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From linear to circular integrated waste management systems: A review of methodological approaches

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ABSTRACT

The continuous depletion of natural resources related to our lifestyle cannot be sustained indefinitely. Two major lines of action can be taken to overcome this challenge: the application of waste prevention policies and the shift from the classical linear Integrated Waste Management Systems (IWMSs) that focus solely on the treatment of Municipal Solid Waste (MSW) to circular IWMSs (CIWMSs) that combine waste and materials management, incentivizing the circularity of resources. The system analysis tools applied to design and assess the performance of linear IWMSs were reviewed in order to identify the weak spots of these methodologies, the difficulties of applying them to CIWMSs, and the topics that could benefit from further research and standardization. The findings of the literature review provided the basis to develop a methodological framework for the analysis of CIWMSs that relies on the expansion of the typical IWMS boundaries to include the upstream subsystems that reflect the transformation of resources and its interconnections with the waste management subsystems.

1. Introduction

Resources within planet Earth are finite by nature. Natural resources whose formation roots in other geologic periods, like mineral deposits, cannot be renewed in human timescales and thus their reservoirs are bound to eventually become depleted if their consumption continues (Prior et al., 2012; Shafiee and Topal, 2009). On the other hand, natural stocks subject to biological cycles (a population of trees for example) yield a sustainable flow of valuable goods and services (such as wood and CO_2 removal from the atmosphere) on a continuous basis (Costanza and Daly, 1992). Nonetheless, since the early 1970s some renewable natural resources are being exploited faster than they can be renewed (Borucke et al., 2013). As a matter of fact, it would take 1.64 planets to regenerate in one year the natural resources consumed in 2016 (Global Footprint Network, 2016). This figure is expected to worsen because of the projected population increase and the improved acquisition levels of the emerging economies (Foley et al., 2011; Karak et al., 2012).

If the consumption of raw materials rises, so does waste generation (Shahbazi et al., 2016). Around 1.3 billion tons of MSW are annually produced in cities all over the world (Hoornweg and Bhada-Tata, 2012), and a significant amount of the waste produced in low and lower-middle income countries is disposed of in open dumps (Hoornweg and Bhada-Tata, 2012) lacking measures to prevent safety and environmental hazards. Under the assumption that every ton of MSW generated in cities worldwide could be stored in 1 m^3 of sanitary landfill (Li et al., 2013), a landfill volume equivalent to that of 347,000 Olympic swimming pools would be required every year. Accordingly, policies against landfills are mostly motivated by a lack of space, particularly in the highly populated areas of Europe and Asia, where landfills are more likely to interfere with other land uses like agriculture (Moh and Abd Manaf, 2014).

In fact, waste valorization might help to overcome one of the most pressing global challenges: securing the food supply. Waste has been suggested as a plausible source to recover phosphorus (Reijnders, 2014; Tarayre et al., 2016; Withers et al., 2015), an essential nutrient to the metabolism of plants and by extension to agriculture, whose remaining accessible reserves could run out as soon as 50 years from now (Gilbert, 2009).

Hence, as the principles of industrial ecology dictate, resources and waste management are key to meeting the future needs of society in a sustainable manner. Waste prevention activities or policies such as restricting planned obsolescence in electronic products and measures like minimizing product weight or design for disassembly (Li et al., 2015) will contribute to tackle these issues.

A reduction in the consumption of natural resources and the amount

Abbreviations: CIWMS, circular integrated waste management system; EFA, energy flow analysis; IWMS, integrated waste management system; LCA, life cycle assessment; LCC, life cycle costing; MFA, material flow analysis; MCDM, multi-criteria decision-making; MSW, municipal solid waste; SFA, substance flow analysis

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of waste generated would also be accomplished if a shift to circular economic and production systems, mimicking the self-sustaining closed loop systems found in nature, such as the water cycle, was put into practice. A circular economy aims at transforming waste back into a resource, by reversing the dominant linear trend of extracting, processing, consuming or using and then disposing of raw materials, with the ultimate goal of preserving natural resources while maintaining the economic growth and minimizing the environmental impacts (Ghisellini et al., 2016; Lieder and Rashid, 2016).

In a circular economy the reduction in the environmental impacts, such as global warming, is due to the improvement in resource and energy efficiencies. For instance, it has been demonstrated that the production of secondary aluminum from scrap consumes less than 5% of the energy needed in the production of primary aluminum (JRC, 2014); this entails that the emission of up to 19 tons of equivalent CO_2 to the atmosphere could be avoided per ton of aluminum that is recycled instead of produced from the mineral ore (Damgaard et al., 2009).

Given all the benefits that the circularity of resources has to offer, the reasonable question to pose is how society and industry can successfully transition to a circular economy. The straightforward answer from an engineering point of view is through the design of efficient CIWMSs that link resource processing and waste treatment, and allow the potential of waste to be fully exploited. A CIWMS is expected to produce not only materials, but also energy and nutrients; additionally, it could deliver certain chemicals.

Therefore, a trade-off between the functions of a CIWMS is unavoidable. A thorough analysis must be carried out prior to the design stage of a CIWMS so that it can assist in the decision-making process. As the analytical framework supported by systems thinking can provide a holistic view on the sustainability challenges that arise from the interconnections between the components of an IWMS (Chang et al., 2011; Singh et al., 2014), so far manifold papers applying a systems-oriented approach to waste management have been published.

That is the reason only the most recent papers focusing on the analysis of IWMSs have been addressed in this study. The aim of this paper is to conduct a critical and comprehensive review of the studies published since 2011 that analyze IWMSs whose input is MSW, in order to gain insight into the strengths and shortcomings of the methodologies currently being applied, and to identify their applicability to a sustainable CIWMS targeting resource recovery. To the best of the authors' knowledge, an IWMS has never been analyzed from the perspective of a circular economy before. The novelty of this review is that the characteristics of a CIWMS are defined, the potential pitfalls of applying the current methodologies deployed in the analysis of linear IWMSs to a CIWMS are identified and possible methodological improvements are proposed.

This review is structured as follows: first, the methodology applied in the selection of the reviewed papers is described. Second, the state-of-the-art technologies and processes for IWMSs are outlined, along with their potential restraints to the development of a circular economy. Third, the characteristics of a CIWMS are defined. Next, the methodologies currently applied to analyze IWMSs are briefly described and the hottest topics regarding the methodological aspects of the analysis of IMWSs are subsequently identified. Finally, the conclusions drawn from the findings of the study are summarized, with special emphasis on the Life Cycle Assessment (LCA) methodology.

2. Method

77 papers analyzing IWMSs that treat MSW and published after 2010 were identified by means of the Scopus database (Scopus Website, 2016). They are listed in Appendix A. The systematic review method was conducted applying four different keyword strings: i) *municipal solid waste*; *integrated*; *system* and *analysis*; ii) *municipal solid waste*; *integrated*; *system* and *methodology*; iii) *municipal solid waste*; *integrated*; *system* and *(sustainable* or *sustainability)*. The papers focusing on the analysis of scenarios regarding alternative waste treatment technologies or processes were excluded from the review.

Once the technological obstacles faced by CIWMSs and the limitations of the methodologies applied for the analysis of IWMSs were detected in the reviewed studies, the search criteria were expanded to cover the specific topics of interest. Those additional papers are listed throughout the document.

3. Technological background

Prior to the proposal of guidelines for the analysis of CIWMSs that enhance the circularity of resources and enable the transition to a circular economy, it is mandatory to recognize the technological restrictions to the implementation of such a system. They are outlined in this section.

3.1. Quality and value of recycled materials

The market penetration of recycled materials is highly dependent on their physical and chemical characteristics, which will determine their price. However, not all the existing recycling technologies enable a fair competition between virgin and secondary materials, because their quality might differ.

Recycling technologies either downgrade or upgrade the materials in respect to the quality of the virgin materials. Downgrading implies that the properties of the recycled material are not as good as those of the virgin material. Instead, upgrading technologies improve the quality of the waste materials at least up to the quality of the virgin materials.

In closed-loop recycling, the material is recycled into the same product system and the inherent properties of the recycled material are maintained virtually identical to those of the virgin material. Oppositely, in open-loop recycling the material is recycled into a different product system and its inherent properties may or may not differ to those of the virgin material (ISO 14044, 2006). Closed-loop recycling is not equivalent to infinite recycling; materials can be used and later recycled within a closed-loop system for a number of times, until microstructural changes in the material or the accumulation of chemical elements and compounds hamper its further reuse (Gaustad et al., 2011).

A case of closed-loop recycling occurs when a glass bottle is recycled into a glass jar, because the glass jar could be recycled back into a glass bottle with the same functionality as the original one (Haupt et al., 2017a), whereas recycling PET bottles into PET fibers is an example of open-loop recycling (Shen et al., 2010); it is an irreversible process.

