front**

The framework methodology proposed in this article makes use of multi-objective optimisation and Pareto optimality concepts (Grossmann and Guillén-Gosálbez, 2010). Multi-objective optimisation involves two or more objective functions to be optimised simultaneously. When these objective functions are incommensurable (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., 2003)), then a single solution that simultaneously optimises each objective does not exist, but there exist several optimal solutions. The solutions to the multi-objective problems are known as Pareto-optimal (also known as non-dominated or Pareto-efficient). They can be graphically represented in a Pareto front (when two objectives are considered) or in a Pareto surface (if the number of objectives is higher than two).

Pareto front consists of all solutions that are not dominated by any other solution (Pareto-optimal), i.e. solutions that are (a) not worse than the other solutions in all objective functions and (b) strictly better than the other solutions in at least one objective functions. In bi-objective optimisation, the Pareto front is also known as a trade-off curve. In fact, 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.

**Results and discussion: Framework methodology and numerical case study**

The methodology presented here provides a valuable tool to support science-based policy-making and decision-making in the context of food waste management. On the other hand, the numerical case study presented hereafter along the methodological steps should be intended as the application of such methodology under specific assumptions. Consequently, its results have a limited validity and should not be directly used to support waste management decisions, as their meaningfulness cannot be generalised nor transferred. The case study, in fact, is an application of the proposed methodology symbolising an average European situation (both technologically and geographically) in which support is needed to help decide how to manage food waste sustainably.
Furthermore, it should be noted that the purpose of this article is to develop a structured methodology for environmental/economic assessment and a numerical application of such methodology, not to analyse specific results in details. In light of all that, results from the case study are briefly presented hereafter, but not thoroughly discussed.

**Step 1: Identification of food waste management options**

A thorough analysis of the waste management context under consideration should be conducted to identify all treatment options for food waste management that are going to be evaluated. This, in principle, includes both the options that are already in place and those that could be installed. The numerical case study considers the options landfilling (L) composting (C), AD and incineration (I).

**Step 2: Definition of functional unit 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 should be defined to unambiguously identify – quantitatively and qualitatively – the functions and services provided by the scenarios considered. Definition of the functional unit also includes specification of the composition (e.g. chemical composition) of the waste input. Based on the chosen functional unit, the system boundaries are defined and include all relevant foreground and background processes that are needed to provide the services included in the functional unit. More information on how to define the LCA functional unit and system boundary are available, for example in ISO 14040 (2006) and European Commission (2010c, 2013a).

The functional unit chosen in the numerical case study is ‘management of 1 tonne of food waste – from collection to final disposal – composed of animal food waste (50% by mass) and vegetable food waste (50% by mass), including the impurities that come along these fractions’ (more detailed analysis of food waste fractions and compositions are available in literature (e.g. European Commission, 2014a)). An extract of the waste composition used is shown in Table 1. The scenarios considered in the case study are presented in Table 2, which also includes all the processes included in the boundary of the evaluation. All necessary input of electricity, fuels, materials, as well as waste collection and transport steps are also accounted for. The scenarios presented in Table 2 are considered under the assumptions that all treatment plants are already established and fully operative.

The collection step is considered as technology specific: The type of collection assumed for those treatment technologies that can treat food waste only is, in general, different from that assumed for those technologies that handle MSW (Table 3). Collection here is intended as taking place from the households to the ‘virtual’ point where the truck is fully loaded with waste. After this point, the transport stage starts. Concerning transport (intended as transport from the collection point to the treatment plant), for all scenarios it is assumed that the waste treatment plant is located 50 km from the collection point and that vehicles must also come back to such a point. Trucks are assumed to reach the treatment plant fully loaded and to return empty to the collection point. This is reflected in the costs for collection and transport assumed in the modelling (Table 5, detailed later).

**Step 3: Evaluation of the environmental performance**

Based on the identified waste management scenarios (Step 1), the chosen functional unit and all the processes included in the system boundaries (Step 2), the evaluation of the environmental performance can be conducted, for example via a dedicated LCA-based software tool.

To help display the results of the LCA, one or more indicator(s) have to be chosen as representative of the environmental performance of the waste management scenarios identified in ‘Step 1’. The choice of such indicator(s) has a key importance because it can directly influence the identification of the most sustainable management options. Possible choices include the following.

