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Life cycle costing of food waste: A review of methodological approaches

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A B S T R A C T

Food waste (FW) is a global problem that is receiving increasing attention due to its environmental and economic impacts. Appropriate FW prevention, valorization, and management routes could mitigate or avoid these effects. Life cycle thinking and approaches, such as life cycle costing (LCC), may represent suitable tools to assess the sustainability of these routes. This study analyzes different LCC methodological aspects and approaches to evaluate FW management and valorization routes.

A systematic literature review was carried out with a focus on different LCC approaches, their application to food, FW, and waste systems, as well as on specific methodological aspects. The review consisted of three phases: a collection phase, an iterative phase with experts' consultation, and a final literature classification. Journal papers and reports were retrieved from selected databases and search engines.

The standardization of LCC methodologies is still in its infancy due to a lack of consensus over definitions and approaches. Research on the life cycle cost of FW is limited and generally focused on FW management, rather than prevention or valorization of specific flows. FW prevention, valorization, and management require a consistent integration of LCC and Life Cycle Assessment (LCA) to avoid tradeoffs between environmental and economic impacts. This entails a proper investigation of methodological differences between attributional and consequential modelling in LCC, especially with regard to functional unit, system boundaries, multi-functionality, included cost, and assessed impacts. Further efforts could also aim at finding the most effective and transparent categorization of costs, in particular when dealing with multiple stakeholders sustaining costs of FW. Interpretation of results from LCC of FW should take into account the effect on larger economic systems. Additional key performance indicators and analytical tools could be included in consequential approaches.

Contents

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1. Introduction

World population might reach 9 billion by 2050 (UN, 2015). The resulting demand for food, feed, and energy will pose an unprecedented pressure on natural resources. However, 795 million people suffer from hunger (FAO et al., 2015) while food systems are inefficiently wasting about one-third of all edible food each year (FAO, 2014). This waste, which has an estimated cost of US$940 billion (FAO, 2014), could be at least partially avoided, as it could be still suitable for human consumption or other uses (Gustavsson et al., 2011).

Food waste (FW) reduction is a priority in the challenge to achieve sustainability in the use of natural resources, such as water and energy (Cuéllar and Webber, 2010; De Laurentiis et al., 2016; FAO, 2013; Kummu et al., 2012; Vittuari et al., 2016), and avoid the resultant loss of money. Global policies are pushing towards the adoption of FW prevention, valorization, and management strategies. The United Nations Sustainable Development Goal 12.3 aims to halve per capita FW, as well as its management costs, while maximizing the value of un-avoidable FW and packaging materials by 2030 (Scherhaufer et al., 2015). The European Commission has the ambition, within the Circular Economy package, to introduce similar measures covering the whole product cycle (European Commission, 2015a).

Since FW prevention, valorization, and management represent priorities for the agenda of national governments and international organizations, their environmental and economic sustainability should be properly assessed. Life cycle approaches represents tools for the evaluation of both the economic and environmental impacts of FW prevention, valorization, and management. Life Cycle Assessment (LCA) is already established in studying environmental impacts of FW (Corrado et al., 2017; Gruber et al., 2016; Notarnicola et al., 2016; Unger et al., 2016). Life cycle costing (LCC) has been only recently applied to assess economic costs of FW management. However, calculating the cost of products and services in a life span or life cycle perspective is a rather longstanding idea. Unlike LCA (ISO, 2006a,b), LCC has been standardized for specific product categories, such as petroleum and natural gas (ISO, 2008, 2000). Hunkeler et al. (2008) provided a classification of LCC into three main approaches: Conventional (C-LCC), Environmental (E-LCC) and Societal Life Cycle Costing (S-LCC) – mainly differing in terms of perspective, costs included, and potential uses. Recent literature (Ciroth et al., 2011; Heijungs et al., 2013; Martinez-Sanchez et al., 2015; Swarr et al., 2011) also adopted such classification.

The term "Conventional" refers to the long established traditional approach of LCC, originated in the 1930s, when the US General Accounting Office included operating and maintenance costs in public procurement (Gluch and Baumann, 2004; Korpi et al., 2008; Woodward, 1997). Most C-LCC have a single stakeholder (producer or consumer) perspective and assess decisions over products or investments requiring high initial capital (Dhillon, 2010). More recently, a specific Society of Environmental Toxicology and Chemistry (SETAC) working group developed E-LCC, aiming at the integration of cost assessment within LCA (Heijungs et al., 2013). The goal is to assess costs occurred during the life cycle of products, services, and technologies (Hunkeler et al., 2008), focusing on the *life cycle* in its LCA-related meaning, rather than the product, service or investment *life span*. E-LCC should have the same product system, functional unit, and system boundaries as LCA, defined by International Organization for Standardization (ISO) standards 14,040/44. E-LCC can entail a multiple stakeholders' perspectives and it can consider externalities that will be potentially internalized (Hunkeler et al., 2008; Swarr et al., 2011). S-LCC further enlarges the boundaries of the analysis by assessing the overall direct and indirect costs covered by the society in a larger perspective (Hunkeler et al., 2008; Petti et al., 2016).

The use of LCC approaches for food products or FW streams is limited and no common methodological approach exists in the literature, especially when integrated with a LCA. Examples of this diversity can be found in Kim et al. (2011), Martinez-Sanchez et al. (2016), and Takata et al. (2012). Thus, the aim of this study is to analyze different LCC methodological aspects and approaches on FW management and valorization routes, through a literature review. The analysis of current methodological practices should serve to identify a list of critical areas needing further systematization and research. Ultimately, results from this paper might foster the development of a coherent modelling of FW prevention, valorization, and management, and an applied analytical framework, so to promote the utilization of LCC by relevant stakeholders and policy-makers.

2. Materials and methods

A literature review of recent scientific articles and reports was carried out to analyze methodological aspects related to life cycle costing of FW, its impact on food systems, and potential prevention, disposal, management, and valorization routes. The design of the review consisted of three phases: (1) a collection phase, (2) an iterative phase with consultation of experts, and (3) the final literature analysis based on selected methodological aspects.

After defining aims and subject of the review, the following keywords were identified: "LCC/life cycle costing" and "food waste". Relevant databases (Scopus, Web of Science, and Google Scholar) were used to retrieve a preliminary list of documents. A brief analysis of abstracts was carried out to exclude non-pertinent sources, such as general reviews or papers adopting exclusively LCA. Only applied LCC studies in relation to FW were selected. Policy guidelines and standards were not included for the review because they did not provide guidance or example of application to FW. Indeed, standards (ASTM International, 2013, ASTM International, 2011, ISO, 2008, 2001a,b, 2000) referred to the application of C-LCC to buildings, petroleum industries and assets. After this first search, a limited amount of literature on FW emerged since only 10 documents out of 35 search results were considered coherent with the research objectives.

