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Research Paper

Life cycle assessment of biowaste and green waste composting systems: A review of applications and implementation challenges

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ABSTRACT

Composting is one of the most widely applied methods for recycling organic waste. This process has been proposed as one option that facilitates the reincorporation of materials into the production cycle. However, composting also generates environmental impacts. Life Cycle Assessment (LCA) is the most common approach to evaluate the environmental impacts of a process at different system stages. Nevertheless, applying LCA in composting facilities is challenging due to the extensive information required, the lack of standardization on the initial assumptions, the definition of system boundaries, and the high diversity of existing composting technologies. This paper systematically reviews LCA studies in biowaste and/or green waste composting. The study highlights the challenges that should be met in order to improving the application of LCA to evaluate the environmental impacts of this type or waste treatment strategy. The review protocol used identified 456 papers published between 2010 and 2022. After the screening, 56 papers were selected, read, and thoroughly analyzed. The results show that: i) about 68% of the studies aimed to compare composting with other solid waste management options; ii) there was a wide diversity among the impact categories considered, which predominantly included climate change and ozone depletion; iii) there was no consensus on the functional unit or the system boundaries; iv) the main gaseous emissions studied were ammonia, methane, and nitrogen oxide, which were generally determined by emission factors; v) the avoided environmental impacts associated with the end-product quality and its application as an organic amendment or soil improver were ignored. This work demonstrates the complexity of conducting credible and valid composting LCA studies and proposes seven recommendations for improving the application of this assessment methodology to analyze this waste management alternative.

1. Introduction

Green waste (GW) and biowaste (BW) represent an important fraction of municipal solid waste (MSW) (Thi et al., 2015; Zhang et al., 2021). According to Lu et al. (2020), the organic fraction makes up 46% of the total MSW globally and up to 64% of the MSW in low-income countries. Disposing BW and GW in dumps or sanitary landfills results in the generation of high-strength leachate and contributes to the emission of greenhouse gases that can cause adverse environmental and public health effects (Oviedo-Ocaña et al., 2016). Proper waste management can contribute to a more sustainable use of Earth's resources, better environmental protection, and reduced climate change emissions (Jensen et al., 2016).

Composting is commonly suggested as an ideal treatment method for organic waste, as it complies with circular economy strategies thanks to the possibility of material recovery (Oldfield et al., 2018; Zhang et al., 2021). Composting is the aerobic biological decomposition and stabilization of organic waste, which was accurately defined by Haug (1993), thirty years ago. During composting, the main process parameters (i.e., porosity, moisture, interstitial oxygen, temperature, C/N ratio, availability of nutrients, pH and biological activity) play an important role in the process performance, as it has been extensively review by Sayara et al (2020) and by Cerda et al. (2018) and Reyes-Torres et al. (2018), which are specific compilations of all the issues related to biowaste and

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green waste composting, respectively., If all these parameters are controlled and evolve properly, composting results in a suitable product for agricultural and horticultural use and erosion control (Saer et al., 2013). Saer et al. (2013) highlighted the environmental benefits of composting for improving soil quality, including i) enhanced carbon storage capacity in the soil, thus, reducing global warming, ii) reducing the need for fertilizers, pesticides, and peat use, iii) improvements in soil structure, density, and porosity, which increases water retention capacity and reduces erosion and nutrient leaching, and iv) incorporation of organic matter, nutrients and electrolytes into the soil. The incorporation of organic matter includes the formation of humic acid, which is known to improve soil quality due to a complex interplay with plants and microbes (Ampong et al., 2022). It is important to mention, according to these benefits, that a complete LCA should cover not only the biological issues related to composting as a biological process or compost as organic fertilizer, but the potential impacts of the entire organic matter collection, conditioning of organic matter prior to composting or compost refining and application, as well as transportation of organic waste and compost and composting plant building and maintenance. In fact, even the type of composting plant can have an important effect on the environmental performance of composting, in relevant aspects such as gaseous emissions (Colón et al., 2010). For instance, Cadena et al. (2009a) revealed the results obtained when investigating two typical composting systems such as in-vessel reactor and turned piles, with different environmental impacts on critical points such as gaseous emissions (directly measured), energy, or water consumption. Colón et al. (2012) highlighted these differences again in a highly-detailed paper where four full-scale biowaste composting plants were environmentally assessed, including home composting. Unfortunately, as commented later, it is difficult to find papers covering the entire process or comparing different composting plants or scenarios (Martínez-Blanco et al., 2010).

Conversely, composting can have negative environmental impacts, such as carbon dioxide (CO_2) emissions from fossil fuel use in transportation and processing equipment. Due to oxygen depletion, further emissions can be generated, such as methane, nitrous oxide, and ammonia, which are also greenhouse gases and potential odor sources (Saer et al., 2013; Oliveira et al., 2016). Therefore, it is critical to evaluate the impacts of composting using sustainability assessment methods and tools (Weligama Thuppahige et al., 2022).

Life cycle assessment (LCA) is considered a holistic approach that covers the life cycle of a product or process from the cradle to the grave. This technique has been extensively used at different stages of waste treatment technologies (Yadav and Samadder, 2018a) to identify and evaluate environmental impacts such as global warming, acidification, ecotoxicity, and human toxicity. LCA could help to select the processes with lower environmental impacts and avoid the activities with higher environmental impacts (Dastjerdi et al., 2021). According to Zhang et al. (2021), LCA studies applied to solid waste management systems typically have three objectives: i) analyze the environmental performance of specific technologies, ii) compare different waste treatment alternatives, and iii) provide practical modifications of existing treatment processes to minimize related environmental impacts. Regarding organic waste composting, the LCA approach identifies the processes and stages with the most significant environmental impacts, including the impacts associated with the collection of organic waste, production, and distribution of compost, and its use as soil conditioner (Saer et al., 2013).

Different studies have applied LCA to analyze BW and/or GW composting (Bong et al.,2017; Abeliotis et al., 2012; Jensen et al., 2016; Oliveira et al., 2016; Martínez-Blanco et al., 2010). These studies show that the environmental impacts of aerobic composting are very sensitive to compost facility management practices for maintaining aerobic conditions (e.g., the technology used and operation conditions). Therefore, local circumstances, data sources, subjective assumptions by different researchers, and other influencing factors lead to different LCA results even across studies with common objectives (Zhang et al., 2021). In summary, applying LCA in composting is challenging due to the extensive information required, the lack of standardization on the initial assumptions, the definition of system boundaries, and the high diversity of existing composting technologies.

On the other hand, several review studies addressed the application of LCA in MSW management (e.g., Laurent et al., 2014a, 2014b; Schott et al., 2016; Yadav and Samadder, 2017; Bernstad and la Cour Jansen, 2012; Morris et al., 2013; Yadav and Samadder, 2018a). However, these studies focused on a particular waste (e.g., solid or food) or waste treatment system (Khandelwal et al., 2019a). None of these studies systematically review the LCA application to BW and/or GW composting.

To bridge this gap, this research systematically reviews scientific literature regarding LCA on GW and/or BW composting. This review identifies differences, uncertainties, and the lack of descriptive data in existing LCA studies. The study highlights the challenges to the robust and reliable application of this environmental assessment approach to analyze BW and/or GW composting processes. To the best of the authors knowledge and despite a tremendous amount of existing literature on composting, the present work is the first to develop an integrated analysis of multiple LCA studies on biowaste and green waste composting and propose recommendations to improve its implementation.

At this point, it is also important to highlight what LCA studies on biowaste and green waste composting do not consider in the assessment. The most important issue, in quantitative terms, is the benefits of compost use. It is true that some LCA studies consider compost as a substitute of chemical fertilizers as it has a certain content of NPK (Quirós et al., 2014a). However, this underestimates other benefits related to the presence of stabilized and fertile organic matter, such as reducing pesticide use, improving soil tilth and workability, or higher carbon sequestration (Favoino and Hogg, 2008). Other types of compost uses, such as soil bioremediation, in landfill covers or as biofilter media are typically not considered in LCA studies. It is evident that these are particular situations, but they are resulting in a large number of experiences and publications (Sayara et al., 2020).