Recycling processes can be further classified as downcycling or upcycling processes. Downcycling has been defined as the recycling of materials into lower value products (Gaustad et al., 2012). The use of wrought scrap in cast products, due to their ability to accommodate higher silicon contamination, is considered downcycling. On the contrary, if the waste materials are recycled into products of higher value, the recycling process is called upcycling (Pol, 2010). Upcycling involves a change in the fundamental properties of the material, like its physical structure or its chemical composition. Novel approaches to upcycling described in the literature entail chemical (Pol, 2010; Zhuo et al., 2012) or biological transformation (Kenny et al., 2008). Fig. 1 compiles the types of recycling processes according to the quality of the recycled materials and the value of the resulting recycled products in respect to the original materials and products.

Although downgrading and upgrading are often used as synonyms of downcycling and upcycling, Fig. 1 shows that is not necessarily true: a waste material may be upgraded to maintain its original function, and later used to manufacture a product of lower value than the original one. The confusion regarding the terminology has recently been intensified by Geyer et al. (2016), who question the usefulness of making

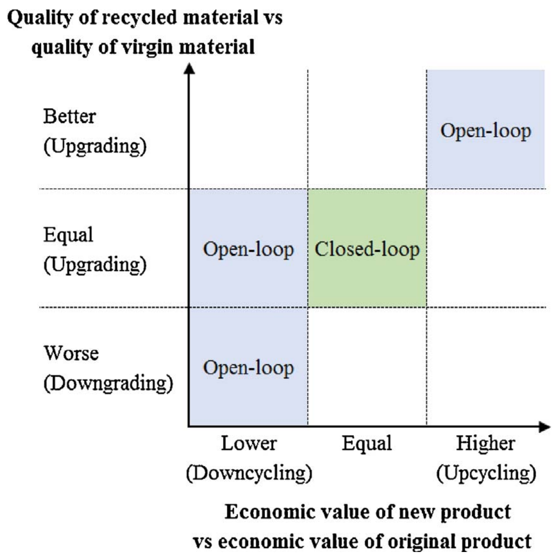

Fig. 1. Classification of recycling processes.

a distinction between open and closed-loop recycling.

3.2. State-of-the-art technologies and processes for IWMSs

Regarding the technical and economic factors that hinder the complete separation and recycling of materials (O'Connor et al., 2016; Ciacci et al., 2015; Reuter, 2011), the concentration of the valuable materials in the discarded products and wastes is one of the critical parameters that will determine the feasibility of the recovery process (Johnson et al., 2007); several authors agree that the *unrecyclability* of some materials stems from the combination of small quantities of multiple materials in one product, like a smartphone (Reck and Graedel, 2012; Chancerel et al., 2013). Hence the need to design systems that contemplate the valorization of all the materials within a given product. Clearly, the solution to this challenge relies on the development of more efficient sorting and disassembly technologies, along with the implementation of policies that promote the separate collection of these wastes.

One strategy that has been proposed to tackle the limitations of the current recycling technologies is to store in landfills the waste that cannot be properly separated or recycled until the pertinent technologies have been developed up to the point that they enable the recovery of the remaining secondary raw materials in waste (Bosmans et al., 2013), which is the prime purpose of landfill mining, along with energy recovery from the stored waste (Jones et al., 2013). Although several environmental and economic assessments of landfill mining have been performed so far (Danthurebandara et al., 2015; Laner et al., 2016; Van Passel et al., 2013), more applied research is needed before the most sustainable pathway to landfill mining is agreed upon (Krook et al., 2012).

Even though recycling efficiencies reached their full potential in the future, MSW is a complex heterogeneous mix of materials, and that prevents it from being treated by a single technology (Arena, 2015). It is important to make a distinction between waste treatment, that is to say, the set of processes seeking to minimize the environmental impacts of waste in order to comply with the pertinent regulations, and waste valorization, which concerns the transformation of waste into a product capable of providing society with a valuable service. However, a given waste management system can provide both functions, that is to say, waste treatment and waste valorization.

A MSW management system focused on valorization must include a subsystem for materials sorting. The paper, cardboard, plastics, glass, aluminum and iron present in MSW are usually sorted in material recovery facilities and sent to recycling industries, where they are upgraded to be reintroduced into the market. For further information about the quality of recyclables and their recovery efficiencies in commingled and single-stream waste, the reader should refer to Cimpan et al. (2016). There are several options for the valorization of both the inorganic and organic remaining materials. The alternative treatments to recycling the inorganic fraction of waste such as leftover plastic or textiles are the waste-to-energy processes like incineration, gasification or pyrolysis; the most developed and widespread of which is incineration (Arena, 2012). These thermochemical processes can also be applied to the organic fraction of waste. The biological processes of anaerobic digestion and composting enable the organic matter to be looped back into the system as fertilizer (digestate or compost) (Brändli et al., 2007), so they could be considered recycling processes. In fact, anaerobic digestion is a strategy to simultaneously recover nutrients from the solid digestate and energy from the biogas produced by the microorganisms (Sawatdeenarunat et al., 2016).

Furthermore, new processes to valorize the organic fraction of waste are being proposed. The fermentation of organic waste has been suggested as a method to produce hydrogen (Poggi-Varaldo et al., 2014). Another example is the enzymatic liquefaction process proposed to separate the solid non-degradable materials that can be upgraded to Refuse Derived Fuel from a bioliquid that can be digested to produce biogas (Tonini and Astrup, 2012). In addition to those, a number of processes to produce valuable chemicals such as levulinic acid (Sadhukhan et al., 2016) from organic waste or Refuse Derived Fuel have arisen; these are upcycling processes that fall within the category of waste refineries. Several authors propose to gasify waste in order to obtain syngas, a precursor to either the catalytic synthesis of methanol or the production of hydrocarbons via the Fischer Tropsch process (Lavoie et al., 2013; Niziolek et al., 2015; Niziolek et al., 2017; Pressley et al., 2014). Of the above-mentioned processes, the only one at large scale is operated by the company Enerkem, with a production capacity of 38,000 m^3 of methanol per year (Enerkem, 2017).

3.3. Materials recycling or energy recovery?

In the specific case wherein the current state of the technologies allows a residual material to undergo either a recycling or an energy recovery process, materials recovery is usually encouraged; the Waste Framework Directive (EP and EC, 2008) states that, unless adequately justified by LCA, the EU Member States must follow the waste management hierarchy, according to which materials recycling takes precedence over energy recovery.

However, whereas the vast majority of studies agree that landfill is the least desired waste management alternative from an environmental point of view (Belboom et al., 2013; Coventry et al., 2016; Eriksson et al., 2005; Erses Yay, 2015; Fiorentino et al., 2015; Manfredi et al., 2011; Tulokhonova and Ulanova, 2013), and there is also consensus on the claim that waste prevention and re-use are the cleanest and most efficient policies, the performed literature review reveals an ongoing debate on the final destination of the recyclable fractions of waste (Blengini et al., 2012; Consonni et al., 2011; Merrild et al., 2012): should they be reintroduced into the production cycles, as new products or compost, or be sent to energy recovery facilities? The answer will greatly depend on the composition of the waste stream, which will determine its heating value and thus, its energy recovery potential. Furthermore, the assumptions made in the analysis, the system boundaries set and the local characteristics of the specific case study, will determine the optimal valorization strategy.

Cossu (2014) analyzed the reasons behind the promotion of recycling. It causes the preservation of natural resources inasmuch as they are being extracted to a lesser degree. Moreover, a reduction in the amount of waste that needs to be properly managed or disposed of gives rise to cost savings in treatment processes. Nevertheless, the assumption that the economic costs and environmental impacts of material recycling are lower than those related to the extraction and processing of the virgin raw materials cannot be substantiated without a thorough analysis.

In the context of a globalized market, one of the factors that play a key role to the detriment of materials recycling is the long transport distances that they must go through to reach their end-users (Merrild et al., 2012), which has both environmental and economic drawbacks. Additionally, Massarutto et al. (2011) proved that if a critical recycling rate (the ratio between the recycled materials and the waste generated) is exceeded, the economic benefits from recycling do not compensate its costs. Their study was based on the assumption that the quality of the collected materials worsens as the separation levels (the ratio between the source separated waste and the total amount of generated waste) increase, which was verified with data from waste management systems.

Several other authors have emphasized the importance of assessing the effect of increasing the recycling rates on the quality of the materials (Arena and Di Gregorio, 2014; Cossu, 2014; Haupt et al., 2017b Rigamonti et al., 2009). Some studies concluded that higher separation levels are not indicative of better materials quality (Consonni and Viganò, 2011; Rigamonti et al., 2009). On the contrary, systems focusing on quality rather than on quantity are likely to outperform the others.

An example of the damaging effects of recycling can be found in the steel manufacturing industry. The increased use of secondary materials in the steel making process causes an accumulation of elements such as copper, which hardens steel decreasing its quality and making it necessary to dilute the amount of recycled scrap (Haupt et al., 2017b). The counter-effect of dilution is that it reduces the market demand for recyclables (Modaresi and Müller, 2012). Hence, as Loughlin and Barlaz (2006) pointed out, recycling policies must make sure that the supply of recycled materials matches the demand.