Additional keywords ("waste" and "food") were then applied to a second database search and complementary LCC literature was retrieved, also including some non-LCC studies. Experts[1] with competence on life cycle thinking and FW were consulted for a first round to provide further inputs and supplementary sources. In particular, 11 LCC food studies, 6 LCC waste studies, and 6 non-LCC FW studies were added. The final inventory comprised 33 sources, 31 of which were published in peer reviewed journals 2 were scientific reports. Most of the reviewed sources (27) used LCC while other 6

[1] Experts were part of the REFRESH (Resource Efficient Food and dRink for Entire Supply cHain) project, funded by the Horizon 2020 Framework Programme of the European Union http://eu-refresh.org/.

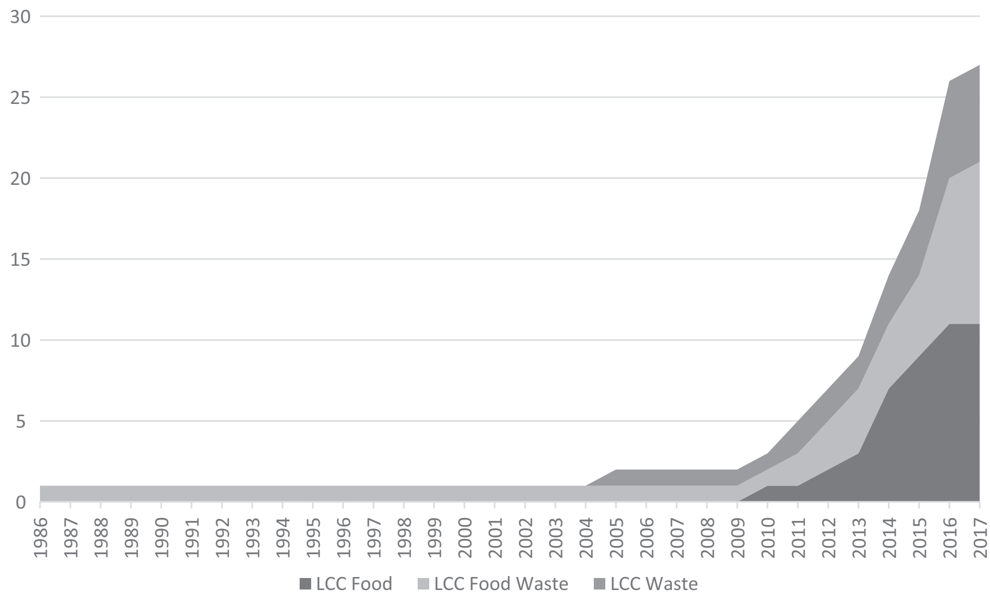

Fig. 1. Cumulative number of reviewed studies by topic (Note: non-LCC references excluded).

sources applied analogous methodologies (no LCC) but with a strong focus on FW. Results from these two sets of sources were presented and discussed separately.

Except for two papers dating back respectively to 1986 and 2005, all documents were published in the 2010–2017 period (Fig. 1).

In the final phase, we drafted a list of relevant methodological aspects based on a preliminary analysis of the inventory. A second round of experts' consultation was deemed useful to review the selected criteria and to provide additional elements for the categorization and discussion of results. Since most of the reviewed studies had a life cycle costing approach and several of them presented a parallel or previous LCA, the analyzed methodological aspects included also typical features of LCA. In particular, the list included:

- LCC approaches (C-LCC, E-LCC, S-LCC) and topic (food, FW, and waste management): reviewed studies were divided by topic addressed and LCC approach adopted. In case authors did not report the specific approach, a decision was made based on the presence of a parallel or previous LCA, the consistency of functional unit, system boundaries, inventory, joint analysis with LCA, and the monetization of externalities.
- Functional units: studies were classified according to the typology of functional unit; in case explicit information was not provided, a reference flow was identified and categorized.
- System boundaries and multi-functionality: this aspect included to the perspective of the study (i.e. cradle to gate/-grave), the type of modelling (attributional vs. consequential), and related approach to multi-output processes.
- Cut-off: similar to LCA, in LCC it was intended as the level of significance that a cost should have to be included in a study (Hunkeler et al., 2008); studies were reviewed to identify a possible typology.
- Cost modelling: costing systems used by authors were classified by typology of cost categorization, number of cost bearers, indirect cost allocation, and discounting.
- Externalities: they were intended as quantifiable costs or benefits on external stakeholders (Hunkeler et al., 2008); studies were analyzed to assess the inclusion of externalities and the related approaches.
- Impact assessment and results interpretation: this aspect included the type of economic assessment methods, joint evaluation with LCA, and techniques for the interpretation of results.

3. Results and discussion

3.1. LCC approaches and application to food, food waste and waste management

Table 1 summarizes the analyzed LCC publications according to the costing approach and the general topic addressed, and information regarding the presence of a previous or parallel LCA, functional unit(s), and system boundaries perspective adopted by each study. Full details on each methodological aspect per each paper can be found in Supplementary Materials. This selection included 27 journal papers, while the remaining 6 sources were separately assessed.

Most (19) of the reviewed studies, especially waste and FW studies, could be classified as E-LCC approaches, while C-LCC was largely used in food studies. Four cases included a societal perspective. LCC approaches were not mutually exclusive but few studies explicitly combined them. In general, distinction between approaches was blurred in reviewed literature. Authors generically identified their approach as LCC, while those providing a specification were usually using very different definitions. For example, Reich (2005) defined E-LCC as a weighting tool for LCA that can provide a welfare impact assessment if combined with a "financial LCC". However, the latter was jointly designed and used with an LCA, as in the E-LCC described by Hunkeler et al. (2008), while merged LCCs (financial plus environmental) could be regarded as a sort of S-LCC. Therefore, this study and others referring to the same approach were considered as E-LCCs. Willersinn et al. (2017) used a full-cost calculation similar to an E-LCC even though it was not defined as such. Likewise, the integration with a previous or parallel LCA (23 papers) was not a distinctive feature of Environmental or S-LCCs, since also 6C-LCCs were accompanied by an LCA. With the exception of Martinez-Sanchez et al. (2016) and Brandão et al. (2010), none of reviewed studies specifically mentioned the LCA approach adopted, whether consequential or attributional.

Table 1
Overview of reviewed publications on food, food waste, and waste management (Note: non-LCC references excluded).