2. Methodology

2.1. Literature search

A systematic review method was considered the most suitable to evaluate LCA studies of BW and/or GW composting since this is a sound approach to searching, selecting, analyzing, and synthesizing research evidence (Dastjerdi et al., 2021). This review included the definition of a search protocol, identifying keywords, and available information sources. For this, we chose the Scopus® academic database.

The search focused on scientific research articles using the following protocol: (i) publication years between 2010 and 2022 (February); (ii) the title and abstract should include the keywords “green waste” or “biowaste” AND “composting” AND “LCA” OR “life cycle analysis”; (iii) only scientific indexed articles; (iv) keywords should contain at least one of the words: “composting”, “green waste”, “biowaste”, “LCA”, “life cycle analysis”, “environmental assessment”, “environmental impact”. This specific search resulted in 127 papers.

2.2. Screening and selection criteria

A screening process followed the literature search to narrow the relevant articles according to their research focus. The following parameters were considered:

a) Title and Abstract: papers with a title and/or abstract that lacked relation to GW or BW composting and LCA were excluded despite the inclusion of the selected keywords.

b) Abstract: Each abstract was read to verify that the work addressed LCA studies of GW and/or BW composting.

c) Content: the entire article was read to identify the studies on LCA applications in BW or GW composting, including information such as methodology, substrate characteristics, impact categories, goal and functional unit, system boundaries, technology, and environmental impacts. The studies that included information on one or more of the previous topics, were selected for analysis. The composting systems considered included both industrial and domestic (i.e., home composting) applications. Similarly, studies comparing composting systems with other organic waste treatment options were included. Studies in which composting was a complement to anaerobic digestion (e.g., digestate composting) were discarded.

2.3. Organization and data structure

An Excel® database was used to organize the bibliography of selected articles, facilitate the search of the different research units according to categories (e.g., authors, journals, publication year, and keywords), and classify the articles into folders according to topics. The supplementary material includes the database (Tables S1 and S2).

2.4. Review and analysis of documents

The Excel® database included information from the articles that addressed LCA on BW and/or GW composting. The following data were extracted from each article: (i) Objective; (ii) LCA Methodological approach; (iii) LCA Software; (iv) Impact categories; (v) Goal and Functional Unit; (vi) Description of systems boundaries; (vii) Data sources (i.e., primary or secondary data); (viii) Information on substrates (e.g., BW, GW, municipal solid waste), technology (process type), emissions and treatment of emissions (e.g., leachate and gases), and energy consumption; (ix) Information on offsets and substitutions rates for electricity, soil improvers, and carbon sequestration/storage; (x) Sensitive variables and assumptions that might drive the particular LCA's results; (xi) Data on the end-product quality.

The information was tabulated and graphed. Each topic of items (i) to (xi) was analyzed, and trends were identified. In addition, each analyzed topic was compared to information reported by the scientific literature. Based on the analysis, challenges and recommendations for applying LCA to evaluate BW and/or GW composting systems were proposed.

3. Results and discussion

3.1. Research trends and current status

Fig. 1 shows the final number of articles considered in this study. From the initial Scopus® report (including 456 articles), only 56 articles (12.2%) were included for further analysis. Of the 456 documents that met the search criteria, 43 were discarded because they were not articles (e.g., conference papers, book chapters, or other reports). Subsequently, 37 documents were discarded for being outside the analysis period defined for the study (2010—2022). Furthermore, articles that did not include the keywords "composting", "green waste", "biowaste", "LCA", "life cycle analysis", "environmental assessment", and "environmental impact" were discarded. The remaining 127 articles were checked to determine if they met the review purpose. As shown in Fig. 2, from 2016 to 2021, between 12 and 18 papers on composting processes were published. All 127 articles were also sorted according to national origin. The countries with the largest number of articles in this review were Italy (18 articles), Spain (14 articles), the United States (14 articles), China (13 articles), Thailand (9 articles), Brazil (7 articles), Canada, France and India (6 articles each) and Australia (5 articles). Most of these studies were conducted in developed countries, showing the need to collect information that facilitates the application of LCAs in developing countries. The necessity of reliable and standardized LCAs is even more apparent when considering that many of the composting facilities in developing countries are characterized by poor monitoring and a lack of information on their design, operation, and maintenance (Oviedo-Ocaña et al., 2016).

A final selection was performed on the 127 pre-selected papers, since only a fraction of these addressed BW or GW composting. 71 articles were discarded being incomplete LCA studies or addressing substrates other than BW or GW. Thus, 56 articles were included for a detailed review; of these, 25 articles performed a detailed LCA of BW and/or GW composting systems and were used to extract the critical information for the review (tabulated and graphed). The remaining (31) articles did not provide detailed information on applying LCA to evaluate green waste and/or biowaste composting systems. However, they were examined to broaden the discussion on the challenges of applying LCA in this field.

In addition to the 56 articles included, other scientific documents

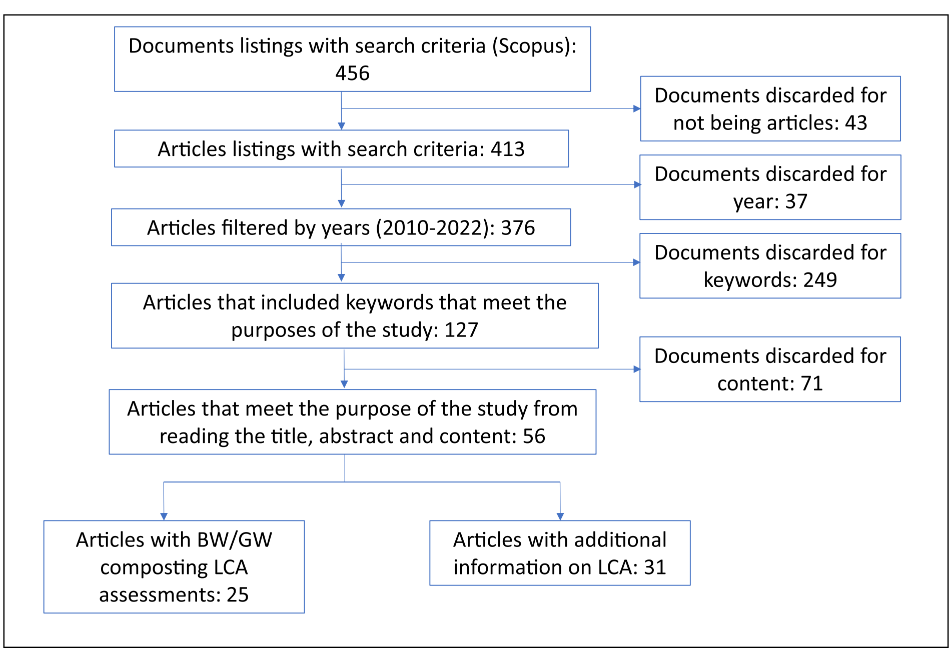

Fig. 1. Outline of the screening of articles for the literature review of Life Cycle Assessment of Biowaste and/or Green waste composting. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