(a) To consider as many indicators 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 (see Step 4), each environmental impact category has to be considered individually (i.e. in isolation from the others).

(b) To consider as an indicator the total environmental impact (Total EI), intended as the sum of the individual impact scores after normalisation and weighting.

(c) To consider as an indicator the sum of the normalised and weighted impact scores of a sub-group of impact categories (potentially down to one individual impact category, e.g. climate change).

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**Table 1. Waste composition considered in the numerical case study (based on EASETECH software database).**

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</tr>
</thead>
<tbody>
<tr>
<td>Animal food waste</td>
<td>57.14</td>
<td>42.9</td>
<td>55.4</td>
<td>1.13</td>
<td>0.53</td>
<td>7</td>
<td>1</td>
<td>1.1E-5</td>
<td>1.2E-4</td>
<td>6E-4</td>
<td>2E-6</td>
<td>7E-6</td>
<td>4.9E-3</td>
</tr>
<tr>
<td>Vegetable food waste</td>
<td>76.99</td>
<td>23</td>
<td>47.5</td>
<td>0.24</td>
<td>1.27</td>
<td>1.9</td>
<td>0.23</td>
<td>9.5E-6</td>
<td>4.5E-4</td>
<td>1.2E-3</td>
<td>2E-6</td>
<td>1E-4</td>
<td>2.5E-3</td>
</tr>
</tbody>
</table>

TS: total solids.
Energy recovery efficiency of heat and electricity are set to 46% and 14% of the 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 stabilised and used as backfilling material displacing gravel. Fly ash output is assumed to be stabilised and sent to a mineral waste landfill.

The volatile solid, carbon and nitrogen degradation in the composting phase is equal to 74% and 65%, respectively. The methane content in the biogas is 63%.

Gas yield is assumed as 150 m³ of biogas (tonne of waste)^−1, with no leakage to the atmosphere. 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 fertiliser and substitutes a chemical fertiliser with substitution ratio for N, P and K, equal to 0.4, 1 and 1, respectively.

Electricity consumption in the composting phase is 0.035 kWh (tonne wet waste)^−1. Compost is used directly on land as a fertiliser and substitutes a chemical fertiliser with a substitution ratio for N, P and K equal to 0.4, 1 and 1, respectively.

Electricity consumption in the composting phase is 0.02 kWh (tonne wet waste)^−1. Gas yield is assumed as 150 m³ of biogas (tonne of waste)^−1, with no leakage to the atmosphere. The methane content in the biogas is 63%.

Gas yield is assumed as 150 m³ of biogas (tonne of waste)^−1, with no leakage to the atmosphere. 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 fertiliser and substitutes a chemical fertiliser with substitution ratio for N, P and K, equal to 0.4, 1 and 1, respectively.

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 subsequently be compared with the indicator representing the economic performance. This is the approach adopted in the numerical case study. Choosing (b), i.e. calculating a single impact score based on the impacts scores...
estimated for all impact categories, implies that the ‘importance’ given to each impact category is proportional to the weighting factor used. 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 in the evaluation context.

The environmental impacts estimated in the numerical case study are shown in Table 4. Figures 1 to 3 also uses some of these results in combination with the cost-related indicators.

**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. All relevant parameters considered in the economic evaluation of the case study are reported in Table 5, including subsidies and taxes. Three different indicators were ultimately selected to express the economic performance of each scenario.

- The ‘cost for the authority (e.g. a municipality) for waste treatment’. Adopting the approach from other studies (e.g. COWI, 2004; European Commission, 2010b), the treatment costs can in this case be represented using as a proxy for the gate-fee, i.e. the cost that the authority pays for delivering waste to that plant. 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 owing to the temporary market condition (e.g. over-capacity for incineration), length of contracts, or direct or indirect subsidies.
- The ‘cost for society for waste treatment’. This is obtained by adding any existing subsidies and subtracting any existing tax from the previously identified gate-fees. This allows the estimation of a sort of ‘raw gate-fee’, which is closer to the sum of capital expenditures and operational costs.
- The ‘cost for society for waste management’. This is estimated by adding the costs of waste collection and transport to the previously estimated ‘cost for society for waste treatment’.

Gate-fees have been estimated considering average plant capacity for waste treatment, and average efficiency. However, it must be highlighted that there is large variability in gate-fees; this is particularly true for incinerators, where gate-fees for a small plant (e.g. <200,000 tay−1) are typically significantly higher than those of larger plants.