Topic	Source	Approach			Parallel or previous LCA	Functional unit			System boundaries perspective	
		C-LCC	E-LCC	S-LCC		Mass	Area	Other	Cradle(Bin)-to-gate	Cradle(Bin)-to-grave
Food	Mohamad et al. (2014)	√			√		√		√	
	Brandão et al. (2010)	√			√		√		√	
	Pergola et al. (2013)		√		√	√	√		√	
	Iotti and Bonazzi (2014)	√				√			√	
	Falcone et al. (2015)	√			√		√		√	
	De Gennaro et al. (2012)		√		√		√		√	
	De Luca et al. (2014)		√		√		√		√	
	Fenollosa et al. (2014)		√		√	√	√		√	
	Tamburini et al. (2015)		√		√	√	√		√	
	Amienyo and Azapagic (2016)		√		√			√		√
	Schmidt Rivera et al. (2014)		√		√			√		√
Food waste	Sargent et al. (1986)	√						√	√	
	Christoforou et al. (2016)	√				√			√	
	Kim et al. (2011)		√		√	√			√	√
	Takata et al. (2012)		√		√	√			√	
	Vinyes et al. (2012)		√		√	√			√	
	Escobar Lanzuela et al. (2015)		√		√	√			√	
	Daylan and Ciliz (2016)		√		√			√		√
	Bong et al. (2016)		√		√			√		√
	Martinez-Sanchez et al. (2016)		√	√	√	√			√	√
	Willersinn et al. (2017)		√		√	√				√
Waste	Reich (2005)		√		√			√	√	√
	Massarutto et al. (2011)			√		√			√	√
	Asselin-Balençon and Jolliet (2014)		√		√	√		√	√	
	Martinez-Sanchez et al. (2015)	√	√	√	√	√			√	√
	Woon and Lo (2016)			√	√	√			√	
	Rigamonti et al. (2016)	√	√		√	√			√	√

Regarding topics, 11 studies dealt with food, 10 with FW and 6 with waste. Most FW studies focused on the analysis of urban FW management, mainly but not exclusively deriving from the consumption segment (Bong et al., 2016; Escobar Lanzuela et al., 2015; Kim et al., 2011; Martinez-Sanchez et al., 2016; Takata et al., 2012). Only Martinez-Sanchez et al. (2016) and Willersinn et al. (2017) included an impact assessment of FW prevention. The latter focused exclusively on loss reduction in the Swiss potato supply chain and for that reason it was included among FW studies, despite having the main product as functional unit. Other studies focused on specific FW flows and their valorization (Christoforou et al., 2016; Daylan and Ciliz, 2016; Sargent et al., 1986; Vinyes et al., 2012). Similarly, waste studies assessed municipal solid waste (MSW) management and only one included agricultural byproducts (Asselin-Balençon and Jolliet, 2014).

Food LCCs typically focused on agricultural productions (Brandão et al., 2010; De Luca et al., 2014; Falcone et al., 2015; Fenollosa et al., 2014; Mohamad et al., 2014; Pergola et al., 2013; Tamburini et al., 2015) and 3 studies analyzed processed food systems (Amienyo and Azapagic, 2016; Iotti and Bonazzi, 2014; Schmidt Rivera et al., 2014). Only in a study on convenience food by Schmidt Rivera et al. (2014), did the authors explicitly address and model food losses, waste, and byproducts from each stage of the supply chain. However, they included these flows for waste management costs in each scenario without carrying out a specific analysis or discussion.

3.2. Functional unit

Functional unit (FUs) is a key methodological aspect in the analysis of life cycle methodologies. No explicit mention to FU is foreseen in C-LCC or other costing approaches, while FU should instead be consistent with provisions of ISO 14,040/44 when LCA and E-LCC are conducted together (Hunkeler et al., 2008; Swarr et al., 2011).

In almost all reviewed papers, one or more FUs were explicitly defined. In two cases it was anyway possible to identify a reference flow (Iotti and Bonazzi, 2014; Sargent et al., 1986). Food studies focused on agricultural production and land management. Area-based FUs were generally used to assess the financial viability of cultivations (e.g. orchards), either alone or in comparison with mass-based FUs to account for yield differences. In those papers studying the whole supply chain of processed products, a specific FU related to the end product was used (Amienyo and Azapagic, 2016; Schmidt Rivera et al., 2014). Considering that FW is more linked to the product(s) than land management, a mass-based FU seems more appropriate for FW modelling in food LCCs.

Most FW and waste studies analyzed integrated or specific management and disposal options. FUs were typically related to the mass of FW treated or valorized, although with different scales. For example, Kim et al. (2011) studied 1 ton of FW managed in different treatment scenarios, Rigamonti et al. (2016), Martinez-Sanchez et al. (2015), and Woon and Lo (2016) analyzed environmental and economic impacts related to 1 ton of MSW. Escobar Lanzuela et al. (2015) referred to the yearly average organic waste from restaurants and catering per person. Martinez-Sanchez et al. (2016) analyzed instead the management of the yearly Danish household FW. Besides the obvious difference between results, small and large scale also imply different perspectives, datasets, and methodological choices, especially when adopting a consequential approach (e.g. relevance of infrastructures, magnitude of changes, influence on markets). Despite their relevance, also detailed aspects of described FU – such as specification of weight (dry or wet), waste fractions, chemical and/or nutritional characterization, etc. – were provided only in some studies. For specific waste flows, Sargent et al. (1986) detailed the moisture and heat content of apple pomace and Christoforou et al. (2016) carried out a proximate and ultimate analysis of olive mill residues. For generic food waste and waste, Martinez-Sanchez et al. (2016, 2015) reported the composition of waste, while Takata et al. (2012) only mentioned that FW was expressed in wet weight. Providing these details is crucial to understand authors' assumptions, for example on substitute products, and for the comparability of results from similar valorization options. Another feature that was not always provided in the literature was the specific reference to generated, collected or treated FW, meaning the point of origin of FW. In fact, different FW collection systems (e.g. used cooking oil collection) might present different efficiencies, resulting in higher or lower amounts of waste treated, but with the same function (Vinyes et al., 2012). As mentioned, Willersinn et al. (2017) were the only authors focusing on a food supply chain (potatoes) and using consumed food as functional unit to analyze different measures for the reduction of losses.

Those studies focusing on valorized products (such as fuels and electricity) rather than treated FW allocated all impacts to other types of FUs, such as 1 km of driving distance with ethanol from lignocellulosic byproducts (Daylan and Ciliz, 2016) and 1 kW-h of electricity produced from anaerobic digestion (Asselin-Balençon and Jolliet, 2014). Finally, some studies (Bong et al., 2016; Reich, 2005; Sargent et al., 1986) focused on the operation of a plant for a certain period of time, respectively of 1 month, 1 year, and the whole life time. Also in these cases, the amount of FW used as input and its characterization are essential to ensure comparability.

3.3. System boundaries and multi-functionality

A cradle-to-gate perspective (i.e., excluding consumption and final disposal from the system boundaries) was used in almost all food studies with 2 exceptions focusing on the whole supply chain of processed products (Amienyo and Azapagic, 2016; Schmidt Rivera et al., 2014). From a FW perspective, this latter option has the undoubtable advantage of a more complete modelling and quantification, considering the potential shift of FW and related costs from one stage to another, which might result from certain measures.

Due to the characteristics of FW and waste treatment, which might include disposal and valorization of incoming materials within the same system or scenarios, system boundaries included both cradle/bin to grave (final disposal) and cradle/bin to gate (valorization) perspectives. In this case, use and disposal phases were not included in the analyzed systems, not being part of their function.