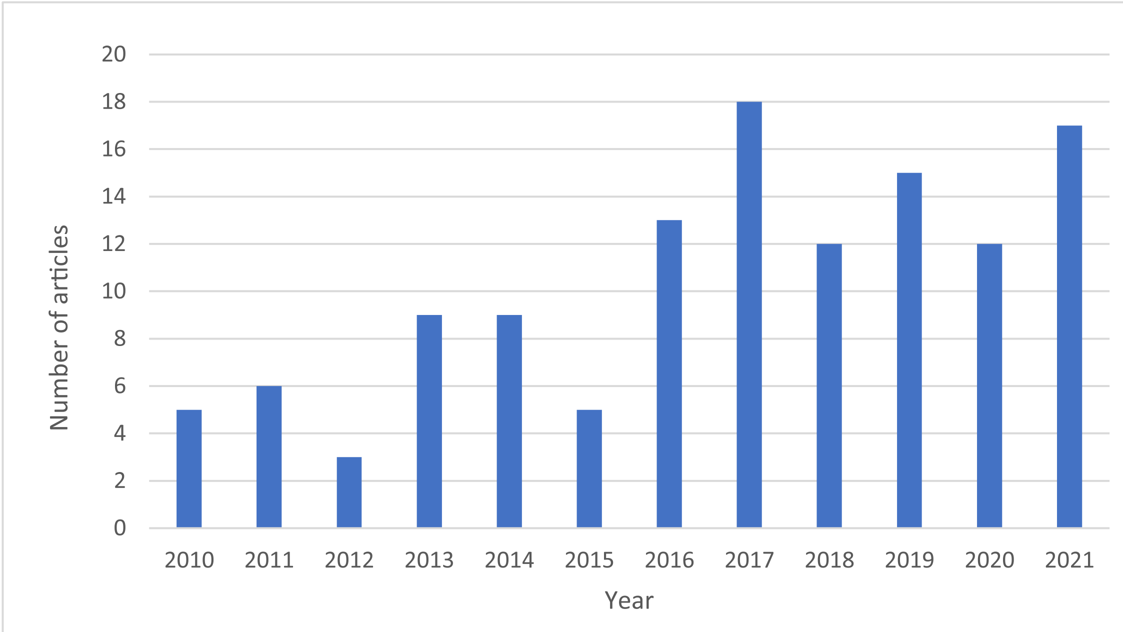

Fig. 2. Distribution of the number of relevant articles addressing the LCA of biowaste and/or green waste composting according to year (2010–2022). Note: Until February 2022, 3 articles were published. Source: Scopus® (2022). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

were consulted that describe information related to mass balances in composting processes, gaseous emissions, and relevant technical data for the environmental analysis of composting.Specific information on the 25 articles regarding the application of LCA to BW and/or GW composting is included in the supplemental material, such as: title, objective, methodology, software, impact assessment, impact

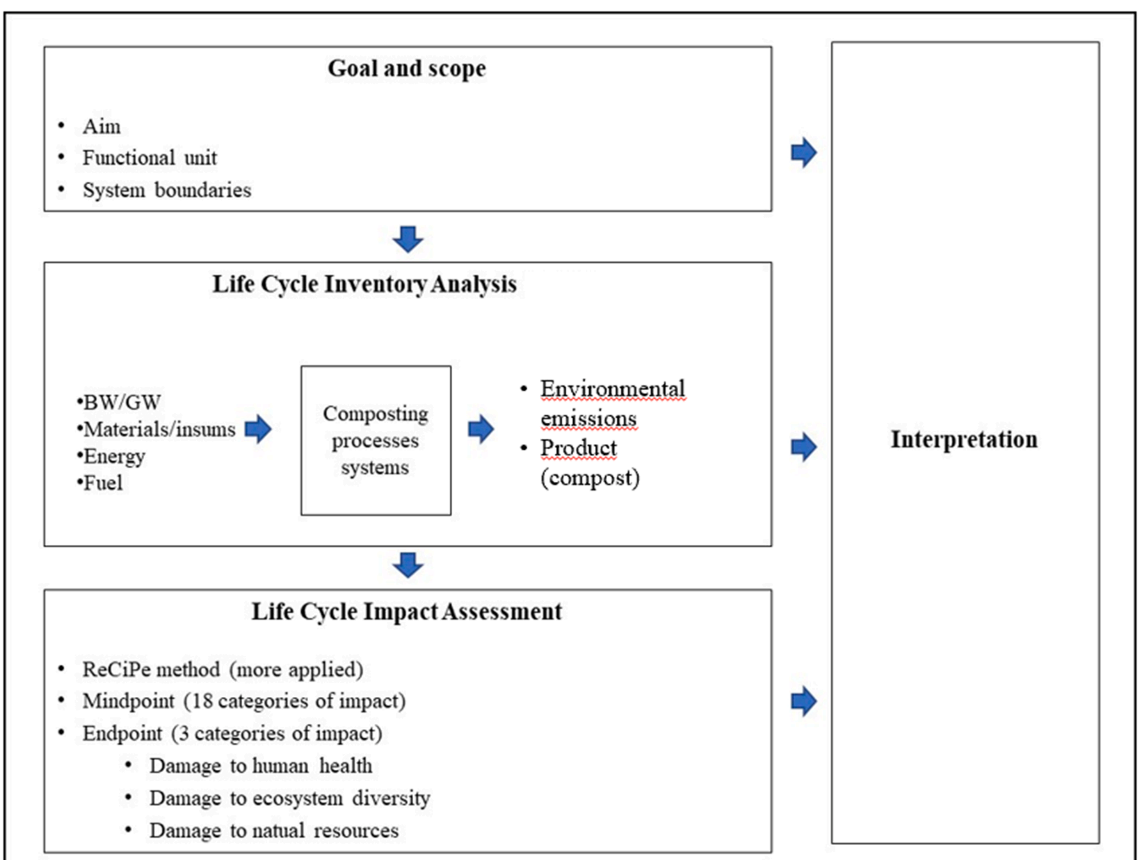

Fig. 3. Phases of the LCA of biowaste and/or green waste composting. Source: adapted from Kobayashi et al. (2020); Note: BW: biowaste; GW: green waste. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

categories, goal, and functional unit, description of the system boundaries, quality, and origin of data in the inventory, substrate, technology, quality of compost, gaseous emissions, leachate, environmental analysis, and conclusions.

3.2. Life cycle assessment of biowaste or green waste composting

3.2.1. Methodology and tools

3.2.1.1. Description of the life cycle assessment methodology. LCA is the most common approach to evaluate the environmental impacts of different waste management strategies (Komilis and Sánchez, 2017; Dastjerdi et al., 2021). According to Christensen et al. (2020), there are six waste management-associated areas where LCA is expected to play a role in the future: (i) understanding an existing waste management system; (ii) improving existing waste management systems; (iii) comparing alternative technologies (technology performance); (iv) technology development (prospective technologies); (v) policy development/strategic development, and 6) reporting.

Generally, in the reviewed studies analyzing composting processes, the LCA methodology agrees with the International Organization for Standardization (ISO) standards (ISO 14001, 14040, 14041, 14044). However, the wording in the given LCA standards is rather general and does not give detailed guidance about specific areas, such as organic waste management (Bernstad and Cour Jansen, 2012). The ISO standard defines four phases (See Fig. 3): i) goal and scope definition, ii) Life Cycle Inventory (LCI), iii) Life Cycle Impact Assessment (LCIA), and iv) interpretation. Variations observed in the studies were associated with differences in system boundaries, technological assumptions, and inventory analysis (Bong et al., 2017).

The goal and scope definition are critical steps since the appropriateness of LCA methodology depends on the purpose of the specific study (e.g., selection of functional unit or impact categories). It is also where the system boundaries (geography, time, technology, etc.) are described, and the functional unit is defined (Christensen et al., 2020). The functional unit refers to a product, a service, or a system whose impacts are addressed.

One of the most critical stages of the LCA is the life cycle inventory analysis, which involves collecting and quantifying inputs and outputs for a specific product or process throughout its life cycle (Liu et al., 2022). Common inventories for composting include transportation, operational machinery, emissions during maturation, and end-product utilization.

The methods of the Life Cycle Impact Assessment (LCIA) for characterizing, aggregating, and normalizing the inventory information into common potential impacts (e.g., global warming, acidification, and toxicity) are not specifically developed for waste management but are generally applicable (Christensen et al., 2020). In this review, all studies included used midpoint indicators to assess the environmental impact, and only 6 used end-point indicators (Abduli et al., 2011; Blengini, 2008; Buratti et al., 2015; Martinez-Blanco et al., 2010; Quirós et al., 2014b; Song et al., 2013).

In the interpretation phase, the results from the LCIA are evaluated in relation to the goal and scope of the study to reach conclusions and recommendations. Often, the results of the first modeling can be incomplete. Therefore, the input data must be adjusted to obtain a good balance and system coverage between goal and scope definition and interpretation (Christensen et al., 2020). On the other hand, LCA requires several assumptions. These assumptions significantly impact the final LCA results. Therefore, a careful examination should be conducted of crucial technical parameters affecting waste recovery and recycling processes in this step (Zhang et al., 2021). Such uncertainties may cause undesirable mistakes in decision-making. Therefore, the identification and quantification of uncertainty in the interpretation phase is essential to test the robustness and reliability or the assessment.