**Step 5: Identification of Pareto-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, a graph showing environmental impacts versus economic impacts can be plotted. This allows visualisation of the objective space and plotting of the so-called ‘Pareto front’, and thus able to identify the Pareto-optimal scenarios, i.e. those that minimise both objective functions at the same time.

Figures 1, 2 and 3 provide samples of the results of the numerical case study. Figure 1 shows results with respect to climate change versus the cost for the authority for waste treatment (i.e. the gate-fees). Under this choice of environmental and economic indicators, only incineration and AD appear as Pareto-optimal scenarios, in the sense that these scenarios are not dominated by the others, i.e. they are (a) not worse than all (non-optimal) scenarios with respect to both environmental and economic performance, and (b) are strictly better than all (non-optimal) scenarios in at least the environmental or economic performance. On the other hand, when considering climate change in combination with the cost for society for waste treatment and for waste management (Figures 2 and 3, respectively), all technologies appear as Pareto-optimal, except AD+C. These results highlight the considerable influence of any existing subsidies and taxes on the overall performance.

In the specific case of the cost for society for waste treatment (Figure 2), the Pareto front shows a steep slope between C and Le. This could make decision makers think that C is a better option than Le because a little increase in cost (i.e. 3 Eurot−1) can significantly reduce the environmental impact. This difference cannot be equally well appreciated in the cost for society for waste management (Figure 3), where the same environmental trade-off is seen as the cost of 44 Eurot−1.

Results from the calculation of all 36 possible Pareto-fronts (Figure 4) (arising from the combination of 12 environmental impact categories with three economic indicators), show that both incineration and landfilling with flares appear as Pareto-optimal in 27 combinations, followed by AD and landfilling with electricity production (20 and 17 times, respectively).

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**Table 3.** Collection schemes considered in the numerical case study (based on expert judgment and the EASETECH database).

<table>
<thead>
<tr>
<th>Collection scheme</th>
<th>Scenarios</th>
<th>Type of truck</th>
<th>Emission standard</th>
<th>Fuel consumption rate [l diesel tonne of waste−1]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Only food waste</td>
<td>AD, C, AD+C</td>
<td>10 t</td>
<td>Euro3</td>
<td>6</td>
</tr>
<tr>
<td>Municipal solid waste</td>
<td>L, Le, Lf</td>
<td>25 t</td>
<td>Euro3</td>
<td>7.5</td>
</tr>
</tbody>
</table>

1 l diesel tonne of waste−1: liters of diesel per tonne of waste.
Step 6: Uncertainty and sensitivity analysis

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). Uncertainties may arise owing to a variety of aspects, such as data variability, erroneous measurements, wrong estimations, unrepresentative or missing data and modelling assumptions (Clavreul et al., 2012).

In the case study, only sensitivity analysis is performed through ‘scenario analysis’, where the values of selected parameters are changed one at a time. Table 6 shows results of the sensitivity analysis for the AD scenario, which was taken as the 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 the example). Then, four different parameters were selected for the sensitivity analysis.

In order to highlight how changes in the values of the selected parameters can affect the selection of optimal technologies in the Pareto front analysis, Figure 5 displays the example of climate change against the cost for society for waste treatment. As shown, AD is found to be a Pareto-optimal solution in all of the sub-scenarios. However, depending on the specific sub-scenario considered, AD+C can also become Pareto-optimal. For the base scenario of AD, AD+C was the only scenario not classifying as Pareto-optimal. On the other hand, when selecting AD/2 and AD/5, AD+C becomes optimal. For the rest of the AD scenarios, AD+C will remain non-optimal.

Conclusions and remarks

While substantial information on European food waste generation and management routes exist, information on the sustainability of food waste management options is somewhat inaccurate. In addition, though a number of tools are available that provide assessment of the performance of waste management options, these 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 such gaps, the work presented in this article provides a life-cycle-based methodology to assess environmental and economic sustainability of food waste management, as well as a simplified numerical case study. Such methodology is able to accommodate additional assessment criteria, for example the social dimension of sustainability. It thus moves towards a comprehensive sustainability assessment framework. As demonstrated in the numerical case study, application of the proposed methodology is, in principle, relatively straightforward. However, 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 aspects considered hereafter are considered to be crucial.