Several studies referred to a previous LCA or presented a parallel study. Only two studies – Martinez-Sanchez et al. (2016) and Brandão et al. (2010) – explicitly specified the typology of modelling, respectively consequential and attributional. These two types of LCA modelling differ in goal, scope, boundaries, data, and indicators. While attributional LCA aims at providing an impact assessment of a system under static and average conditions, consequential LCA focuses on all the consequences deriving from a decision or a change in the system, including external effects (Weidema, 2000; Weidema and Schmidt, 2010). In the other studies, authors did not address this issue, by discussing methodological differences in terms of LCC approach. Even in comparative analyses focused on potential differences between scenarios, it was possible to identify an attributional perspective. This aspect reinforces the argument for further research on an improved methodological definition of attributional and consequential LCC approaches and their consistency with the corresponding LCA, as also suggested by recent academic debate (Hannouf and Assefa, 2016; Klöpffer and Ciroth, 2011; Wood and Hertwich, 2013). Eventually, a process of standardization of E-LCC could address this issue and differentiate between approaches.

A potential difference between attributional and consequential E-LCC could regard the methodological approach to multifunctional processes. Life cycle costing can deal with multifunctional issues similar to LCA, by partitioning processes and costs among different products and by allocating shared or indirect costs through several criteria. When in the parallel LCA a system expansion is adopted – as in the case of consequential modelling – also E-LCC boundaries could include the substitution of products deriving from coproducts (Hunkeler et al., 2008).

Joint productions, byproduct valorization, or recycling often characterized food and FW product system, and several studies (11) among the reviewed literature dealt with multi-functionality. All of them used SE regardless of the modelling framework. Among food case studies, only Schmidt Rivera et al. (2014) included revenues from processing byproducts. Contrarily, most FW studies and almost all waste studies presented a system expansion approach to deal with coproducts from management or valorization processes. In LCA with system expansion, product substitution – taking into account substitute product(s) as possibly avoided impact – is standard practice. Similarly, in LCCs corresponding revenues from co-products were considered as negative (avoided) costs in all cases, with two exceptions: the inclusion of average market prices of byproducts as measure of benefits (Kim et al., 2011) and the translation of revenues into positive financial costs for the external system (Reich, 2005). However, although this approach is consistent with LCA, literature did not provide insights on several issues needing clarification. First, the inclusion of revenues as avoided costs is not entirely coherent with the LCC framework when more stages (and cost bearers) are included, as they are transfers between stakeholders. A calculation of the full LCC for the whole supply chain should exclude these money flows. Instead, they could be accounted for in the assessment of cost and benefits distribution among multiple cost bearers as well as in the estimation of value added. Second, using revenues or market prices as avoided cost should not be regarded as a proxy of the economic impact of a product substitution. For example, the substitution of mineral fertilizer with composted FW usually generate important environmental benefits. The evaluation of the corresponding net economic effect might be more complex than the avoided purchase of mineral fertilizer by a farmer, because of contingent market consequences and economic dynamics, such as rebound effects.

3.4. Cut-off

Another aspect linked to system boundaries and the integration between LCC and LCA is cut-off. Hunkeler et al. (2008) underlined how certain life cycle stages, activities, and processes disregarded by LCA might have a large impact on costs. Therefore, cut-off criteria might present differences between the two methods even with consistent boundaries. In all reviewed literature, with the exception of Reich (2005) and Massarutto et al. (2011), some cut-off was used, although not in an explicit way (Table 2).

In general, it was possible to identify three cut-off levels:

– Environmental cut-off: cash flows directly linked to material flows (energy, materials, emissions) inventoried in the LCA were considered; studies thus focused on the costing consequences of resource use;
– Semi-financial cut-off: further cash flows related to processes (labor, capital, etc.) inventoried in the LCA were included; this approach allowed for a more complete assessment of the economic dimension and was adopted by a vast majority of reviewed papers;

Table 2
Overview of cut-off approaches for reviewed publications (Note: non-LCC references excluded).

Topic	Source	Approach			
		Environmental	Semi-financial	Financial	Not defined
Food	Mohamad et al. (2014)		√		
	Brandão et al. (2010)	√			
	Pergola et al. (2013)		√		
	Iotti and Bonazzi (2014)			√	
	Falcone et al. (2015)		√		
	De Gennaro et al. (2012)		√		
	De Luca et al. (2014)		√		
	Fenollosa et al. (2014)		√		
	Tamburini et al. (2015)		√		
	Amienyo and Azapagic (2016)	√			
	Schmidt Rivera et al. (2014)	√			
Food waste	Sargent et al. (1986)		√		
	Christoforou et al. (2016)		√		
	Kim et al. (2011)		√		
	Takata et al. (2012)		√		
	Vinyes et al. (2012)		√		
	Escobar Lanzuela et al. (2015)		√		
	Daylan and Ciliz (2016)		√		
	Bong et al. (2016)		√		
	Martinez-Sanchez et al. (2016)		√		
	Willersinn et al. (2017)		√		
Waste	Reich (2005)				√
	Massarutto et al. (2011)				√
	Asselin-Balençon and Jolliet (2014)		√		
	Martinez-Sanchez et al. (2015)		√		
	Woon and Lo (2016)		√		
	Rigamonti et al. (2016)		√		

– Financial cut-off: further processes not inventoried in the LCA but generating costs (research and development, trainings, etc.) were added; this criterion is the closer to usual business accounting since it takes into account also initial phases of product development that can be very capital intensive.

This distinction also had consequences in terms of data requirements, impact assessment, and comparability between studies. Due to lack of standardization, reviewed literature did not provide a straightforward insight on the appropriate choice for FW studies. LCC practitioners should identify the appropriate cut-off in the scoping phase of the study depending on the aim of the analysis. An environmental cut-off can be useful in identifying resource efficiency hotspots (e.g. surplus resource use associated to FW), a semi-financial cut-off could be needed to analyze potential capital or labor intensive measures (e.g. new processing machinery or increased workload from preventing losses), and a financial cut-off would be required to include expenses like new software or training for workers.

3.5. Cost modelling

In LCC, cost modelling represents a core methodological issue. Several categorizations can be applied and in this study it was deemed useful to follow the 4 levels identified by Hunkeler et al. (2008), namely: economic cost categories, life cycle stages, activity types, and other cost categories. Economic cost categories are related to the general type of cost, such as market costs, budget costs, and social costs. Life cycle stages categories relate to the segments of the supply chain analyzed, from product design and development to material extraction, use phase, and end of life. Activity types categories are a detailed specification of stages, including processes involved. Other cost categories are detailed cost items within activities and stages. These levels were not mutually exclusive and were usually combined in the reviewed studies (Table 3). Cost categorization by activity type was widely adopted in 13 papers, frequently together with a life cycle stage classification. These studies usually focused on food products and were characterized by a high level of detail. Costs were grouped by life cycle stages in 11 studies, usually in food LCC and long life span (e.g. orchards). A classification by economic cost typology was applied in other 10 cases, mainly dealing with short supply chains, FW, and waste management systems.

While all these categorization models are valid, the specific choice should be tailored on the goal and scope of the LCC study. Internal, external, avoided costs and revenues should not be aggregated if the focus is on system changes. Likewise, a detailed distribution of labor and capital costs along the supply chain might be useful for value added estimation.