3.2.1.2. Tools used for life cycle assessment. Numerous tools for conducting LCA or supporting the different applications and phases of LCA exist. Several LCA computer programs have been developed to model and assess the products and processes involved. Some of these programs have focused on solid waste management. These computer-based tools help collect and analyze data, analyze waste management systems, and assess emissions and their environmental impacts (Khandelwal et al., 2019b). Although using modeling software to perform an LCA is not mandatory, this could help acquire, organize, and analyze the inventory. These tools are designed explicitly considering the LCA framework, thus facilitating systematically performing complex and lengthy calculations (Iqbal et al., 2020).

The LCA models developed to analyze waste management solutions include integrated waste management (IWM)-1 and 2, WARM (Waste Reduction Model), WASTED (Waste Analysis Software Tool for Environmental Decisions), EASWASTE (Environmental Assessment of Solid Waste Systems and Technologies), WRATE (Waste and Resources Assessment Tool for the Environment), MSW-DST (Municipal Solid Waste - Decision Support Tool). SimaPro and Gabi (Ganzheitliche Bilanzierung – holistic balancing) are other tools used, even though that are not designed specifically for analyzing solid waste management systems. The most widely used software for performing LCA studies on MSW is SimaPro, and the most employed database is Ecoinvent. SimaPro and other software usually treat the waste as separate fractions, not as a whole mass, an advantage over other commonly used software. Using an LCA modeling tool is a user choice that can vary based on the study's objectives, purchasing cost of the tool, the use of the database from the software, and the software's user-friendliness (Iqbal et al., 2020). In this review, the SimaPro software predominated (15 of 25 studies); other computer programs used were GaBi, EASWASTE, and IWM-2.

3.2.2. Goal and scope definition

3.2.2.1. Description of the analyzed composting systems. Table 1 includes the 25 articles addressing detailed LCAs of BW and/or GW composting. Type of waste, composting technology, functional unit, and system boundaries (i.e., pre-composting, composting, post-composting) are reported. In the following sections, these aspects are individually discussed.

3.2.2.2. Goal. Fig. 4 presents the general goal set in the reviewed LCA studies. It is important to note that composting is normally compared with other technologies (17 studies), which are quite different in terms of objectives and results (e.g. landfill and incineration). In this case, it performs much better when composting is compared with these other options for organic waste treatment or disposal from an environmental perspective.. In more recent studies (10), different composting approaches (e.g., industrial composting and/or home composting) are compared, which results in much more similar environmental impacts. Even though there are no many studies where composting is compared with anaerobic digestion, the preferable option is the combination of both technologies in the case of biowaste (Font and Sánchez, 2021).

3.2.2.3. System boundaries. System boundaries define the inclusion and exclusion of a subprocess or a variable from the inventory analysis and greatly influence the evaluation results (Iqbal et al., 2020). Typically, the system boundaries of a particular composting system comprise three stages: pre-, during-, and post-composting (Bong et al., 2017) (Fig. 5). However, these three stages might be addressed differently. Morris et al. (2013) performed a *meta*-analysis of 82 studies that compared multiple end-of-life management options, including composting. Comparing different strategies to address an LCA, they concluded that environmental impacts might be considered only based on emissions at the organic waste management facility. Other authors might include the impacts of organic waste collection and transport. Finally, one might

Table 1
Information about studies on the Life Cycle Assessment of Biowaste and/or Green waste composting.

Reference	Waste input[1]	Type of Technology	Functional unit	System boundaries		
				Pre-composting	composting	Post-composting
Abduli et al. (2011)	OFMSW	Windrow composting	1 ton of OFMSW	X	X	
Andersen et al. (2012)	OHW	Home composting	1 ton of OHW	X	X	X
Banias et al. (2020)	OFMSW	Not specified	1 ton of OFMSW/year	X	X	X
Blengini (2008).	BW	In-vessel composting	0.001 ton of input BW	X	X	X
Buratti et al. (2015)	OW	Enclosed agitated bed system/ outdoor aerated static piles	1 ton of OW	X	X	X
Colon et al. (2012)	OFMSW	composting in-vessel windrow composting Home composting	Reduction of 1 DRI unit in 1 ton of OFMSW		X	
Yay (2015)	KW/YW	Aerobic static pile composting	1 ton of MSW	X	X	
Jensen et al. (2016)	OHW	In-vessel and Windrow composting	1 ton of OHW	X	X	
GKhandelwal et al. (2019b)	OFMSW	windrow composting	1 ton of MSW	X	X	
Keng et al. (2020)	FW	Open-air static pile	0.2 ton/day of OW	X	X	X
Liu et al. (2022)	OSW	Static heap, windrow composting, membrane-covered composting, and reactor composting	1 ton of OSW		X	
Lu et al. (2020)	GW / FW	windrow composting, In vessel, home composting, home composting community	1 ton of OW	X	X	X
Martínez-Blanco et al. 2010	OFMSW	Home composting, in vessel and windrow	1 ton of OW	X	X	
Manfredi et al. (2011)	OW	Tunnel composting	1 ton of OW		X	X
Mu et al. (2017)	FW	In-vessel	1 ton FW	X	X	X
Oldfield et al. (2018)	FW/GW	Windrow and in-vessel composting	0.001 tons of FW	X	X	X
Oliveira et al. (2016)	MOW	Home composting	waste input	X	X	
Padeyanda et al. (2016)	FW	Not specified	1 ton of FW	X	X	X
Pergola et al. (2020)	GW	windrow composting with turning and forced aeration	1 ton of compost	X	X	X
Quirós et al. (2014b)	FW	Home composting	6.8 tons $ha^{-1}$of horticultural crops of cauliflower		X	X
Rana et al. (2019)	OFMSW	Not specified	1 ton of MSW	X	X	X
Saer et al. (2013)	FW	Windrow composting	1 ton of compost	X	X	X
Song et al. (2013)	FW	Not specified	321,752 tons of MSW	X	X	X
Yadav & Samadder (2018b)	FW/YW	Windrow composting	1 ton of MSW	X	X	X
Weligama Thuppahige et al. (2022)	OFMSW	windrow composting	1 ton of OFMSW		X	X

Note: Pre-composting: collection and pre-treatment; composting: emission composting, energy usage, water usage, emission water, leachate; Post-composting: fertilizer replacement, peat replacement, product distribution, post-application.
[1] DRI: Dynamic Respiration Index; FW: food waste; GW: Green waste; KW; kitchen waste; YW: Yard waste; MOW: Municipal organic waste; MSW: Municipal solid waste; OFMSW: Source-separated organic fraction of the municipal solid waste; OHW: Organic household waste;

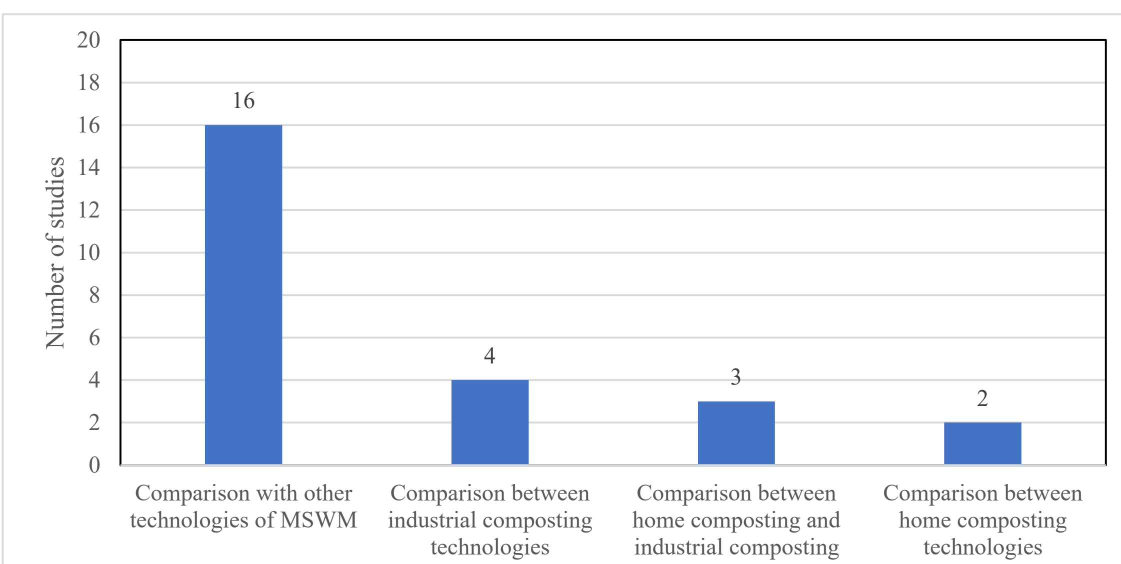

Fig. 4. General goal of LCA of biowaste or green waste composting (n = 25 studies). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