Table 4. Environmental impacts from management of 1t of food waste in the numerical case study.

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Acronym</th>
<th>Units</th>
<th>AD</th>
<th>C</th>
<th>AD+C</th>
<th>I</th>
<th>Lf</th>
</tr>
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<tbody>
<tr>
<td>Climate change</td>
<td>CC</td>
<td>kg CO₂ eq.</td>
<td>−7.6×10⁻²</td>
<td>1.03×10⁻¹</td>
<td>1.03×10⁻¹</td>
<td>0.61</td>
<td>1.03×10⁻¹</td>
</tr>
<tr>
<td>Ozone depletion</td>
<td>OD</td>
<td>kg CFC-11 eq.</td>
<td>−7.4×10⁻²</td>
<td>1.6×10⁻²</td>
<td>1.6×10⁻²</td>
<td>0.61</td>
<td>1.6×10⁻²</td>
</tr>
<tr>
<td>Human toxicity, cancer effects</td>
<td>HTH,ce</td>
<td>kg NH₃ eq.</td>
<td>−7.6×10⁻²</td>
<td>1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>0.61</td>
<td>1.0×10⁻²</td>
</tr>
<tr>
<td>Human toxicity, non-cancer effects</td>
<td>HTH,nce</td>
<td>kg NH₃ eq.</td>
<td>−7.6×10⁻²</td>
<td>1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>0.61</td>
<td>1.0×10⁻²</td>
</tr>
<tr>
<td>Ionising radiations, human health effects</td>
<td>IR,hh</td>
<td>kg U₂³⁵ eq. (to air)</td>
<td>−0.34</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
</tr>
<tr>
<td>Photochemical ozone formation</td>
<td>POF</td>
<td>kg NMVOC eq.</td>
<td>−0.34</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
</tr>
<tr>
<td>Freshwater eutrophication</td>
<td>FE</td>
<td>kg N eq.</td>
<td>−1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>0.61</td>
<td>1.0×10⁻²</td>
</tr>
<tr>
<td>Marine eutrophication</td>
<td>ME</td>
<td>kg N eq.</td>
<td>−1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>1.0×10⁻²</td>
<td>0.61</td>
<td>1.0×10⁻²</td>
</tr>
<tr>
<td>Ecotoxicity or aquatic freshwater</td>
<td>Fecotox</td>
<td>kg eq.</td>
<td>−0.34</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
</tr>
<tr>
<td>Acidification</td>
<td>A</td>
<td>mol H⁺ eq.</td>
<td>−0.34</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
</tr>
<tr>
<td>Terrestrial eutrophication</td>
<td>TE</td>
<td>kg PM₂.₅ eq.</td>
<td>−0.34</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
<td>0.15</td>
</tr>
</tbody>
</table>

aComparative toxic unit for humans.
bComparative toxic unit for ecosystems.
Table 5. Estimation of the economic performance in the numerical case study, as Euros tonne\(^{-1}\) of food waste (based on Confederation of European Waste-to-Energy Plants, 2008; COWI, 2004; European Commission, 2010b; WRAP, 2015).

<table>
<thead>
<tr>
<th>Scenarios</th>
<th>Subsidies</th>
<th>Taxes</th>
<th>Cost of collection and transport</th>
<th>1st cost indicator</th>
<th>2nd cost indicator</th>
<th>3rd cost indicator</th>
</tr>
</thead>
<tbody>
<tr>
<td>AD</td>
<td>33</td>
<td>0</td>
<td>105</td>
<td>55</td>
<td>88</td>
<td>193</td>
</tr>
<tr>
<td>C</td>
<td>0</td>
<td>0</td>
<td>105</td>
<td>65</td>
<td>65</td>
<td>170</td>
</tr>
<tr>
<td>AD+C</td>
<td>33</td>
<td>0</td>
<td>105</td>
<td>80</td>
<td>113</td>
<td>218</td>
</tr>
<tr>
<td>I</td>
<td>45</td>
<td>0</td>
<td>64</td>
<td>100</td>
<td>144</td>
<td>208</td>
</tr>
<tr>
<td>Le</td>
<td>12</td>
<td>30</td>
<td>64</td>
<td>80</td>
<td>62</td>
<td>126</td>
</tr>
<tr>
<td>Lf</td>
<td>0</td>
<td>30</td>
<td>64</td>
<td>80</td>
<td>50</td>
<td>114</td>
</tr>
</tbody>
</table>

Figure 1. Numerical case study: Climate change impact vs cost for the authority for waste treatment.