Another key aspect of cost modelling is the allocation of indirect costs. If cost breakdown is carried out at a unit process level as in LCA (Martinez-Sanchez et al., 2015), some expenses, usually denoted as overheads and other costs that cannot be directly related to a product, need to be allocated. This is particularly relevant for FW studies, since food processing plants, composting and bioenergy plants, and disposal facilities usually treat more than one product or waste flow. In reviewed studies, indirect costs allocation was used in 7 cases. Tamburini et al. (2015) attributed all overheads to the studied product while Escobar Lanzuela et al. (2015) applied a fixed rate. In the other works, specific rates, either weight based (Kim et al., 2011; Schmidt Rivera et al., 2014; Vinyes et al., 2012) or usage rate of plants (Martinez-Sanchez et al., 2015; Massarutto et al., 2011), were calculated. In general, a specific rate based on physical or economic criteria and reflecting real situations is recommended over assumed fixed rates, but the choice depends on the analyzed system.

In a life cycle perspective, since costs might occur for different actors, a differentiation of costs by cost bearers can be included in the cost modelling. This is very relevant for FW studies, since various stakeholders sustain direct and indirect costs along the food supply chains and for FW management. In particular, costs from FW prevention, valorization, and management can be shifted

Table 3
Overview of cost models for reviewed publications (Note: non-LCC references excluded).

Topic	Source	Cost categories				Cost beares	
		Economic typology	Life cycle stage	Type of activity	Detailed cost typology	Single/none	Multi
Food	Mohamad et al. (2014)		√	√		√	
	Brandão et al. (2010)			√		√	
	Pergola et al. (2013)		√	√		√	
	Iotti and Bonazzi (2014)			√		√	
	Falcone et al. (2015)		√			√	
	De Gennaro et al. (2012)		√	√		√	
	De Luca et al. (2014)		√			√	
	Fenollosa et al. (2014)	√		√		√	
	Tamburini et al. (2015)			√	√	√	
	Amienyo and Azapagic (2016)		√	√		√	
	Schmidt Rivera et al. (2014)		√	√	√		√
Food waste	Sargent et al. (1986)	√				√	
	Christoforou et al. (2016)	√				√	
	Kim et al. (2011)		√	√		√	
	Takata et al. (2012)	√		√		√	
	Vinyes et al. (2012)				√	√	
	Escobar Lanzuela et al. (2015)	√			√	√	
	Daylan and Ciliz (2016)	√				√	
	Bong et al. (2016)	√			√	√	
	Martinez-Sanchez et al. (2016)	√					√
	Willersinn et al. (2017)	√	√		√		√
Waste	Reich (2005)			√		√	
	Massarutto et al. (2011)		√	√		√	
	Asselin-Balençon and Jolliet (2014)	√				√	
	Martinez-Sanchez et al. (2015)	√					√
	Woon and Lo (2016)	√				√	
	Rigamonti et al. (2016)		√			√	

from one actor to another by several measures and policies. Thus, a FW LCC study should include the analysis the distribution of costs among stakeholders. Nevertheless, multi-actor perspective was not common in reviewed studies and it was always related to the application of E-LCC with cradle (bin) to grave system boundaries (Martinez-Sanchez et al., 2016, 2015; Schmidt Rivera et al., 2014; Willersinn et al., 2017). The remaining literature usually adopted a single actor perspective, either explicitly or not.

Since LCC might deal with costs and benefits occurring at different times, discounting is another common feature of cost modelling, especially in C-LCC. A discount rate can be applied to cash flows also in E-LCCs, while discounting of results can be carried out only in C-LCC and S-LCC (Hunkeler et al., 2008). In 12 of the reviewed studies some discount rate was applied to cash flows with no particular differentiation between approaches. There was instead some linkage to the scope of the papers. In most cases, studies dealt with either food systems or waste management and only two papers on FW discounted costs over time (Martinez-Sanchez et al., 2016; Sargent et al., 1986). In the case of food products, authors discounted expenses for permanent crops (De Gennaro et al., 2012; De Luca et al., 2014; Falcone et al., 2015; Mohamad et al., 2014; Pergola et al., 2013) or in multi-year evaluation (Iotti and Bonazzi, 2014). No discounting was used in LCCs with an environmental cut-off criterion due to the exclusion of the cost of machinery (Amienyo and Azapagic, 2016; Schmidt Rivera et al., 2014) or to a steady-state assumption (Tamburini et al., 2015). While all waste studies adopted some discounting, this was not the case for FW, where some studies only applied depreciation to machinery and other capital costs and others did not mention or apply any discounting. As far as discounting methods are regarded, among the reviewed studies there was a prevalence of fixed rates, either assumed or estimated. Some of these studies applied a private interest rate, others a social rate (e.g. planning authorities). Only Iotti and Bonazzi (2014) used a variable rate equal to the historical government bond rates minus inflation. In general, sector-specific rates might be more appropriate for case studies characterized by a relatively small scale and with a focus on internal costs. Official rates from relevant authorities should be instead used when focusing on full supply chains and in societal studies. With regard to Europe, the European Commission suggested some examples of financial and social discount rates within the Guide to Cost-Benefit Analysis of Investment Projects for Cohesion Policy 2014–2020 (European Commission, 2015b).

3.6. Externalities

Externalities are quantifiable costs or benefits that occur when the actions of organizations and individuals have an effect on other stakeholders (Hunkeler et al., 2008). Those environmental and social impacts that can be expressed in monetary terms can be included in life cycle costing. However, their inclusion varies depending on the specific LCC approach adopted: in C-LCC they are usually not included; an E-LCC could include external costs that are judged to become internal costs in the relevant future; a S-LCC should monetize all costs for all stakeholders, including externalities (Hunkeler et al., 2008).

Monetization of externalities is so far lacking both standardization – the ISO 14,008 "Monetary valuation of environmental impacts from specific emissions and natural resources" should be published by December 2018 –, methods, and data. As remarked by Martinez-Sanchez et al. (2016), in a E-LCC coupled with LCA, externalities are covered in the environmental assessment. However, the economic cost of such impacts is not considered, to avoid double counting of the same impact in both assessment (e.g. CO_2 emissions assessed for their global warming potential and the related carbon tax). On the other hand, in a S-LCC, only externalities with accounting prices that can be calculated or estimated are monetized, thus some environmental impacts could be excluded from the economic assessment.

Despite several external costs that can be attributed to food, FW, and waste systems – from chemicals and greenhouse gases (GHG) emissions to disamenities and land use – only seven studies

Table 4
Externalities in reviewed studies (Note: non-LCC references excluded).