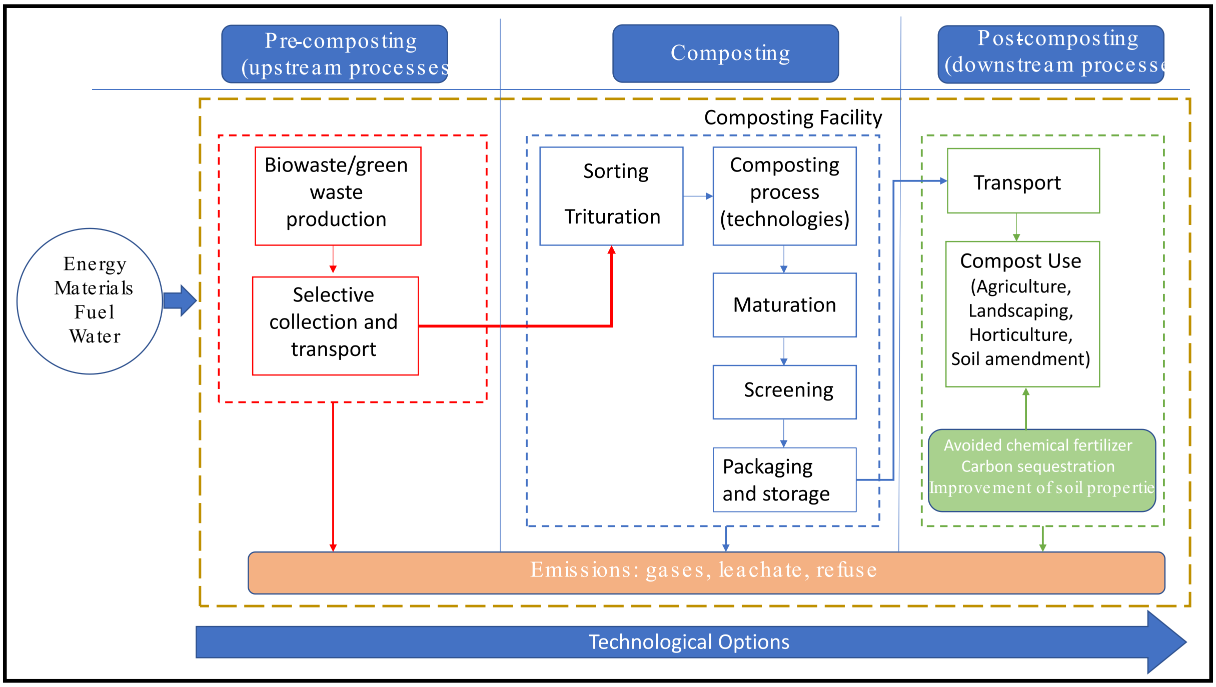

Fig. 5. System boundaries typically considered in LCA of biowaste and green waste composting systems. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

consider implications such as those from compost applied to soil (i.e., agriculture, horticulture, landscaping, or improvement soil). The degree of quantification of resource inputs, product outputs, and emissions from these processes must agree with the system boundary chosen for the study (Iqbal et al., 2020).

In comparative analyses, the system boundaries for different options must be, if not the same, at least justly comparable. When comparing different facilities, they must have similar objectives regarding the end-product (compost) quality and stability.

Composting systems have different operational phases (construction, operation, and decommissioning). Zhang et al. (2021) state that emissions in the construction and decommissioning phase are insignificant compared to the operation phase; thus, they are usually ignored in the LCA of the municipal solid waste management (MSWM) system. In addition, Buratti et al. (2015) indicate that construction, implementation, maintenance, and/or demolition (decommissioning) of the mechanical and biological plants and equipment are frequently not included due to a lack of data. In this review, all studies considered the operation phase, and only three included the construction phase (Martínez-Blanco et al., 2010; Pergola et al., 2020; Liu et al., 2022).

In agreement with the above-described findings by Morris et al. (2013), Bong et al. (2017), and Buratti et al. (2015), our review shows variations between system boundaries in studies regarding LCA on BW and/or GW composting. Table 1 shows a summary of the system boundaries of each examined study. 20 studies considered the pre-composting stage (Upstream process), 25 studies the composting stage, and 17 studies the post-composting stage (Downstream process).

In general, the active operational phase was always included. At the same time, some studies discarded pre and post-composting. According to Bong et al. (2017), researchers often exclude pre-composting activities such as waste transportation and collection. These authors assumed the travel distance for all waste management scenarios to be the same. On the other hand, some studies also exclude the benefits of post-composting, such as using compost to improve soil amendments, enhance crop health, increase water holding capacity, and reduce pesticide consumption. These types of choices are made typically on the basis of the goal of the study.

3.2.2.4. Functional unit (FU). When comparing alternative technologies, these must be well-understood, and the FU must be well-defined (Christensen et al., 2020). The FU is defined as the reference flow. It may be selected considering the waste input, product output, services provided, or the technologies being compared. The FU must be consistent with the system goal and scope, according to ISO 14,040 and 14,044.

LCA studies of waste management systems have adopted several types of FUs with different purposes (Zhang et al. (2021)). The commonly used FU in the reviewed studies is based on mass. Oldfield et al. (2018) highlight that the FU based on mass is inappropriate when a system focuses on waste valorization and circular economy. Instead, they suggest selecting a FU that reflects the downstream and secondary processing function. For example, the FU should allow the comparison of value-added products such as nitrogen (kg N), energy, or other second-generation products. Likewise, Colon et al. (2012) assessed four full-scale facilities treating source-separated OFMSW using the Respiration Index Efficiency as the novel FU instead of the classical LCA approach based on the total mass treated (i.e., reduction of 1 Dynamic Respiration Index unit in 1 Mg of OFMSW). The authors conclude that to assess the environmental impacts of biological waste treatment plants, the FU must consider the performance of the biological treatment (waste stabilization).

Table 1 summarizes the FU selected for the LCA studies on BW and/or GW composting. Most studies (23/27) use a FU based on the input mass. Three works are specific (Saer et al., 2013; Quirós et al., 2014b; Pergola et al., 2020), considering as FU the amount of composting product or productivity of a specific crop type.

3.2.3. Life cycle inventory (LCI)

3.2.3.1. Data used. The comprehensive nature of LCA studies demands data representing foreground and background processes (Henriksen et al., 2019). The foreground data refer to data required to describe a waste management system to model the environmental impacts. These data include waste composition and the total amount of waste managed in a process. The background data (or the life cycle inventory data) are the basis for assessing the environmental emissions and considering all process-relevant inputs and outputs, used resources, and energy per FU (Ikhlayel, 2018).

Information on the characteristics of composting facilities is essential for applying the LCA. However, achieving adequate life cycle inventory data has been reported as difficult in various studies. Buratti et al. (2015) mention that the LCA methodology's applicability is firmly subordinated to data available on the processes involved. Some of the required information include plant capacity, land occupation, input data (i.e., quantities and characteristics of treated waste, energy, fuel, and water consumption), and output data (i.e., quantities and characteristics of the end-product and destination, gaseous, solids and liquid emissions) (Cadena et al., 2009b; Weligama Thuppahige et al., 2022).