Particular attention must be paid to the potential hazards of recycling because of human exposure to pollutants and toxic compounds. Bisphenol A was found in an array of waste paper samples, possibly as a consequence of the recycling of secondary waste paper (Pivnenko et al., 2015). Recycling has also been recently pointed as a potential source of phthalates in plastics (Pivnenko et al., 2016); as a consequence, the application of recycled plastics in products sensitive to phthalate content, such as toys and food packaging, must be restricted.

The risk for human health is in fact the main argument that the detractors of energy recovery technologies hold, despite the fact that the thermochemical processes and anaerobic digestion are a means to simultaneously reduce the volume and mass of solid waste and produce heat and electricity. Incineration has been traditionally regarded by the public opinion as a threat to human health and the environment, because of the high concentrations of heavy metals, dioxins and furans present in the flue gases prior to the development of the current sophisticated Air Pollution Control Systems (Brunner and Rechberger, 2015). However, with the state-of-the art technologies, these pollutants do not pose a risk any longer, since they are well below the air emission limit values established by the European legislation, which are quite restrictive in comparison to those of other countries (Vehlow, 2015).

Furthermore, several studies report that savings on the environmental impacts can be achieved displacing conventional energy sources by MSW (Boesch et al., 2014; Fruergaard and Astrup, 2011). Hence the importance of linking the analysis of the energy and waste management systems (Juul et al., 2013), as Eriksson and Bisaillon (2011) and Münster et al. (2015) did.

The competition between materials recycling and energy recovery is of particular interest for those materials such as cardboard and plastic with high calorific values (Merrild et al., 2012), which make them attractive fuels for heat and electricity production, whereas deliberately subjecting the incombustible materials, i.e. metals and glass, to energy recovery processes seems pointless. However, a fraction of the metals that cannot be separated by mechanical and magnetic methods can be recovered after the incineration process, because of their enhanced concentration in the residual ash (Cossu and Williams, 2015).

Taking into account all the considerations described above, it is reasonable to conclude that materials recycling and energy recovery should complement each other to meet the local demands; even in the utopian scenario wherein it is technologically and economically feasible to completely close the material loops, there might still be a demand for virgin materials, not only because of their higher quality, but also because of social objections.

4. Framework for the analysis of CIWMSs

The precise definition of a CIWMS is instrumental to the development of a framework that relies on that concept. The previously discussed barriers to the development of CIWMs should provide a basis for the delimitation of their system boundaries and the definition of their functions. These notions, which are based on the principles of the cradle-to-cradle design (McDonough and Braungart, 2002), are explored to a greater extent in this section.

4.1. Previous application of the circular economy approach to the design of IWMSs

Although specific guidelines for the design and assessment of CIWMSs from a systems perspective have not been found in the literature, Arena and Di Gregorio (2014) proposed a series of principles, consistent with the targets of the circular economy, that IWMSs should follow: “An integrated and sustainable waste management system should be defined and developed according to the following criteria: i) to minimize use of landfills and ensure that no landfilled waste is biologically active or contains mobile hazardous substances (…); ii) to minimize operations that entail excessive consumption of raw materials and energy without yielding an overall environmental advantage; iii) to maximize recovery of materials, albeit in respect of the previous point; and iv) to maximize energy recovery for materials that cannot be efficiently recycled, in order to save both landfill volumes and fossil-fuel resources”.

4.2. Proposed definition

A description of the concepts of IWMSs and CIWMSs is provided in this section. An IWMS denotes a system whose main input is waste and comprises a number of processes to sort this waste and give each waste fraction the most appropriate treatment according to its chemical composition and the desired function of the system outputs. However, this definition corresponds to that of a linear IWMS, like the one shown in Fig. 2. If an IWMS is to be studied from the perspective of a circular economy and waste prevention, this definition is incomplete. A CIWMS is a type of IWMS that seeks to enhance the circularity of resources by strengthening the link between waste treatment and resource recovery. Thus, CIWMSs can be considered an instrument that enables to fulfill the goals of a circular economy. The definition of CIWMSs could also apply to a system that focuses on just one waste fraction, such as organic waste.

The purpose of a sustainable CIWMS is to achieve the maximum economic profit and benefits for society at the expense of the minimum environmental impacts and consumption of natural resources. Under this perspective, materials upcycling is favored over downcycling. To accomplish these sustainability goals, the maximum amount of waste is expected to be valorized to expand its lifetime, so that it can serve a function to society. This entails that the amount of waste sent to landfill is minimized, although landfills cannot be totally replaced (Cossu, 2012) because all the other subsystems generate certain amount of waste that the current technologies cannot valorize.

A CIWMS can be as complicated as the designers wish, but a CIWMS that manages mixed MSW would ideally deliver materials, energy and

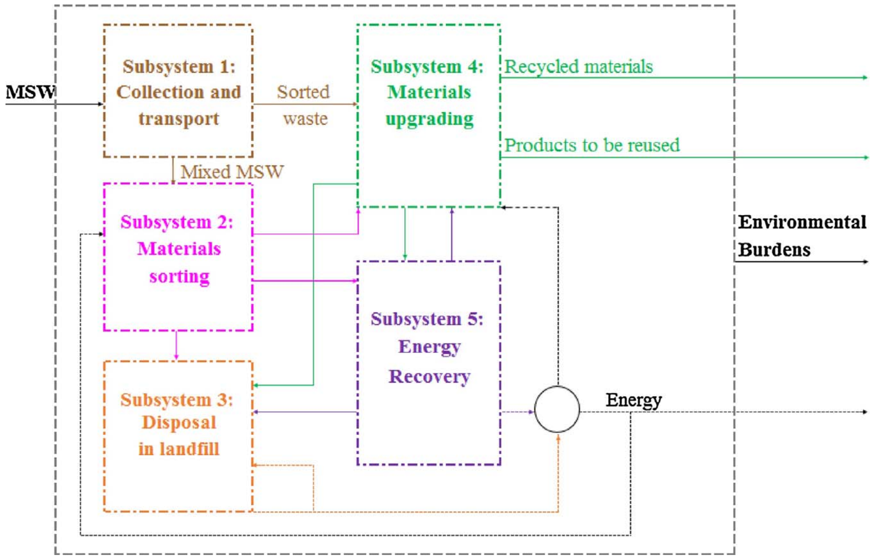

Fig. 2. Linear IWMS (2-column fitting image).

nutrients. It could also supply some chemicals, a relatively novel approach to waste management. The waste refinery concept, analogous to that of an oil refinery but taking waste as a feedstock, has gained popularity in recent years (Richards and Taherzadeh, 2015). A waste refinery is a type of IWMSs wherein chemical reactions take place to upcycle mixed waste or a fraction of waste into marketable chemicals.

4.3. Configuration and boundaries of a CIWMS

A CIWMS should encompass the subsystems that connect the transformation of raw materials into waste with the waste treatment subsystems, so that the consequences of the recirculation of the materials into the upstream subsystems can be fully accounted for. A CIWMS that relies to a lesser extent on the consumption of virgin raw materials would result from the connection of the upstream subsystems with those of a traditional linear IWMS, as shown in Fig. 3. As many transport subsystems as necessary should be added to the system depicted in Fig. 3 for each particular case under study. From an LCA perspective, the subsystems 0–2, which comprise the upstream and midstream processes, constitute the background system of the model, whereas the remaining downstream subsystems, which concern those processes under the control of the decision-maker (Frischknecht, 1998), belong to the foreground system.

These system boundaries intend to capture the whole life cycle of the materials that compose waste, including the stages concerning the consumption of the services derived from the transformation of the natural resources extracted from the ecosystems. Once consumed, some products such as food or cosmetics leave the system as air emissions or wastewater. On the other hand, many products like textiles and furniture provide a service for a time period without being consumed. It is worth mentioning that the primary raw materials delivered by subsystem 0 cannot be compared to the secondary materials produced in subsystem 6 on a mass basis; the comparison must be based on the functions provided by those materials. For instance, 1 kg of primary aluminum might not be functionally equivalent to 1 kg of recycled aluminum, because of their different chemical composition and physical properties.