Source	What	How
Tamburini et al. (2015)	External cost of fertilizers and pesticides use	Unit cost (per kg of emission) of abatement, restoration or replacement of ecosystems and depuration of drinking water from literature
Kim et al. (2011)	Private benefits from byproducts	Unit market price
	Environmental benefits from avoided CO_2	Unit carbon price from the Clean Development Mechanism market
Vinyes et al. (2012)	CO_2 emissions mitigation	Unit cost from the international CO_2 market
Martinez-Sanchez et al. (2016)	Several emissions to air (CO_2, CH_4, N_2O, $PM_{2.5}$, NO_X, SO_2/SO_4, CO, HC, Hg, Pb, Dioxins)	National reduction costs, marginal reduction costs or taxes
	Indirect income effects	Environmental impacts and costs from available consumers income due to FW prevention
Massarutto et al. (2011)	Emissions to air (PM_{10}, NO_X, SO_2, VOC, CO, HCl, As, Cd, Ni, Cr VI, Hg, HF, Pb, Dioxins)	Data from EU project and literature
	CO_2 emissions	Average price of national emission trading certificate
	Disamenity and leachate	Data from EU project and literature
Martinez-Sanchez et al. (2015)	Emissions to air (CO_2, CH_4, N_2O, PM_{10}, NO_x, SO2, CO, HC, Hg, Pb, Dioxines, As, Cd, Cr VI, Ni)	National accounting prices
Woon and Lo (2016)	Opportunity cost of land	Sales comparison approach
	Disamenity	Housing unit price reduction
	Emissions to air (PM_{10}, $PM_{2.5}$, NO_X)	Impact pathway analysis from EU project

included externalities. This is likely due to the lack of agreement and methodologies. Table 4 summarizes the specific externalities covered and the methodologies adopted by different authors. Emissions into air were usually taken into account by considering the specific costs or price for the reduction or mitigation of impacts. Emission trading was considered a reliable proxy of CO_2 value. When landfill was the traditional disposal options, the external cost of disamenity was also included. Indirect (external) economic impacts were seldom included. In LCA-LCC joint results evaluation, externalities were not scored in the costing part to avoid double counting.

3.7. Impact assessment and LCC results interpretation

There is some debate around the appropriateness of LCC for assessing the economic pillar of sustainability, especially in combination with consequential LCA studies. The main criticism is that, by focusing on monetary costs for individual(s), LCC fails to grasp larger economic impacts, but is a useful tool for measuring companies' sustainability through their products (Gluch and Baumann, 2004; Jørgensen et al., 2013, 2010). The microeconomic perspective of LCC is still crucial since "environmentally friendly products have often higher purchasing costs, but frequently turn out to be cheaper if the use phase and/or the end-of-life phase are taken into account" (Jørgensen et al., 2010). LCC would have to be integrated within a Life Cycle Sustainability Assessment (LCSA) framework and include further indicators (e.g. added value) to fully capture the economic dimension of sustainability in case of consequential approaches (Hannouf and Assefa, 2016; Klöpffer and Ciroth, 2011; Wood and Hertwich, 2013).

Different costing approaches lead to diverse applications and perspectives. For example, C-LCC had a focus on the economic viability or impacts of investment costs (Mohamad et al., 2014) and did not include environmental implications. E-LCC was usually simultaneous with LCA and, unlike C-LCC, it could also show the distribution of net costs or savings within the supply chain (Schmidt Rivera et al., 2014). Finally, S-LCC was reputed to be useful in estimating larger welfare impacts (Martinez-Sanchez et al., 2015).

In reviewed articles, several techniques were applied besides cost assessment. Net present value (NPV) and internal rate of return (IRR) were the most frequent indicators, especially in studies dealing with long-term investment (orchards and plants) with a C-LCC or E-LCC (Fig. 2). However, these financial indexes usually imply a single actor perspective, rather than a systemic one. In four E-LCC studies, costs and/or benefits for the external system were assessed, also to derive a ratio. For example, Woon and Lo (2016) calculated both the cost-to-benefit ratio and the ratio between private cost/benefit to external cost/benefit. The distribution of costs among different cost bearers was calculated only in E-LCC studies, such as Martinez-Sanchez et al. (2016, 2015). Revenues were used to derive an estimate of income or profits in three cases. Willersinn et al. (2017) assessed profits distribution, Schmidt Rivera et al. (2014) estimated the value added along the supply chain, and Asselin-Balençon and Jolliet (2014) calculated the life cycle differential between scenarios. As for cost categorization, also cost assessment techniques should be chosen basing on the goal and scope of the assessment.

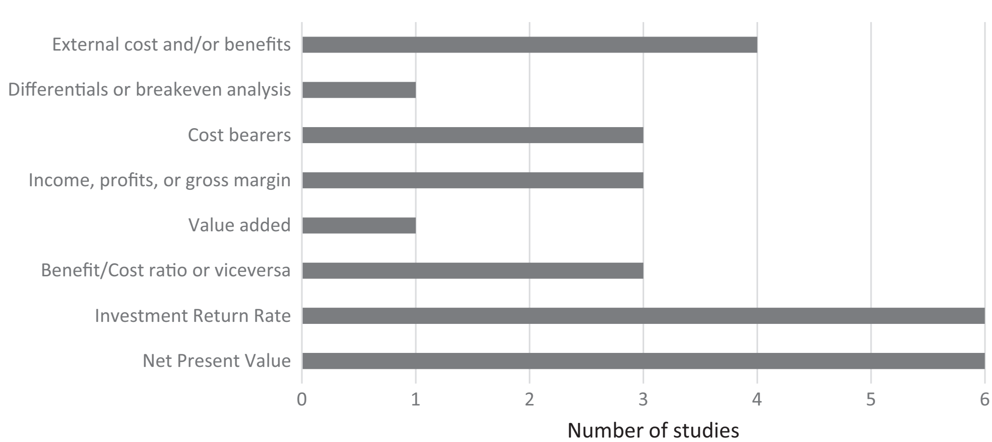

Fig. 2. Economic assessment methods in reviewed literature (Note: more than 1 method per paper; non-LCC references excluded).

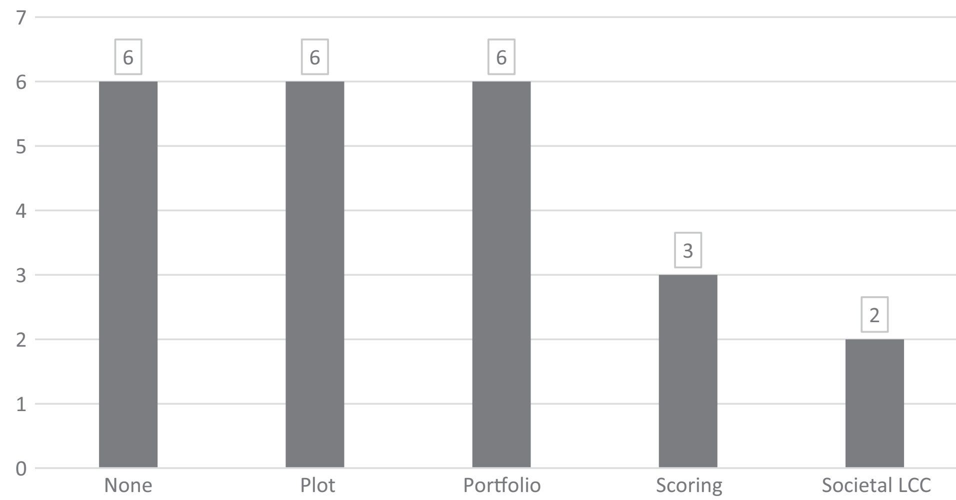

Fig. 3. Impact evaluation in case of joint LCA-LCC (Note: non-LCC references excluded).