Some data from composting facilities can be obtained from their managers through surveys or reports, but others must be directly measured at the facility (Martínez-Blanco et al., 2010). Komilis and Sánchez (2017) indicated that original inventory data are essential for a thorough analysis. A researcher must seek these data by performing original laboratory research or field measurements. In addition, meetings and interviews with people in full-scale facilities are required to obtain data, especially about water consumption, effluent discharges, electrical energy and diesel consumption, among other data.

The studies analyzed in this review relied on mixed methods for collecting the data needed to apply the LCA (i.e., primary and secondary sources). Data on performance was obtained directly at the composting facility. In some cases, experimental tests were carried out to estimate information associated with gas emissions (Martínez-Blanco et al., 2010; Quirós et al., 2014b). In other studies (Olivera et al., 2017; Jensen et al., 2016; Behrooznia et al., 2018), emission factors were assumed from literature reports.

Despite data collection efforts, it is common that some data remain missing. Data gaps could be defined as unavailable values. However, filling the gaps would improve the accuracy and reliability of LCA results. Data gaps that are ignored, i.e., not filled by some alternative data, may bias the LCA results (Henriksen et al., 2019). Researchers frequently fill data gaps using bibliographic data. However, bibliographic data may not accurately represent the actual operating conditions of a facility (or case) under study (Komilis and Sánchez, 2017). Therefore, it is essential to consider criteria that facilitate the incorporation of relevant data when these are not available.

3.2.3.2. Substrates. The BW or GW characteristics influence the composting process, end-product quality, and emissions of gases and leachate. For example, an imbalance in the C/N ratio, depending on the process conditions, could affect the emission of gases such as ammonia (NH_3) or nitrous oxide (N_2O). However, the studies reviewed rarely report the quality of the processed waste. Only in a few cases (Saer et al., 2013; Keng et al., 2020; Oldfield et al., 2018), the physicochemical characteristics of substrates were analyzed. Most of the studies only specified the amount of waste to be processed and its main composition (i.e., yard waste, fruit waste, pruning waste, kitchen waste, among others).

On the other hand, in this review, only three studies (Pergola et al., 2020; Lu et al., 2020; Oldfield et al., 2018) explicitly mentioned the incorporation of GW. The remaining studies refer to organic solid waste, food waste, biowaste, organic fraction of municipal solid waste, source-separated organic fraction of municipal solid waste, organic household waste, and municipal organic waste.

3.2.3.3. Composting technologies. Composting can be implemented at different scales. Centralized composting (or industrial) requires waste collection and transport to a single facility. Different technologies, such as in-vessel systems, biodynamics, and windrow composting, can be applied. On the other hand, home-scale composting is an alternative to centralized composting, promoted as a simple and low-cost solution for managing household organic waste (Martínez-Blanco et al., 2010). Like centralized composting, home composting can produce a nutrient-rich humus-like material to use as a soil amendment (Lu et al., 2020).

The reviewed articles implemented LCA in BW/GW composting to evaluate the environmental impacts of this technology and other organic waste management options (See Table 1). Some studies compare the environmental impacts of industrial composting and home composting (Oliveira et al., 2016; Colon et al., 2010; Colon et al., 2012), and others compare composting with different organic waste management options (i.e., anaerobic digestion, incineration, composting, landfill) (Blengini, 2008; Oliveira et al., 2016; Jensen et al., 2016; Keng et al., 2020; Behrooznia et al., 2018, Yay, 2015). Other research compares composting technologies (i.e., static heap, windrow composting, membrane-covered composting, reactor composting) (Colon et al., 2012; Liu et al., 2022). Table 1 shows the type of composting technology considered in the different studies. Windrows, in-vessel, and home composting predominate among the composting technologies evaluated.

Equipment, aeration mechanisms, and other technological components can modify the energy requirements and the emissions generated. According to Liu et al. (2022), static heap composting transforms organic matter into compost via a natural fermentation process that can take long periods (e.g., 3 to 6 months) to be completed. However, the process can be accelerated when forced aeration and turning equipment are applied, as example in windrow composting. Although much faster, these options are associated with a strong odor, significant ammonia losses, and large nitrous oxide emissions. The membrane-covered aerobic composting technology has been verified to effectively reduce odor and greenhouse gas (GHG) emissions. Finally, in-vessel composting is a closed process integrating feeding and discharging, aeration, stirring, and exhaust gas control for aerobic digestion.

In another study, Lu et al. (2020) compare windrows and in-vessel composting. They found that in-vessel composting allowed greater control of gaseous emissions and undesirable odors than windrow composting. In addition, in-vessel composting can process the same amount of waste with less space than the window system. However, in-vessel composting requires higher technical skills and energy consumption. Although better controlled in vessel composting technologies, the emission of odors and other contaminants (e.g., volatile organic compounds) cannot be neglected.

Apart from the previously mentioned impacts, the different composting technologies have different energy requirements for waste transport and processing. Moreover, advanced systems show a higher demand for water, energy, and infrastructure (Oliveira et al., 2016; Haug, 1993). From this perspective, home composting is more environmentally-friendly and has the advantage that collection and transport of organic waste are not needed (i.e., a significant amount of GHG emissions emitted during fuel combustion, mainly fossil CO_2, can be avoided) (Oliveira et al., 2016; Lu et al., 2020). Since home composting avoids the use of transport, it is an attractive and complementary alternative to industrial composting in low-population-density areas (Oliveira et al., 2016). However, there are also some environmental concerns; for example, gas treatment systems are absent (Colon et al., 2010). Lu et al. (2020) argue that savings from avoiding waste collection and transport might not significantly improve environmental performance since the distance between the residential area and the centralized facility is another factor to consider. Regarding decentralized composting, in recent years, there has been increasing interest in community composting, including techno-economic and environmental analysis (Marcello et al., 2021). However, to our knowledge, this alternative's gas emissions or environmental impacts have not been studied yet through approaches such as LCA considering the direct measurement of GHG emissions (Sánchez, 2022).

Although various investigations have shown the reduction of gaseous emissions and odors using biofilters in composting systems (Pagans et al., 2006; Colon et al., 2009; Saraya and Sánchez, 2021), few papers evaluated the reduction of environmental impacts by the use of biofilters in LCA studies. For example, Bernstad and Cour Jansen (2012) indicated that the reduction of emissions of NH_3, methane (CH_4), and N_2O using biofilters in composting facilities has a significant impact on the "Climate Change" category (i.e., it could reach up to 25% in Global Warming Potential - GWP).

On the other hand, LCA has been used to choose solid waste management alternatives from an environmental perspective by comparing different technologies – 17 studies in this review. Bernstad and Cour Jansen (2012) compared twenty-five LCA studies, which addressed food waste treatment processes, including landfills, thermal treatment, composting (small and large scale), and anaerobic digestion. The climate change impact related to these treatment alternatives varied among the studies. Significant differences in the setting of system boundaries, methodological choices, and variations in user input data were highlighted. Therefore, it is critical to distinguish between the direct emissions generated by the composting process that are related to the biodegradation of organic matter and the volatilization of some compounds of the organic waste and the indirect ones associated with pre- and post-composting processes and the equipment necessary for these operations (mainly diesel and electricity consumption and related emissions, among others). Bernstad and Cour Jansen (2012) precisely describe these emissions sources as follows:

a) Direct emissions (directly linked to the waste management), originate from collection/transportation, treatment, and post-treatment of the waste;
b) Indirect emissions or avoided emissions (or indirect burdens or avoided burdens occurring in the background system). - Upstream indirect activities (e.g., production of materials and energy carriers used in the treatment chain or construction of machinery and treatment facilities used in the treatment) and - Downstream indirect activities (e.g., avoided emissions when substituting materials and energy carriers by activities in the waste management chain).

In summary, environmental impacts from different composting technologies are related to energy consumption, which varies across different composting technologies, operation modes, and management practices. Environmental issues such as energy consumption, gaseous emissions, leachate generation, and compost production should be analyzed to evaluate environmental performance (Bong et al., 2017).