Fig. 4 illustrates the exchanges between a CIWMS and the surrounding ecosystems, and how a CIWMS is capable of transforming one type of environmental burden (waste) into a resource that might

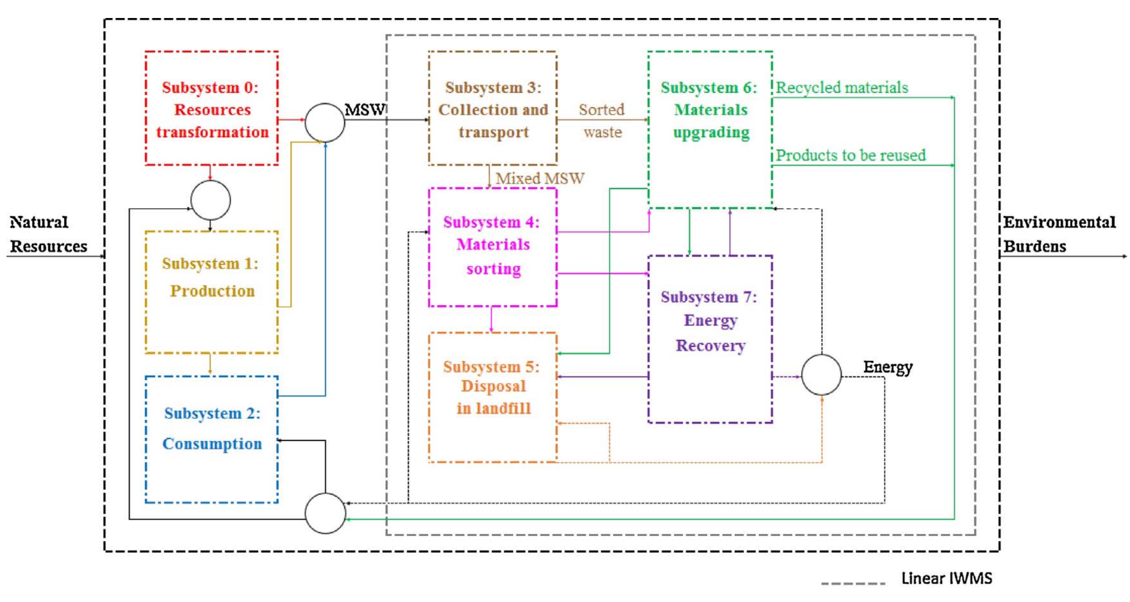

Fig. 3. Configuration and boundaries of a CIWMS.

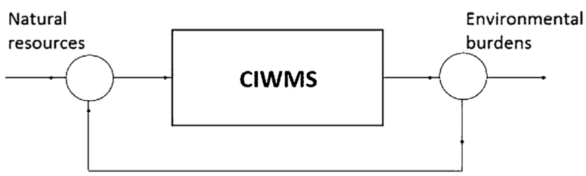

Fig. 4. Overview of the exchanges between a CIWMS and the ecosystems.

displace the consumption of virgin resources that would provide the same function.

The scope of a CIWMS that manages mixed MSW is so broad that the only systems within the technosphere that it might be related to are the wastewater and the industrial waste treatment systems. Those systems are outside the scope of the study of the CIWMS shown in Fig. 3 and thus, the consequences of the decisions affecting those systems will not be considered.

4.4. Link between industrial symbiosis and CIWMSs

According to Chertow (2000), industrial symbiosis engages traditionally separate industries in a collective approach to competitive advantage involving physical exchange of materials, energy, water, and/or by-products. The keys to industrial symbiosis are collaboration and the synergistic possibilities offered by geographic proximity. Thus, the proposed CIWMS is analogous to an industrial symbiotic systems, in the sense that a resource exchange network can be stablished. Nonetheless, although industrial symbiotic systems could play a major role in the circular economy, the concept of a CIWMS is much broader; it is not restricted to nearby industrial systems, but it also includes waste managers, consumers and the supply chains. That is to say, not all the materials within a CIWMS are reintroduced into the production cycles because of an agreement between companies.

Hence, the generic methodological approaches proposed in the literature to assess the performance of industrial symbiotic systems (Martin et al., 2015; Mattila et al., 2012) should not be, a priori, extended to CIWMSs.

4.5. Recommended tools for the analysis of CIWMSs

Because of the wide range of existing technologies to manage waste, process engineers must carefully study the available possibilities at the design phase of a CIWMS. The superstructure that might emerge after considering process integration could be quite complex. Thus, the selection of the optimum configuration of the system is not a trivial matter, and it might require mathematical programming techniques. Moreover, since the chemical composition of waste will determine the type of processes that it can be subjected to, it can be concluded that the design of a CIWMS should be based on mathematical programming and Material Flow Analysis (MFA), so that the circularity of materials is warranted. The combination of these tools with scenario analysis techniques that assess the consequences of changes in waste composition and quantities or possible technological improvements, could be a valid strategy to account for the dynamic variables that might fluctuate during the studied time horizon.

On the other hand, the assessment of the performance of a CIWMS must analyze all its sustainability dimensions. The sustainability criteria regarding the economic and social dimensions of CIWMSs are at least as important as the environmental aspects and must be likewise assessed; nonetheless, they will not be deeply discussed in this Critical Review.

5. Methodologies applied in the literature

Regarding the methodological approaches reported to be applied in the literature, Chang et al. (2011) and Juul et al. (2013) classified the system analysis tools that have the potential to assist in the design of IWMSs and the decision-making processes as:

i) System engineering models, which focus on supporting the design of the system. These are simulation models, optimization models, forecasting models, cost-benefit analysis or multi-criteria decision-making (MCDM).
ii) System assessment tools. They focus on assessing how an existing system performs. LCA, MFA and risk assessment are examples of such tools.

Coupling these two types of methodologies is recommended not only because it will lead to a better understanding of the IWMS (Pires et al., 2011c), but also because the sustainability analysis of an IWMS requires an integrated approach; the applied methodologies should complement each other so that all the sustainability dimensions can be properly evaluated and the economic, environmental and social objectives are balanced.

Another strategy that has been suggested to support the decision-making process is taking a participatory approach. This can be done by either asking multiple stakeholders to participate in the analysis (Blengini et al., 2012), or by applying a game-theoretic approach that seeks the fair distribution of benefits and costs (Karmperis et al., 2013).

The methodological approaches applied in the 77 reviewed papers are shown in Fig. 5. Whereas over one third of the reviewed papers focus solely on the environmental impacts associated with the IWMS (all of them by means of LCA), only one study relies solely on an economic assessment, based on Life Cycle Costing (LCC) (Massarutto et al., 2011). More information on the application of LCC to waste management systems can be found in Martinez-Sanchez et al.'s paper (2015).

Over one fifth of the reviewed studies assessed more than one sustainability dimension. A few papers (Chang et al., 2012; Levis et al., 2013; Levis et al., 2014; Martinez-Sanchez et al., 2017; Münster et al., 2015; Tabata et al., 2011), combine the LCA methodology and optimization techniques to broaden the scope of the study and include other sustainability criteria. Mirdar-Haridani et al. (2017) combined optimization and social LCA. Multi-objective optimization, applied in some of the reviewed papers (Chang et al., 2012; Chang and Lin, 2013;

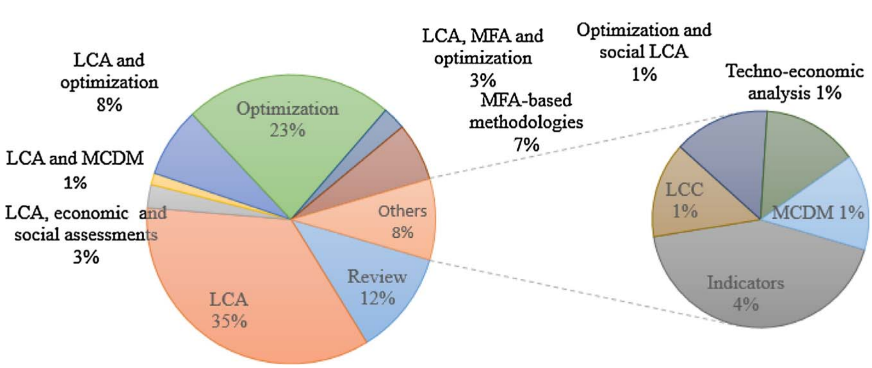

Fig. 5. Methodological approaches applied in the reviewed studies.

Santibañez-Aguilar et al., 2013; Santibañez-Aguilar et al., 2015; Srivastava and Nema, 2012; Vadenbo et al., 2014a,b), is possibly the most adequate technique to take into account all the sustainability criteria. Oppositely, other authors (Menikpura et al., 2012; Tulokhonova and Ulanova, 2013) combined LCA with a set of indicators to account for the other sustainability dimensions of an IWMS.

On the other hand, MFA and/or Substance Flow Analysis (SFA) enable to explicitly consider the waste characteristics and thus help provide a more detailed description of the system under study and track each waste fraction throughout the system. Additionally, Energy Flow Analysis (EFA), which was applied in two studies (Herva et al., 2014; Tonini et al., 2014), might prove useful to determine the most suitable valorization treatment to each waste fraction.

So far, the theoretical framework required to combine LCA, multi-objective optimization and MFA techniques has only been described by Vadenbo et al., (2014a,b) although the methodology was not applied to an IWMS.

6. Hot topics

The most discussed methodological aspects in the reviewed studies and the challenges and possibilities of their application to the design and assessment of CIWMSs are presented in this section aiming at providing some helpful and critical insights into the development of a theoretical framework for the analysis of CIWMSs.

6.1. Accounting for waste prevention

Wastage of goods and products is a tremendous global challenge; taking the food supply and consumption chains as an example, around one third of the food produced for human consumption worldwide is currently lost or wasted (FAO, 2013).