When carried out in parallel with an LCA, an E-LCC can be used to identify trade-offs or win-win situations between environmental and economic impacts (Hunkeler et al., 2008). In this case, it is possible to report and analyze results from both assessments together. The three main options are portfolio presentations, plotting of results, or potential scoring for aggregated indicators. Fig. 3 shows the amount of reviewed papers adopting LCC and LCA (23), including S-LCCs, by typology of joint impact evaluation, if any. Six studies carried out separated evaluation of environmental and economic aspects. Apparently, this was not related to specific methodological aspects, nor was it dependent on whether the LCA was parallel or precedent to the LCC study. Half of the studies presented selected LCA and LCC indicators in a portfolio or in plot graphs. Usually, portfolios were tables or graphs showing few indices for each assessment, without weighting or normalization, e.g. De Luca et al. (2014) and Falcone et al. (2015). However, some portfolio presentations could already imply a scoring of scenarios. For example, Schmidt Rivera et al. (2014) summarized several environmental results of different meal scenarios and ranked them with a qualitative approach in a "heat map". A color ranking was assigned to each scenario in each criterion, rankings were then summed for each scenario (assuming equal weight), and final scores could be compared for an overall ranking (the lower the sum the higher the ranking). Plotting was used to specifically analyze cases of tradeoffs or win-win solutions. Asselin-Balençon and Jolliet (2014) plotted GHG emissions per FU against costs to derive a GHG abatement cost. Escobar Lanzuela et al. (2015) plotted profits for different scenarios against several corresponding LCA results (i.e., abiotic depletion savings against change in profits). Rigamonti et al. (2016) compared different waste systems by comparing environmental indicators (energy and material recovered per t of waste) against the economic indicator (costs per ton of waste), to identify the best possible win-win scenario.

As far as scoring is regarded, LCC and LCA results were weighted and normalized to be summed in an integrated scoring system. For example, Vinyes et al. (2012) first distinguished negative and positive indicators to identify total scores per scenario, based on their impact on sustainability (e.g. costs are negative). Values for each indicator were then converted in comparative percentages (100% is the worst or best scenario). Different scales (1–5) for negative (100% = 1) and positive indicators (100% = 5) were used to assign scores. Total scores per scenario and assessment were calculated as sum and then recalculated in relative terms (0–1): the closer to 1 the higher the contribution to sustainability. Willersinn et al. (2017) used a hierarchical attribute tree with percentage weights for each basic attribute. Reich (2005) instead used three weighting methods to convert LCA results into monetary units and sum them to LCC results. As already mentioned, this approach is very similar to a S-LCC, which was explicitly used by Martinez-Sanchez et al. (2016, 2015). Considering the intrinsic subjectivity of scoring mechanisms, portfolios and plots are inherently more transparent. However, in the communication of combined LCC and LCA results to stakeholders without the appropriate knowledge, a scoring system might be useful, especially if weighting is carried out in a participatory way.

An important step for a correct result interpretation is sensitivity analysis. Potential key assumptions that can have large effects on outcomes are: discount rates; period of analysis; incomplete or unreliable data or assumptions; expected variations in prices, also due to normative changes; value choices (Hunkeler et al., 2008). Nevertheless, less than half of authors (8) carried out a sensitivity analysis. De Gennaro et al. (2012) assigned uncertainty to techno-economic parameters and then applied a quite common tool as the Monte Carlo analysis to derive the probability distribution of profits. Furthermore, they assessed the variability of NPV and IRR in response to changes in the selling price of final product. Escobar Lanzuela et al. (2015) also used a Monte Carlo analysis to identify significant variables. Similarly, Christoforou et al. (2016) carried out a parametric analysis to identify ranges of three parameter influencing the potential competitiveness of the investment. However, various authors (Asselin-Balençon and Jolliet, 2014; Martinez-Sanchez et al., 2015; Massarutto et al., 2011; Sargent et al., 1986; Schmidt Rivera et al., 2014) simply tested alternative assumptions or parameters to assess changes in ranking scenarios and break-even values.

3.8. Other costing approaches

Other studies (6) included in the review process did not use an LCC approach, but they were considered and analyzed due to their thematic (FW) and methodological (costs or economic impacts) relevance. While not applying an LCC, these studies provided nonetheless interesting inputs for the discussion of current literature and further research needs. The main contribution was the FAO (2014) Full-cost accounting report, which can be regarded as a global assessment of FW costs from a societal perspective. A mix of defensive expenditure, damage costs, and well-being valuation were used to estimate the unit value of several impacts, from

GHG and ammonia emissions to land occupation, from biodiversity to pesticide poisoning. A narrower framework was represented by the set of studies authored by Nahman et al. (2012), Nahman and de Lange (2013), and de Lange and Nahman (2015). These works focused on FW related costs in South Africa, respectively at the household level, along the supply chain, and incorporating inedible FW. In the first two cases, the average market price of food was used as a proxy of the economic value lost due to wastage, and disposal impact included landfilling financial and external costs. The cost of inedible share was estimated through the price of unrecovered resources (energy and compost). Scherhaufer et al. (2015) reviewed several studies on the environmental and economic impact of FW, suggesting that several trade-offs of FW reduction measures could arise from the interaction between demand and supply of food and prices. Finally, Reynolds et al. (2015) focused on FW redistribution by charities and NGOs through an Input-Output framework, evaluating recovery costs, saved food value, calories, embodied water, energy, and greenhouse gases. Thus, these references could represent a potential source of data and methodologies for S-LCCs or E-LCCs that include externalities.

3.9. Dataset gaps and limitations

Another key aspect of LCC in general and of food waste in particular, is the quality and availability of reliable cost data. As highlighted by Hunkeler et al. (2008), access to primary cost data might often be affected by confidentiality issues. Furthermore, even good quality data on costs might be quite variable and have a relatively short validity. Thus, unlike LCA, the development of LCC databases is destined to face some hurdles. As far as FW is regarded, important gaps might specifically hinder studies with a larger perspective, e.g., country level or societal approach, and a consequential approach. An example is represented by the use of substitution in case of mulfunctionality. As mentioned, many reviewed studies adopted such approach, even without a consequential modelling framework. Both the point of substitution (i.e., what product is substituted) and the market dynamics of such substitution might play a pivotal role in terms of impact. Similarly, unintended trade-offs and price dynamics are not easy to address. As underlined by Martinez-Sanchez et al. (2016), the inclusion of direct and indirect land use change cost could represent another important feature for the evaluation of economic welfare impact of food waste management scenarios. No other reviewed paper addressed this data gap and no common methodology for the estimation of these costs exists. In a similar way, a standardization of more complete methods for the monetization of externalities would favor a societal perspective and the potential application of FW LCC for policy making.