3.2.3.4. End-product quality. Compost utilization allows diverting the organic waste fraction from landfills, recycling nutrients, and improving soil characteristics. In addition, compost utilization promotes the restoration of soil carbon by adding organic matter and the formation of humic substances, thus counteracting the continuous organic matter loss from soils due to agricultural activities (Bong et al., 2017; De Feo et al., 2016).

Many factors determine compost quality, such as waste stream composition, production management, and weather conditions. The physicochemical and biological characteristics and the heavy metals content (e.g., Zn, Cu. Ni, Cr, Pb, and Cd) in the end-product influence the agronomical and environmental performance of systems where compost is used (Quirós et al., 2014b).

Few studies (Rashid and Shahzad, 2021) analyzed the end-product characteristics and their agronomic value to estimate its use's environmental and economic benefits. Bong et al. (2017) highlight the importance of studying the relationship between end-product quality and the benefits of using compost. However, other authors (Burrati et al., 2015) assume that the environmental impacts or benefits of compost application are similar to those of other conventionally employed soil amendments and therefore ignore them.

In summary, the studies typically ignore the end-product quality or its potential use (e.g., agriculture – a different type of crop, garden, improvement soils, erosion reduction) to estimate the environmental benefits of compost (Lu et al., 2020). However, some studies use standardized compost application factors; for example, Banias et al. (2020) mentioned that 1 ton of compost is equivalen to 23 kg of N-fertilizer, 9.5 kg of P-fertilizer, and 9 kg of K-fertilizer. Weligama Thuppahige et al. (2022) indicated that the average amounts of N, P_2O_5, and K_2O in 1 ton of compost are 6.9, 1.61, and 7.3 kg, respectively. Studies are needed to delve into the relationship between end-product quality, compost use, and reduction of environmental impacts.

One of the difficulties in estimating the environmental benefits of using compost is its diverse nutrient composition and quality, notably in developing countries where the quality labeling for compost is poor. There is also limited data on the environmental performance of the compost after its application. This limitation leads to the exclusion of the post-application phase in most of the life cycle inventory analyses on composting (Bong et al., 2017).

Finally, some studies indicate that compost use generates environmental impacts. For instance, Silvenius et al. (2016) suggested that the compost application affects the emissions of gases and leachate during the life cycle of lawn areas. The impact of eutrophication from N and P leachate can also be important since compost-based substrates can be nutrient-rich.

3.2.3.5. Gaseous emissions. The composting of organic matter can generate gaseous emissions, as previously mentioned. The amount and characteristics of the gases emitted from composting processes vary and are related to the initial feedstock materials and the composting methodology adopted (Dhamodharan et al., 2019). Saer et al. (2013), Sayara and Sánchez (2021), and Pergola et al. (2020) indicated that the four main gases resulting from feedstock decomposition are CO_2, CH_4, N_2O, and NH_3 (together with volatile organic compounds - VOC, 99% of the total emission).

CO_2 emissions from the degradation of organic material are considered biogenic (and not fossil-derived). Therefore studies ignore these emissions in assessing climate change impacts (Cadena et al., 2009b; Bernstad and Cour Jansen, 2012; Saer et al., 2013; Bong et al., 2017; Manfredi et al., 2011). In addition, CO_2 emissions from produced biofertilizers later used on land are also considered carbon neutral (Bernstad and Cour Jansen, 2012).

Otoma and Diaz (2017) state that the assumption made generally in current models that the global warming potential of CO_2 of biogenic origin is zero, and that only CH_4 emissions are accounted for, is erroneous. They argue this is an erroneous assumption. Instead, they developed a model that quantifies all CO_2 and CH_4 emissions to the atmosphere from the degradation of organic Carbon (DOC) and then discounts the amount of CO_2 originating from that DOC during photosynthesis. This approach is considered more realistic to balance emissions from biogenic sources. In this regard, Christensen et al. (2009) suggest that biogenic carbon emissions can be either positive or neutral, and long-term carbon storage neutral or credit, as long as the same evaluation methodology is adopted (Iqbal et al. 2020). Therefore, the parameter choice is an arbitrary practice based on the boundary settings and assumptions of the studies. However, a transparent analysis requires a carbon mass-flow (mass balance) for all scenarios (Bernstad and LaCour Jansen, 2012).

On the other hand, there is no consensus in the literature on accounting for CH_4 emissions in LCA on composting since only some studies report methane production depending on the operating conditions of the composting process. Given the importance of CH_4 as a greenhouse gas, the selection of the CH_4 emission factor could significantly change the analysis (Saer et al., 2013; Bernstad and Cour Jansen, 2012). In this review, 15 out of 25 studies included methane emissions to estimate environmental impacts. Buratti et al. (2015) indicate that several studies ignore methane emissions, arguing that they were negligible or undetectable.

N_2O is also formed in anaerobic pockets and is mainly produced at the end of the composting process when the available carbon has been utilized without stabilization of all nitrogen (i.e., during denitrification that reduces NO_3^- to N_2O) (Boldrin et al., 2009). A poorly managed composting process can generate N_2O emissions, which have 298 times the global warming impact per unit weight of CO_2 (Saer et al., 2013). The inclusion of N_2O emissions can have a significant effect on the overall climate change impact of these systems (Boldrin et al., 2009). In this review, 15 studies included N_2O in gaseous emissions.

Similarly, NH_3 could be produced when temperatures rise above 40 to 50 °C, and therefore some studies included ammonia in assessing the impacts (i.e., 13 of 25 studies) (Saer et al., 2013). Aeration rate, pH, and initial total ammonium nitrogen content influence ammonia emissions directly in a composting process (Beck-Friis et al., 2001; Cadena et al., 2009b). According to Bernstad and Cour Jansen (2012), studies including NH_3 emissions show an increased environmental impact on acidification and eutrophication.

Another prominent group of gaseous pollutants from composting facilities is VOCs. According to Dhamodharan et al. (2019) and Cadena et al. (2009b), most VOCs in composting plants are emitted at the beginning of the process (tipping floors, shredder, and during the initial forced aeration composting period). Incomplete or insufficient aeration during composting can result in the emission of sulfur compounds with intense odor, whereas incomplete aerobic degradation processes release emissions in the form of alcohols, ketones, esters, and organic acids (Cadena et al., 2009b; Sayara and Sánchez, 2021). In the present work, seven studies reported VOCs emissions in their LCAs.

Finally, Buratti et al. (2015) consider H_2S and particulate emissions in their study. H_2S was measured over static piles, while particulates were measured in the exhaust air from the compost refining area. They found that only a few data on H_2S and particulate emissions in full-scale composting facilities are reported in the literature. The emissions estimated were 1.56 kg/ton of organic fraction for H_2S and 0.074 kg/ton of organic fraction for particulate. Fig. 6 shows gaseous emissions considered in the LCA of composting systems.

Despite the importance of estimating gaseous emissions in the composting process, actual data on gaseous emissions released from full-scale composting plants are challenging. An exhaustive sampling campaign is necessary to get representative and reliable data on a single plant (Cadena et al., 2009b). Therefore, some studies (Saer et al., 2013; Oliveira et al., 2016; Jensen et al., 2016; Behrooznia et al., 2018; Yay, 2015) applied emission factors for different gases to estimate the environmental impact through LCA (i.e., 17 studies). However, it is necessary to be careful when adopting these factors. These emission factors may be helpful in specific situations but do not provide information about biomass degradation processes. It is necessary to know if the degradation occurs under anaerobic, semi-aerobic, or aerobic conditions (Otoma and Diaz, 2017). Some examples of emission factors are:

a) Cadena et al. (2009b) reported emission factors from the biological treatment process for NH_3 and VOCs were 3.9 kg Mg $OFMSW^{-1}$ and 0.206 kg Mg $OFMSW^{-1}$ respectively (1 Mg of OFMSW was considered as the functional unit). Although in this study, the methodology developed was applied to a specific configuration of a composting plant, it can be used with different technologies and wastes.

b) Saer et al. (2013) reported emission factors of organic waste composting using windrow technology (i.e., CH_4: 0.021 – 11.9 kg/1000 kg-feedstock; N_2O: 0.003 – 0.252 kg/1000 kg-feedstock; NH_3: 0.025 – 1.3 kg/1000 kg-feedstock).

c) Weligama Thuppahige et al. (2022) adopted values proposed by the Intergovernmental Panel on Climate Change (IPCC) for aerobic composting: 4 kg CH_4 and 0.3 kg N_2O per ton of OFMSW.