Waste prevention stands at the top of the waste management hierarchy, as a strategy to be implemented in the life cycle stage prior to waste generation that seeks to minimize the depletion of natural resources and its subsequent environmental burdens. The term *waste prevention* refers to any measures taken before a substance, material or product become waste, that reduce: a) the quantity of waste, b) the adverse impacts of the generated waste and c) the content of harmful substances in materials and products (EP and EC, 2008).

Nevertheless, the analysis of waste prevention activities in the framework of LCA has not been normalized yet; only a few studies outline the methodological steps to follow (Cleary, 2010; Gentil et al., 2011; Nessi et al., 2013), concurring that this is an active area of research.

LCA models of waste management typically calculate the environmental burdens on a waste mass basis. This is the most straightforward option to choose the functional unit. However, it makes this approach inadequate for the comparison of scenarios including waste prevention strategies, given that the amount of waste produced varies among them (Ekvall et al., 2007). Moreover, these models usually rely on the “zero burden approach”, which does not include the upstream processes within the system boundaries because it is assumed that their primary function is not to produce waste and thus none of the environmental burdens generated in the upstream processes are associated with it. Nonetheless, if different amounts of waste are produced in each scenario, the zero burden approach cannot be considered because the contribution of the upstream processes to the overall environmental impacts of the system will differ (JRC, 2011). Consequently, a proper methodological approach to deal with waste prevention activities from a life cycle perspective should define:

i) A functional unit that accounts for the amount of waste prevented.
ii) System boundaries that include the upstream processes involved in waste generation.

Another issue that must be considered when waste prevention activities are being accounted for is the allocation procedure of the environmental impacts among the products or services delivered by the IWMS. Applying the direct substitution approach in order to avoid allocation among several products is not recommended, given that negative results might be obtained, leading to the erroneous conclusion that a greater amount of waste leads to less environmental impacts (Giugliano et al., 2011).

Cleary (2010) recommends an attributional approach with system expansion to account for the upstream processes associated with waste production, arguing that a consequential approach does not consider waste prevention as a waste management strategy functionally equivalent to the others in the waste management hierarchy, since no environmental burdens are attributed to waste prevention activities; that is to say, it simply quantifies the consequences of reducing the waste inputs in the system. Only Gentil et al. (2011) claim to apply a consequential LCA model. These authors expand the system boundaries to the upstream processes related to the waste generation processes, although they acknowledge that the cascading effects of waste prevention should have been further assessed.

All of the above mentioned studies define the functional unit as the sum of the waste managed through conventional methods and the amount of waste prevented, although nuances in the applied approach can be found among the studies.

6.2. Quantifying biogenic carbon

Whether biogenic CO_2 emissions are considered neutral or an environmental burden to an IWMS will have a significant influence on the results and conclusions drawn from the analysis. Since studies relying on different assumptions are hard to compare, it is imperative to standardize this matter, not only within the waste management sector.

The EPA (2017) defines biogenic CO_2 emissions as CO_2 emissions related to the natural carbon cycle, as well as those resulting from the combustion, harvest, digestion, fermentation, decomposition, or processing of biologically based materials. It is worth remarking that the origin of fossil fuels, produced millions of years ago, is also biological (DOE, 2017).

The first difficulty that arises when calculating the carbon footprint of a given IWMS is the differentiation between biogenic and fossil carbon. A rigorous MFA should be performed in order to trace back the carbon source and identify the carbon sinks. Carbon (biogenic or not) may be released as an environmental burden or remain in the anthroposphere, in any of the following forms:

i) Emissions to the atmosphere. In the presence of oxygen, carbon is oxidized to CO_2. Under anaerobic conditions carbon is reduced to CH_4.
ii) Wastewater pollution and landfill leachate wherein carbon is present in a variety of organic compounds.
iii) Sequestered carbon in landfills or in soil amendment products (compost and digestate).

It must be highlighted that the distinction between an environmental burden and the accumulation of a substance in the IWMS under study is often unclear; the system boundaries need to be precisely established at the definition of the scope of the work.

Within an efficiently designed IWMS water is not considered a final carbon sink. After the adequate treatment, the carbon present in the leachate leaves the liquid phase as CO_2 or CH_4 (Wang et al., 2014), whereas the carbon in wastewater is distributed between the gaseous emissions and the sludge (Rodriguez-Garcia et al., 2012), being the latter subsequently treated as solid waste. Even though Griffith et al. (2009) estimate that up to 25% of the carbon content in wastewater is of fossil origin, it is widely assumed that the totality of carbon is biogenic, and thus it is typically not accounted for (Rodriguez-Garcia et al., 2012).

Table 1
GWP and other methodological considerations regarding biogenic carbon in the reviewed papers.

	Biogenic CO_2		Stored biogenic carbon		Specified carbon source?	Zero burden approach?
	Value	Unit	Value	Unit		
Aghajani et al. (2016)	0	kg CO_2-eq/kg CO_2	–	–	No	Yes
Blengini et al. (2012)	1	kg CO_2-eq/kg CO_2	−1	Unspecified	No	Yes
Chang et al. (2012)	0	kg CO_2-eq/kg CO_2	–	–	No	Yes
Manfredi et al. (2011)	0	kg CO_2-eq/kg CO_2	−44/12	kg CO_2-eq/kg C	Yes	Yes
Minoglou and Komilis (2013)	0	kg CO_2-eq/kg CO_2	–	–	No	Yes
Tabata et al. (2011)	0	kg CO_2-eq/kg CO_2	–	–	Yes	Yes
Turner et al. (2016)	0	kg CO_2-eq/kg CO_2	0 or −44/12	kg CO_2-eq/kg C	Yes	Yes
Vergara et al. (2011)	0	kg CO_2-eq/kg CO_2	−1	Unspecified	Yes	No
	1	kg CO_2-eq/kg CO_2	0	Unspecified	Yes	Yes

Although emissions from leachate treatments are estimated in some of the reviewed papers (Chang et al., 2012; Manfredi et al., 2011), none of them made express reference to the carbon source. The reviewed articles that accounted for biogenic CO_2 are shown in Table 1. The procedure followed to determine the carbon origin is not clearly stated in many cases. Whereas Tabata et al. (2011) and Vergara et al. (2011) consider that biogenic CO_2 is derived from the biogenic fraction of waste, only Manfredi et al. (2011) and Turner et al. (2016) explicitly consider the fraction of biogenic carbon in the input waste.

Regarding the stored carbon in landfills and the carbon emissions to the atmosphere, for the specific case in which an LCA is performed with the objective of comparing different scenarios but there is no interest in knowing the values of their individual carbon footprints, Christensen et al. (2009) proved that, provided that the assumptions concerning biogenic CO_2 emissions and carbon sequestration are consistent (considering biogenic CO_2 emissions either neutral or not neutral) and the system boundaries are clearly established, the emission ranking of scenarios remains the same.

As can be seen in Table 1, biogenic CO_2 emissions are assigned a GWP factor (expressed as kg of equivalent CO_2 per kg of emitted CO_2) of zero in most studies, which implies that no environmental impacts in terms of climate change potential are attributed to them. Applying this GWP is analogous to expanding the system boundaries to include the upstream processes of photosynthesis. Thus, unless biogenic CO_2 is being stored, the CO_2 that is captured during the growth of biomass and comes into the system, is balanced with the biogenic CO_2 that leaves the system, achieving carbon neutrality. For the sake of coherence, a negative GWP must be assigned to the carbon that is captured in the photosynthetic processes and remains sequestered in the system. Nonetheless, as Vergara et al. (2011) point out, by applying this procedure only the environmental benefits of the upstream processes are being taken into account, disregarding their environmental burdens. As a consequence, this approach might lead to higher environmental credits than burdens, entailing that landfills and soil amendment products contribute to climate change mitigation (Turner et al., 2016).

To correct this incoherence, the carbon flows that connect the system to the environment (primarily as CO_2 and CH_4) must be inventoried. If the system boundaries are expanded to include the upstream processes, once the elemental composition of the waste and products is known, the incoming carbon flows can be easily calculated: every mole of biogenic carbon present in the products, waste and emissions originates from a mole of CO_2 that was absorbed by biomass in the photosynthetic process. Afterwards, the carbon flows that come into the system must be subtracted from the carbon flows that leave the studied system.

This systematic approach allows applying the same GWP (1 kg CO_2-eq/kg CO_2) to CO_2 emissions from scenarios with different system boundaries, regardless of the CO_2 origin.

The proposed procedure, which relies on the waste composition provided by the MFA, ensures that the CO_2 removed from the atmosphere, whose carbon eventually leaves the system as CH_4, is accounted for. The studies compiled in Table 1 make no express reference to a correction in the GWP of biogenic CH_4, when in reality CH_4 constitutes a significant fraction of the outlet stream of some technologies that process biogenic waste, such as anaerobic digestion.

6.3. Accounting for uncertainty

Models aiming at describing complex systems carry a level of uncertainty whose effect on the outcome might be hard to predict without the right methodology. There are plenty of sources of uncertainty within an IWMS, such as waste composition, the efficiency of the treatment processes, the substitution ratio of virgin materials or the effect that the seasonal changes in weather may have on the waste degradation rate. For a detailed compilation of uncertainty sources, the reader should refer to Clavreul et al. (2012). However, the paramount variable with which uncertainty is associated, regardless of the complexity of the model, is waste composition.