Figure 2. Numerical case study: Climate change impact vs cost for society for waste treatment.

Figure 3. Numerical case study: Climate change impact vs cost for society for waste management.

Figure 4. Number of possible combinations (considering the combination of 12 environmental impact categories and three economic indicators) in which each scenario appears as Pareto-optimal.

With respect to the identification of alternative food waste management options (Step 1), while some may already be available in the waste management context selected, others may not be available (e.g. a waste incineration plant is not present, or is very far) but could equally be considered. This will influence all subsequent steps, in that, for example 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.

The choice of the environmental and economic indicators to express the sustainability performance (Steps 3 and 4, respectively) is critical. Such choice, in fact, determines the way results are displayed, 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...
be considered when displaying the results of the assessment. In the evaluation of the economic performance (Step 4), the choice of the economic indicator depends on the decision context. Using gate-fees can help provide straightforward estimations of costs.

LCAs that include food waste management may become particularly complex as, in addition to technical processes, biological processes also 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. 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. For the time being, the proposed methodology only partially accounts for such factors (e.g. it distinguishes between biogenic and fossil carbon and accounts for carbon sequestration), thus room for improvements exist.

The consideration of uncertainty and sensitivity (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 as shown in the case study. The authors of this article are currently involved in expanding the proposed methodological framework to make it capable of the following.

- Assess the social dimension of sustainability, so that a comprehensive sustainability assessment framework is developed.
- Analyse the identified solutions (both the optimal and the non-optimal) towards improving their sustainability performance. For instance, with respect to the environmental objective function, environmental efficiency analysis can be performed via data envelopment analysis in order to set targets for the non-optimal management options that could, if achieved, make them optimal.
- Identify targets for prevention of food waste along the food supply chain and assess the sustainability implications of achieving such targets.

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**Table 6.** Sensitivity analysis for the AD scenario in the numerical case study [new values of parameters are chosen based on expert judgment].

<table>
<thead>
<tr>
<th>Scenario/sub-scenario</th>
<th>Process</th>
<th>Parameter</th>
<th>Default value in the case study (%)</th>
<th>New value in the case study (%)</th>
<th>Impact value [kg CO₂ eq.]</th>
</tr>
</thead>
<tbody>
<tr>
<td>AD</td>
<td>Base scenario</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>−7.6 × 10⁻²</td>
</tr>
<tr>
<td>AD/1</td>
<td>Electricity substitution</td>
<td>Efficiency</td>
<td>0.25</td>
<td>0.4</td>
<td>−88.4</td>
</tr>
<tr>
<td>AD/2</td>
<td>Use on land of the digestate</td>
<td>Distribution of carbon − C [soil storage]</td>
<td>13.2</td>
<td>9</td>
<td>14.22</td>
</tr>
<tr>
<td>AD/3</td>
<td>Use on land of the digestate</td>
<td>Distribution of carbon − C [soil storage]</td>
<td>13.2</td>
<td>14</td>
<td>−2.776</td>
</tr>
<tr>
<td>AD/4</td>
<td>Use on land of the digestate</td>
<td>Distribution of nitrogen − N₂O [air]</td>
<td>2.78</td>
<td>1</td>
<td>−1.43 × 10²</td>
</tr>
<tr>
<td>AD/5</td>
<td>Fertiliser substitution</td>
<td>Average N fertiliser</td>
<td>0.4</td>
<td>0.2</td>
<td>52.7</td>
</tr>
<tr>
<td>AD/6</td>
<td>Fertiliser substitution</td>
<td>Average N fertiliser</td>
<td>0.4</td>
<td>0.6</td>
<td>−52.78</td>
</tr>
</tbody>
</table>

**Figure 5.** Numerical case study: Example of sensitivity analysis for the modelled scenarios.

Note: The upper-right box is a zoom of the bottom-left box.
Damgaard from the Department of Environmental Engineering of the Technical University of Denmark (DTU-Environment); Cristina Torres de Matos, Fabrice Mathieux and Cynthia E. L. Latunussa from the Institute for Environment and Sustainability (IES) of the Directorate General Joint Research Centre (DG JRC); Michele Giavini from ‘ARS Ambiente’.

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