3.10. Main recommendations and research needs

Table 5 summarizes the critical areas identified in the reviewed literature for each of the methodological aspects. In general, the lack of standardization produced a variety of approaches. However, depending on the aim, the number of the FW stakeholders, the type of perspective, one of the three approaches(C-LCC, E-LCC, S-LCC) might be more appropriate. Delimitations between current definitions are blurred. In this review, a mix of criteria was proposed for the categorization of studies according to Hunkeler et al. (2008). The presence of a parallel or previous LCA is a prerequisite for an E-LCC and S-LCC, but it is not a distinctive feature. The use of the same functional unit and consistent system boundaries is not sufficient to distinguish between C-LCC and E/S-LCC *per se*, if only one relevant stage is considered (i.e. in a cradle to farm gate perspective). On the opposite, if several stages of the supply chain are included, then the costing approach is rather different from C-LCC, e.g. using a multiple cost bearers perspective. Another key aspect is the coherence of inventory with LCA: the adoption of a unit process level also for costing as in Martinez-Sanchez et al. (2016) might help to distinguish E-LCC and S-LCC, as it is not usual in C-LCC. Finally, the distinction between S-LCC and other approaches seem sharper, since the monetization of potentially all environmental impacts into costs is a key feature. In this case, the development and testing of models for external cost estimation are critical to foster S-LCC distinctiveness and its application, for example, in the policy cycle.

Table 5
Critical areas for selected methodological aspects.

Methodological aspects	Critical areas	Recommendations or research needs
LCC approach	Blurred definitions and adoption	Approach based on aim, nr. of actors, integration with LCA, society perspective
	Unspecified modelling frameworks	Identify possible specificity of “consequential” LCC
	Guidance on FW	Identify FW assessment situations
Functional units	Variety of FUs	Identify correct FU depending on assessment situations included
System boundaries and multi-functionality	Consistency with LCA and modelling frameworks	In consequential modelling with system expansion, identify levels and indicators of economic consequences (e.g. effect on substituted products)
	Lack of standard approach	Approach based on modelling framework In system expansion, identify differences between revenues, avoided costs, avoided impacts
Cut-off	Guidance on levels	Environmental cut-off when focus is on resource efficiency Semi-financial cut-off for potential capital or labor intensive measures Financial cut-off to include R&D
Cost modelling	Categories	Ensure consistency with study aims and assessment methods
	Bearers	Inclusion is dependent on approach, aim and modelling framework
Externalities	Monetization of environmental impacts	Need to develop and test more monetization methods
	Consistency with modelling framework	In consequential LCC identify external economic effects
Impact assessment and results interpretations	Joint LCA-LCC assessment	Explore weighting and normalization methods
	Cost minimization vs. economic sustainability	Ensure consistency of approach and modelling framework with aim of the study

In assessing alternative future scenarios, there is also a strong need for consistent consequential modelling frameworks for LCC. While several guidance has been provided on attributional approaches, consequential LCA is less standardized, (Unger et al., 2016). For LCC this issue is even more critical considering its traditional microeconomic focus on costs rather than on larger economic effects. In general, consequential approaches should be more relevant for E-LCCs and S-LCCs, since both are conducted with LCA, includes several supply chain stages and costs bearers, and possibly external effects (i.e. consequences on other systems). Thus, further efforts should be devoted by the research community to the development of an integrated LCA-LCC analytical framework, including guidance on system expansion, indicators of economic consequences and external effects, and a coherent approach to multifunctionality and product substitution.

The lack of coherent approaches implies a high degree of variability among FW studies in terms of methodologies. This hinders the systematization of cases and the comparison of results. Basing on the type of study (e.g., attributional vs consequential), guidance on FW studies scoping should identify a set of assessment typologies, highlighting the differences between scenarios to be analyzed (e.g. prevention, valorization, and management) and practical solutions for comparing such situations (e.g. what FU, what to include in system boundaries, how to deal with avoided products).

Other aspect that should be addressed by a coherent analytical framework are cut-off level, basing to the options here proposed, and cost modelling, providing guidance on the consistency between cost typologies, aims of the assessment, and evaluation methods.

4. Conclusions

FW prevention, valorization, and management represent a societal imperative need, considering FW environmental and economic costs. The transition of food systems towards circularity will require consistent approaches for the assessment of current impacts and future scenarios. Life cycle methodologies can be appropriate tools for the identification of win-win solutions, maximizing environmental impact reduction and economic resource efficiency.

As emerged in the present analysis, in comparison with LCA, the standardization of LCC is still in its infancy. This generated a blurring of definitions, approaches, and methods. Only a limited amount of research has been conducted so far on life cycle costs of FW, either directly or indirectly by focusing on food supply chains and waste management systems. Most FW studies dealt with urban FW management, few focused on prevention of FW from the supply chain, or the valorization of specific flows. Furthermore, few food studies specifically modelled FW and waste studies mainly analyzed MSW.

Some basic recommendations and research needs can be derived from the analyzed results. First, the integration of LCC with LCA is recognized as a possible strategy to identify and avoid trade-offs between environmental and economic impacts. Consistent modelling frameworks allowing stakeholders from the supply chain to identify assessment situations and corresponding methodological choices are needed. However, FW prevention, valorization, and management scenario are often assessed before their realization. Further investigation could focus on the potential distinctive features of a "consequential LCC", in comparison with the more established attributional modelling. In fact, several differences might arise on functional units, system boundaries, and multi-functionality.

A third aspect is related to cut-off criteria. In this paper, it was possible to derive three different levels, based on the inclusion of more costs or cost-generating processes. Depending on the aim analysis, more focused on resource efficiency rather than labor, capital cost, or research and development, a cut-off criterion could be more suitable than another. This choice could be included and properly justified in the scoping phase to ensure comparability between studies.

Cost models applied in the studies were rather different, sometimes grouping costs by activity type, life cycle stage, or economic typology. Also in this case, further efforts could be aimed to determining the most effective and transparent categorization of costs. Since several stakeholders usually sustain costs of FW, modelling frameworks should be able to aggregate items without losing the ability to provide a distributional analysis.

Another relevant consideration regards impact assessment and results interpretation. The focus on costs is usually criticized as it implies that lowering costs is economically sustainable, but it does not reflect any larger impact on the economic system. It must be noted how even the assessment of revenues, profits, or value added would still be motivated by profit maximization. Further research could address how to integrate LCC with other key performance indicators and analytical tools, especially in case of consequential approaches. In addition, a joint evaluation of impacts with LCA results is always desirable when analyzing FW prevention, valorization, and management. However, no common approach exists on the eventual weighting or scoring of the two sets of results.

The critical areas identified by this paper could serve as a basis for the development of clear modelling frameworks and methodological guidance. Due to the role these tools can play for relevant decision-makers and stakeholders in tackling FW, crucial actors like the European Commission Joint Research Center or the Life Cycle Initiative might take the lead to ensure the harmonization and improvement of guidelines for the use of LCC.

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Appendix A. Supplementary material

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