As Laurent et al. (2014a,b) pinpointed in their review, LCA studies do not usually account for waste composition very accurately. This asseveration could be further extended to waste management models in general, even though waste composition will determine the results of the subsequent analysis, simulation or optimization, given that the available treatment options and the type and amount of emissions resulting from the different waste treatment alternatives strongly depend on the elemental composition of waste. This is the reason coupling MFA with other analysis tools is the precursor to identifying the optimal configuration of an IWMS. Nevertheless, adequately characterizing the waste composition is a difficult task because of the heterogeneity of the material flows, and it might require complex statistical analysis. Thus, representative data of the average waste composition inevitably brings uncertainty into the model.

The elements that are excluded from the analysis without a clear justification also represent a source of uncertainty. For instance, the environmental impacts related to capital goods might have a significant influence on the results of an LCA (Brogaard and Christensen, 2016), but they are often not modeled (Chi et al., 2015; Laurent et al., 2014a,b; Suwan and Gheewala, 2012).

Stochastic modeling, which relies on the propagation of probability distributions, is the most frequently deployed methodology to consider the effect of uncertainties on the LCA results, although scenario analysis is more commonly applied for the LCA of waste management (Clavreul et al., 2012). Sensitivity analysis to investigate the effects of a change on an assumption or the value of a parameter are routinely performed in many of the reviewed studies (Blengini et al., 2012; Boesch et al., 2014; Bovea et al., 2010; Chi et al., 2015; Cleary, 2012; Eriksson et al., 2005; Fiorentino et al., 2015; Fruergaard and Astrup, 2011; Giugliano et al., 2011; Jeswani and Azapagic, 2016; Koci and Trecakova, 2011; Koroneos and Nanaki, 2012; Manfredi et al., 2011; Pressley et al., 2014; Rigamonti et al., 2009; Song et al., 2013; Tonini and Astrup, 2012; Tonini et al., 2013; Turner et al., 2016; Vergara et al., 2011; Wang et al., 2015). Massarutto et al. (2011) also carried out a sensitivity analysis in their LCC analysis. Notwithstanding only three of the abovementioned studies (Pressley et al., 2014; Tonini and Astrup, 2012; Tonini et al., 2013) analyzed the impact that different waste compositions would have on the results.

Hanandeh and El-Zein (2010) considered the uncertainty related to the input waste composition, among other parameters. Comparing the results of the stochastic model of an IMWS with those of a deterministic model, they found that when uncertainty is taken into account, the environmental burdens of one of the studied impact categories became environmental credits, proving that the uncertainty of the data in their case study was definitely not negligible. However, Clavreul et al. (2012) claim that probability distributions, which are oftentimes dependent on incomplete information, should be applied cautiously. Instead, they proposed a systematic sequential approach to quantify uncertainty in LCA models of waste management systems that comprises a number of complementary methodologies for uncertainty analysis.

Regarding the quantification of uncertainty in the models aiming at optimizing IWMSs, two methodologies can be differentiated in the reviewed literature:

i) After the initial optimization of the objective functions a sensitivity analysis is performed to check the effect of a change in the input parameters or the assumptions made on the optimal solution. Tabata et al. (2011), Tan et al. (2014) and ThiKimOanh et al. (2015) apply this methodology.
ii) A methodology to quantify uncertainty is embedded in the model or the optimization technique. Table 2 compiles the modeling and optimization methodologies applied for that purpose in the reviewed studies.

As can be seen in Table 2, some studies apply a combination of techniques. Interval programming, in which uncertainties are expressed as interval values, is the most common programming technique to quantify uncertainty. Stochastic and fuzzy programming are also popular; the difference between them is that in stochastic programming uncertainty is modeled through discrete or continuous probability functions, whereas fuzzy programming considers random parameters as fuzzy numbers and constraints are treated as fuzzy sets (Sahinidis, 2004).

Finally, an approach to quantify uncertainty within MCDM models was proposed by Pires et al. (2011a). They developed a MCDM framework that integrates an interval-valued fuzzy method with the analytic hierarchy process (AHP) and the technique for order performance by similarity to ideal solution (TOPSIS) in order to help decision-makers prioritize waste management scenarios.

The extensive amount of methodologies developed to account for uncertainty makes it hard for the non-experts to choose the most appropriate one for the analysis of their IWMS. Two trends have been observed in the literature: the performance of sensitivity analysis and the combination of several methodologies. The former risks not capturing the complexity of the model, while the latter may become a time consuming process that considerably increases the researchers' effort.

In any case, a meaningful uncertainty analysis must be based on the correct identification of the parameters and assumptions that will bring uncertainty into the model, which are not always clearly listed in the reviewed studies.

6.4. Dynamic modeling

Most of the reviewed models, with the exception of multi-period optimization models (Cui et al., 2011; Dai et al., 2011; Levis et al., 2013, 2014; Li and Chen, 2011; Mirdar-Haridani et al., 2017; Srivastava and Nema, 2011, 2012; Tan et al., 2014; Zhai et al., 2016; Zhou et al., 2016; Zhu and Huang, 2011), describe static IWMSs that do not account for changes in the system variables throughout time. Oppositely, multi-period optimization models assume that the constraints and the parameters remain constant within a given time period, although they may differ between different stages. Hence, in spite of being time dependent, the outputs of these models are not a function of time, but a function of the time period. In fact, models introducing time series have been classified as *quasi-dynamic* (Lundie et al., 2007), under the argument that the results of one period do not determine the results of the next period. The implementation of dynamic models whose outputs are a function of time would bring a higher degree of complexity into the analysis; for instance, modeling the behavior of markets throughout time would add realism to an LCA, but because of the large data requirements, it is not usually considered a feasible option (Lundie et al., 2007).

Thus, the definition of time stages appears to be the most straightforward and practical route to account for the time-dependent changes in the system, such as the need to manage obsolete goods after they have provided the expected service. The shorter the established time periods, the more reliable the model will be. The time periods should be established so that the seasonal variations in waste composition are accounted for. Of the reviewed studies, only Levis et al. (2014) took into account the changes in waste composition in the studied time period. If the study aims at quantifying the environmental impacts and the consumption of natural resources of the system, successive LCAs should be performed for each time period in which the input waste composition varies. Accordingly, different functional units referring to each specific time period should be defined.

The seasonal changes in waste composition (proved for example by Castrillón et al. (2013)) pose a challenge to the design of CIWMSs, given that they must be flexible enough to adjust to the changes in the feed composition. Furthermore, since manufacturers cannot count on a steady supply of secondary materials, the fluctuations in waste composition hamper the shift to a circular economy.

It is important not to confuse the duration of the supply of goods and services provided by the system, which is identified by the functional unit, with the time horizon of the LCA (JRC, 2011), which is the time length during which the flows that connect the IWMS with the environment are accounted for. Additionally, the selected time horizon determines the value of the characterization factors used to calculate the contribution of the different substances exiting the system to each of

Table 2
Methodologies to quantify the effects of uncertainty in the reviewed optimization models.

	Fuzzy programming	Stochastic programming	Interval programming	Factorial design	Minimax regret analysis
Cui et al. (2011)			x		x
Chang and Li (2013)	x				
Dai et al. (2011)			x		
Li and Chen (2011)	x	x	x		
Srivastava and Nema (2011)	x				
Wang et al. (2012)	x	x	x		
Zhai et al. (2016)			x	x	
Zhou et al. (2016)		x			
Zhu and Huang (2011)		x			

the impact categories studied on the LCA (JRC, 2010). Thus, the time horizon must be long enough to include all the relevant emissions to the environment. This guideline is of particular interest for modeling landfills, since their emissions may prevail for a long time in the order of thousands of years (Finnveden, 1999).

For the defined time period in which a CIWMS is analyzed, certain waste fractions might travel within the system for a number of times; depending on the time at which the system is being described, some materials may be part of the waste or the products. In fact, the products into which a material is transformed might even be different if they undergo an open-loop recycling process. A methodology to calculate the average number of times a material is used was proposed by Yamada et al. (2006).

The disparities in the material flows within a given time period can only be solved by assuming that the model concerning each time period is a steady-state model; that is to say, that the incoming natural resources and the flows of waste and products within the system are constant and homogeneously distributed along the studied time period. Following this methodology, materials should be counted as both waste and products as many times as cycles they describe within the system in the defined time period.

7. Application of the cradle-to-cradle approach

The boundaries of a CIWMS do not enable to implement the traditional linear cradle-to-grave LCA; thus, a cradle-to-cradle approach must be applied. In this section the adjustments to the LCA scope that this new perspective requires will be discussed, focusing on the modeling framework, the multi-functionality problem and the definition of the functional unit, all of which are intrinsically related to one another and will be determined by the goal and scope definition.

7.1. Goal and scope definition

The goal of the LCA of a given CIWMS might differ among studies, which makes it hard, if not impossible, to compare their results. The proposed methodology discussed in this section will be coherent with this goal: to identify possible improvements in the design of a CIWMS wherein waste prevention activities are implemented, so that its environmental impacts and its consumption of natural resources can be minimized. Hence, the analysis is intended to assist the decision-makers in the design of a CIWMS.

7.2. Multi-functionality problem

The LCA practitioner might come across a multi-functionality problem: how to allocate the environmental impacts between all the functions that the system supplies if the further subdivision of the subsystems that configure the CIWMS cannot be applied to avoid allocation, because of the interconnection between them. To deal with this multi-functionality problem, two strategies, which depend on the selected modeling approach, can be applied (Finnveden et al., 2009; ISO 14044, 2006): system expansion or allocation. According to ISO 14044 (2006), system expansion should be deployed wherever possible in order to avoid partitioning the environmental burdens.

Most studies analyzing IWMSs apply the direct substitution (also called avoided burden) method (Abeliotis et al., 2012; Al-Salem et al., 2014; Evangelisti et al., 2015; Antonopoulos et al., 2013; Belboom et al., 2013; Blengini et al., 2012; Boesch et al., 2014; Bovea et al., 2010; Chi et al., 2015; Dong et al., 2014; Eriksson et al., 2014; Fiorentino et al., 2015; Fruergaard and Astrup, 2011; Gentil et al., 2011; Giugliano et al., 2011; Jeswani and Azapagic, 2016; Manfredi et al., 2011; Menikpura et al., 2012; Menikpura et al., 2013; Montejo et al., 2013; Pandyaswargo et al., 2012; Pires et al., 2011b; Pressley et al., 2014; Rada et al., 2014; Rigamonti et al., 2013; Suwan and Gheewala, 2012; Tonini and Astrup, 2012; Tonini et al., 2013; Tulokhonova and Ulanova, 2013; Tunesi, 2011; Turner et al., 2016; Vergara et al., 2011; Wang et al., 2015); they consider that the primary aim of their system is to treat waste, and they expand the system boundaries to include within the system the other products and services supplied, like materials and energy, and subtract their environmental impacts from those of the original system. However, a CIWMS does not operate under the assumption that waste needs to be treated in order to minimize its negative impacts, but valorized, so that the consumption of natural resources is reduced.

7.2.1. Functions of a CIWMS

According to the system boundaries set in Fig. 3, the functions fulfilled by a CIWMS are twofold:

i) To supply the services that society demands, regardless of the origin of the raw materials.
ii) To exploit the maximum amount of the generated waste, by either producing new products from it or recovering its energy, with the ultimate goal of minimizing the consumption of natural resources.

The second function is a consequence of the first one, and the first one can be partially achieved due to the accomplishment of the second function. However, if waste upgrading and energy recovery processes were not implemented, the supply of the services demanded by society could still meet the demand, relying solely on the extraction of natural resources. Thus, it can be agreed that the primary function of a CIWMS is waste exploitation.

According to the definition of the system functions, it is not necessary to disaggregate any of them by the type of services and products provided in order to solve the multi-functionality problem. This way, the uncertainty brought into the model by the choice of the allocation procedure is reduced. Moreover, the problem of allocation in open-loop recycling, which is a recurrent discussion in the LCA literature (Ekvall, 2000; Ekvall and Finnveden, 2001; Finnveden, 1999; Yamada et al., 2006; Shen et al., 2010), is avoided.

7.2.2. System expansion approach

If the LCA practitioners are interested in analyzing the overall environmental impacts of the whole system, the system expansion approach must be followed. The studied CIWMS should be compared to a functionally equivalent system whose functions are provided by alternative subsystems (Finnveden, 1999); for instance, a linear IWMS that depends exclusively on virgin raw materials.The environmental benefits of the complete CIWMS could be estimated as the difference in the environmental impacts of the linear and circular IWMSs.

If on the contrary, the study aims at investigating the environmental impacts derived from the primary function of the CIWMS, the direct substitution or avoided burden approach could be applied by expanding the system boundaries to include alternative subsystems responsible for the secondary function, based entirely on virgin raw materials. Their environmental impacts should be subsequently calculated and subtracted from the environmental impacts of the studied CIWMS. Accordingly, the resulting environmental impacts are assumed to be due to the primary function of the system. This might result in overall negative environmental impacts and, as a consequence, the system could be mistaken for an environmental burdens sink.

If system expansion is applied, a choice between marginal and average data must be made to model the system functions. Marginal data is used to model systems whose outputs change in response to decisions regarding the life cycle of the system under study, for example a decrease in the demand for the electricity produced from natural gas as a consequence of the supply of electricity from waste-to-energy processes. Average data, on the other hand, represents the mean data in a region; the average electricity mix refers to the grid mix of that region, and it does not reflect any changes in fuel consumption because of the changes in the electricity demand. Although average data might lack accuracy, it is more appropriate if the effects that the decisions taken have on the surrounding systems are not certain. The selection of the data is closely related to the LCA modeling framework applied. Whereas "attributional LCA focuses on describing environmentally relevant physical flows to and from a life cycle, consequential LCA aims at describing how the environmentally relevant physical flows to and from the life cycle will change in response to possible decisions" (Finnveden et al., 2009).

7.2.3. Allocation approach

Heijungs and Guinée (2007) are firm advocates of allocation procedures because the assumptions on which the direct substitution approach is based are likely to introduce considerable uncertainty into the model. Whereas they recognize that the allocation approach is subject to essentially arbitrary allocation factors, they argue that it is extremely hard to predict what system would be affected if the secondary function of the studied system was meant to replace one of the functions of another system, and up to what extent the environmental impacts caused by the other system would be avoided. Although the selection of a 100% substitution ratio is common, several authors suggest that a complete displacement is unlikely (Geyer et al., 2016; Vadenbo et al., 2017; Zink et al., 2016, 2017).

In addition to that, if the substituted function was produced in a multi-functional system, the system boundaries would have to be further expanded until mono-functional systems were found, significantly increasing the complexity and the uncertainty of the system. Ekvall and Finnveden (2001) also acknowledged the importance of the uncertainty caused by system expansion; they stated that system expansion is an adequate procedure to solve the multi-functionality problem as long as data for the competing production of the secondary function is available, and the data uncertainties are not too large, which agrees with the guidelines of ISO 14044 (2006).

This argument can be easily extrapolated to the case of a CIWMS aiming at valorizing MSW. The resources transformation subsystem, responsible for the secondary function of a CIWMS, comprises many production subsystems; modeling the alternative processes relying on virgin raw materials would bring multiple sources of uncertainty into the model, not to mention that it would be an extremely time consuming task.

If an allocation procedure is selected to solve the multi-functionality problem, it must be borne in mind that except when physical causal relationships are deployed as a basis for allocation, the property according to which the allocation is performed depends entirely on the choice of the LCA practitioner.

The chemical composition of the flows within a CIWMS, determined by the MFA, is a valid causal criterion to allocate the input-specific environmental impacts. However, given that the composition of the recycled materials should be, a priori, identical to the composition of the virgin materials, this criterion could only be applied in the cases wherein either the recycled materials carry pollutants accumulated in the recycling process, or certain materials cannot be recycled and thus the environmental impacts derived from the processing of those materials are due to the incoming virgin materials into the system. Furthermore, the environmental impacts caused by the process specific emissions, such as dioxins and furans produced in the incineration processes (Margallo et al., 2014), which are dependent on the operating conditions and the applied technologies, cannot be allocated according to the chemical composition of the input flows.

Hence, a different allocation factor that enables to partition all the environmental impacts between the system functions must be defined. There are basically two types of approaches to perform the allocation of environmental impacts in the cases wherein causal relationships cannot be found, those relying on a physical parameter, such as mass or volume, and those that are based on socioeconomic criteria. Even though both approaches are internally consistent as long as the selected physical property or socioeconomic indicator is also applied to quantify the performance of the system and used to calculate the functional unit, different results will be obtained for different allocation factors, and they might show opposite trends. Therefore, the choice of the allocation factor should never be made based on an arbitrary decision, it should respond to the goal and scope of the LCA instead (Pelletier et al., 2015).

One of the reasons for not including socioeconomic parameters in the LCA is that if more than one of the sustainability dimensions (economy, environment and society) are studied jointly, some of the trends in the results might be overlooked. For instance, the objective of the study of the carbon footprint of a CIWMS wherein the functional unit is defined as the revenues generated in a given time period, could be to detect what changes in the configuration of the CIWMS would result in a minimization of the ratio kg CO_2-eq/€. Expressing the results as a ratio between those two variables might make it harder to identify if only the environmental impacts, only the economic revenues or both the environmental impacts and the economic revenues are improved as a consequence of a change in the technical parameters of the system.

Moreover, since the goal of the LCA was defined at the beginning of this section from a technical perspective, making no reference to economic criteria, a physical parameter is more appropriate to allocate the environmental impacts. The different material fractions emerging from the materials sorting subsystem will be transformed into a variety of goods and services, which hinders the selection of a single allocation factor based on a physical property that enables to assess the multiple functions of the goods and services delivered. Nonetheless, the mass of waste before it has been transformed into products or supplies any services could be viewed as an indicator of its potential. Hence, mass seems to be the most appropriate physical parameter to perform the allocation of the environmental impacts of a CIWMS.

In the context of a CIWMS, MSW is a substitute for natural resources; in particular, for raw materials. If the amount of energy, materials and products derived from waste that enter subsystem 1 rises, the incoming raw materials to subsystem 0 decrease in order to maintain the functions delivered by the CIWMS constant. Therefore, the allocation factor of the environmental impacts to the primary function of the system (*AF*) could be defined as the ratio between the mass of the MSW that is valorized in subsystems 6 and 7 ($MSW_{6,7}$), and the mass of raw materials *(RM)* and the valorized MSW, as shown in Eq. (1).

$$AF = \frac{MSW_{6,7}}{RM + MSW_{6,7}} \tag{1}$$

7.2.4. Summary of approaches to solve the multi-functionality problem

The LCA practitioner should ponder the disadvantages of each approach and apply the one that fits the best the goal of the study and the data availability. Table 3 sums up the main disadvantages of the application of the different methodological approaches to the LCA of a CIWMS.

Table 3
Summary of the drawbacks of alternative methodological approaches.

			Attributional	Consequential
Allocation	By mass		a	Not applicable
	By economic value		a, b	
System expansion	Average data	Comparison	c, e	Not applicable
		Substitution	d, e	
	Marginal data	Comparison	Not applicable	c
		Substitution		d

a. Consequences on the exported functions of alternative systems not considered.
b. Hard to separately identify the response of revenues and environmental impacts to changes in the IWMS.
c. Environmental impacts of the overall system; specific environmental impacts of the primary function not known.
d. Negative results not coherent with waste prevention activities.
e. Data uncertainty modeling alternative processes.

7.3. Functional unit

Regarding the functional unit, it must describe the performance of the CIWMS in terms of the fulfillment of the primary function of the system; its aim is to quantify the performance of a system so that it can be used as a reference unit (ISO 14040, 2006).

Two thirds of the reviewed LCA studies deployed a round functional unit (1 ton or 1000 tons of MSW), which, as highlighted by Laurent et al. (2014a,b), simply quantifies a waste flow, without describing the performance of the IWMS. On the other hand, the functional unit of several of the reviewed studies was the incoming amount of waste into the system. Notwithstanding, the shift in the perspective of the analysis from waste (in a typical linear IWMS) to resource (in the defined CIWMS) should be reflected on the functional unit. Therefore, since the ultimate goal of a CIWMS is to reduce the extraction of raw materials, the mass of the incoming raw materials into the system could be accounted for in the definition of the functional unit of a CIWMS.

Furthermore, if waste prevention activities are considered one of the targets of a CIWMS, the amount of raw materials prevented as a consequence of the waste prevention activities should also be taken into account in the definition of the functional unit, so that scenarios with and without waste prevention activities can be compared on the same basis.

Thus the functional unit of a CIWMS could be defined as the sum of the incoming raw materials into the system in the selected time period and in a given region plus the amount of raw materials that would have been consumed if waste prevention policies had not been implemented in that time period in that geographic area.

These recommendations are provided for a generic CIWMS that manages the variety of materials present in MSW. The discussion would be different if the system under study aimed at valorizing a specific type of waste and sending it back to the subsystem where it was generated. In this scenario, the selected functional unit could be a parameter different from the mass of the raw materials that reflects the precise primary function of the system.

Taking a CIWMS that focuses on the management of food waste as an example, its functions are to provide food for the population of a given region, and to valorize the generated organic waste into a fertilizer that is looped back into the food production subsystem. One parameter that could quantify the primary system function (waste valorization into a fertilizer) better than the incoming mass of raw materials into the system would be the area of land that is fertilized.

8. Conclusions

Based on the insights gained in the literature review, it was concluded that some of the shortcomings that applying the current methodological approaches to a CIWMS would entail could be solved by expanding the boundaries of a traditional linear IWMS to include upstream subsystems that link the transformation of raw materials into MSW with the waste treatment subsystems. This approach is also helpful to the analysis of waste prevention activities and the quantification of the biogenic carbon present in waste.

Waste composition will determine the functions fulfilled by the CIWMS. A CIWMS managing mixed MSW could deliver materials, energy, nutrients and even chemicals. Because of the wide range of technologies that each waste fraction can be subjected to, mathematical programming and MFA are essential to the design of CIWMSs. However, these techniques must be combined with system assessment tools, such as LCC and LCA.

Unarguably, the benefits derived from the implementation of CIWMs are due to the reduction in the consumption of natural resources. However, the economic and environmental benefits of CIWMSs are not self-evident and need to be proven by an in-depth analysis.

One of the challenges of performing the LCA of a given CIWMS lies on the multiplicity of materials that the system can handle, which translates into the great variety of services supplied and makes it hard to select the functional unit, which should reflect the shift in the perspective of the analysis from waste to resource.

Nonetheless, the main difficulty that will arise from the recommended approach will probably not stem from the integration of different methodologies, but from the upstream subsystems; considering their large size, their detailed analysis will increase the complexity of the model and the researchers' efforts needed in the modeling phase.

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Appendix A

See Table A1.

Table A1
Reviewed studies and applied methodologies.

Reference	Methodology
Abeliotis et al. (2012)	LCA
Aghajani et al. (2016)	MCDM
Akbarpour Shirazi et al. (2016)	Optimization
Allesch and Brunner(2014)	Review
Antonopoulos et al. (2013)	LCA
Arena and Di Gregorio (2014)	MFA and SFA
Belboom et al. (2013)	LCA
Blengini et al. (2012)	LCA
Boesch et al. (2014)	LCA
Bovea et al. (2010)	LCA
Chang et al. (2011)	Review
Chang et al. (2012)	LCA and optimization
Chang and Lin (2013)	Optimization
Chi et al. (2015)	LCA
Consonni et al. (2011)	Review

(continued on next page)

Table A1 *(continued)*

Reference	Methodology
Consonni and Viganò (2011)	Material and energy analysis
Cui et al. (2011)	Optimization
Dai et al. (2011)	Optimization
Eriksson and Bisaillon (2011)	LCA
Eriksson et al. (2014)	LCA and financial cost calculation
Erses Yay (2015)	LCA
Falzon et al. (2013)	LCA
Fernández-Nava et al. (2014)	LCA
Fiorentino et al. (2015)	LCA
Ghiani et al. (2014)	Review
Giugliano et al. (2011)	LCA
Herva et al. (2014)	EFA, MFA and Ecological footprint
Ionescu et al. (2013)	Environmental indicators
Jovanovic et al. (2016)	LCA and MCDM
Juul et al. (2013)	Review
Karmperis et al. (2013)	Review
Koci and Trecakova (2011)	LCA
Koroneos and Nanaki (2012)	LCA
Laurent et al. (2014a,b)	Review
Levis et al. (2013)	LCA and optimization
Levis et al. (2014)	LCA and optimization
Martinez-Sanchez et al. (2017)	LCA and optimization
Li and Chen (2011)	Optimization
Massarutto et al. (2011)	LCC
Menikpura et al. (2012)	LCA, economic and social assessments
Menikpura et al. (2013)	LCA
Minoglou and Komilis (2013)	Optimization
Mirdar Harijani et al. (2017)	Optimization and social LCA
Münster et al. (2015)	LCA and optimization
Ng et al. (2014)	Optimization
Niziolek et al. (2015)	Optimization
Pandyaswargo et al. (2012)	LCA
Pires et al. (2011a)	LCA
Pires et al. (2011b)	Review
Pressley et al. (2014)	LCA
Rada et al. (2014)	LCA
Rigamonti et al. (2013)	LCA
Rigamonti et al. (2016)	Materials recovery, energy recovery and costs indicators
Sadhukhan et al. (2016)	Techno-economic analysis
Santibañez-Aguilar et al. (2013)	Optimization
Santibañez-Aguilar et al. (2015)	Optimization
Satchatippavarn et al. (2016)	Optimization
Song et al. (2013)	LCA
Srivastava and Nema (2011)	Optimization
Srivastava and Nema (2012)	Optimization
Suwan and Gheewala (2012)	LCA
Tabata et al. (2011)	LCA and optimization
Tan et al. (2014)	Optimization
ThiKimOanh et al. (2015)	Optimization
Tonini and Astrup (2012)	LCA
Tonini et al. (2013)	LCA
Tonini et al. (2014)	MFA, SFA, EFA, optimization
Tulokhonova and Ulanova (2013)	LCA, economic and social assessments
Tunesi (2011)	LCA
Vadenbo et al. (2014a,b)	MFA, LCA, optimization
Wang et al. (2012)	Optimization
Wang et al. (2015)	LCA
Zaccariello et al. (2015)	MFA and efficiency indicators
Zhou et al. (2016)	Optimization
Zhu and Huang (2011)	Optimization

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