Bong et al. (2017) indicate that gaseous emissions are also due to other system components (i.e., not only the composting operation: but also shredding, turning, transferring, and screening). This aspect includes fuel consumption in waste collection and transport, energy consumption (diesel or electricity) during pre-treatment, and energy consumption in tool manufacturing (e.g., composters and gardening tools). In addition, emissions savings refer to the avoided emission from the waste collection and transport to the landfill, emissions from the landfill, replacement of mineral fertilizer (MF), and peat extraction (Bong et al., 2017). Enhanced soil carbon sequestration through compost utilization was reported as the key contributor for reducing the emissions of Greenhouse Gases (GHG) (Saer et al., 2013).

In summary, activities related to GHG emissions in composting systems are: a) Upstream: waste collection, waste transportation, and materials consumption; b) Operational: screening (pre/post), shredding, turning, composter, compost packing; c) Emissions during composting of: CO_2, N_2O, CH_4, VOCs. Activities related to savings of GHG are: avoided landfill emissions, avoided transportation to landfills, avoided post-extraction, avoided use of conventional soil amendments, and carbon sequestration (Bong et al., 2017).

Compost post-application could also contribute to the GHG emissions. During the cultivation phase, the compost is transported to the

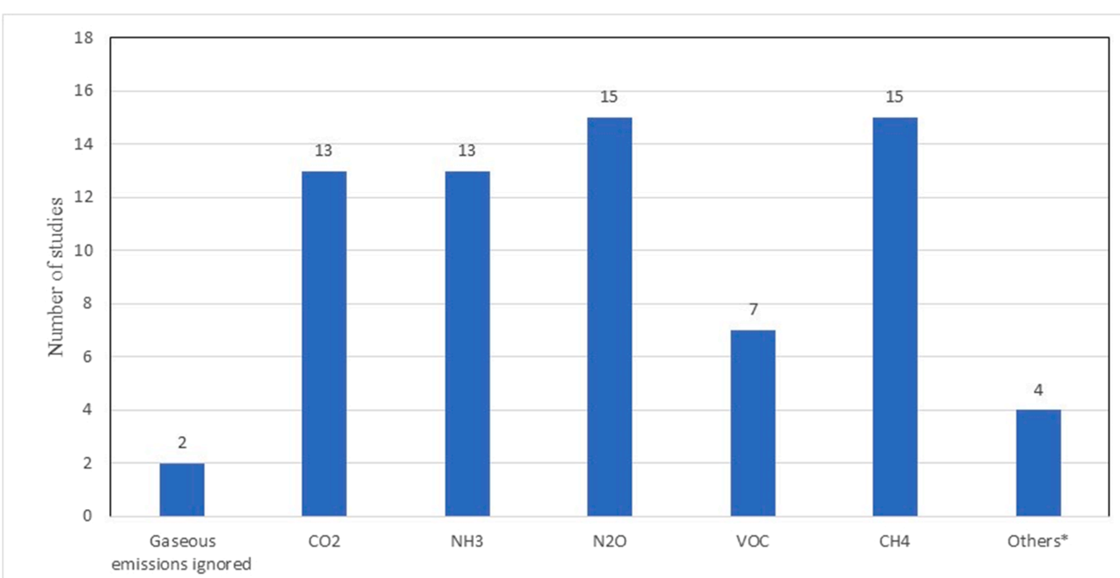

Fig. 6. Gaseous emissions considered in LCA of biowaste and green waste composting systems (n = 25 studies). Note: * include H_2S, SO_2, HCl, CO, particulate material. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

crop area; the crop is harvested and transported to the storage buildings. These activities consume materials and energy for infrastructure and machinery production and use, and cause emissions from the field and nursery plot (Quirós et al., 2014b). It has been reported that inventories from other studies often exclude the post-application of compost due to the complexity of quantifying the emissions and the lack of data (Bong et al., 2017).

On the other hand, for industrial composting, the emissions depend on the operation mode of the composting system. Windrow systems seldom include emissions for the manufacturing of composting equipment. The system consumes energy from the continuous air or oxygen supply for the aerated static pile. For the in-vessel system, the GHG emissions for the manufacturing of the composter is often considered. The GHG emissions due to different operational modes and energy consumption are accounted for; this includes turning, aeration, shredding, screening, and compost spreading. Saer et al. (2013) conclude that the major contributor to GHG emissions in composting systems is the direct emission during organic matter decomposition. Other processes, such as waste collection and transport, infrastructure, ancillary materials production, and energy consumption (e.g., for ventilation equipment), had a low contribution to GHG emissions. Despite this, these emissions cannot be neglected.

The lower GHG emissions due to the avoidance of waste collection and the lower investment costs are advantages of home composting. For home composting, inventories of other studies often included the GHG emissions for manufacturing the related tools such as the bin and garden chipper, direct GHG emissions, energy consumption during composting, and the compost post-application. Home composting shows a significant variation in GHGs inventories among various studies that have limited its application from a scientific perspective (Colon et al., 2010). Quirós et al. (2014b) compared two home composting systems with high and low GHG emissions. They identified a significant difference of 4-, 5-, and 52-times higher emissions of CH_4, N_2O, and NH_3 in the high-emission system with no mixing compared to the low-emission one with frequent turning. Colon et al. (2012) compared home composting and industrial composting (composting in-vessel and turned windrow composting). They measured the gaseous emissions finding the following values (per ton of processed material), respectively: NH_3: 0.84 kg, 0.11 kg, and 8.63 kg; VOCs: 0.56 kg, 0.36 kg, and 5.70 kg; N_2O: 0.676 kg, 0.075 kg, and 0.251 kg; CH_4: 0.16 kg, 0.34 kg, and 4.37 kg. Home composting did not always present have the lowest gaseous emissions for the three systems compared.

3.2.3.6. Energy and water consumption and other emissions. Energy and water consumption and leachate generation are rarely addressed in composting studies. Some studies assume that the production of leachate and emissions related to water do either not occur or are insignificant (i.e., due to the low water content and the high temperature in the piles) (Bernstad and Cour Jansen, 2012). However, industrial-scale composting often requires the addition of water to maintain optimal biological conditions for converting organics into marketable compost. Also, in-sink food waste disposers need running tap water to facilitate grinding and flushing (Morris et al., 2013). Therefore, water-related emissions should not be neglected. In this review, only 3 of the analyzed studies considered leachate in their assessment (Andersen et al., 2012; Oldfield et al., 2018; Bong et al., 2017). Some studies motivated the fact that water-related impacts were not considered, due to low quantities of leachate generated (Keng et al., 2020; Martínez-Blanco et al., 2010; Colon et al., 2012). A third option was suggested by Yadav and Sammader (2018b) and Colon et al. (2012), who assumed that leachate could be recirculated into the windrow to avoid the necessity of treating the generated leachate.

Industrial composting typically does not provide an energy benefit to counterbalance the energy needed for collection, transport, and composting facility operations (other than the offset energy for amendment production). However, home composting uses little energy input among the available composting technology options. Home composting relies almost entirely on human labor, which is not considered in LCAs that compare energy consumption (Morris et al., 2013). Some values reported are:

a) Bernstad and Cour Jansen (2012): showed that the energy input in composting varies largely in different studies; the values found by the authors are 15.1–55.0 kW h electricity/ton and/or 0.01–15.3 L diesel/ton (i.e., to windrow, closed reactor, and home composting technologies).

b) Buratti et al. (2015) reported a total electricity demand of 26.03 kWh/ton of OW and a diesel demand of 1.06 kg/ton of OW (i.e., enclosed agitated bed system technology).

c) Cadena et al. (2009b) quantified emissions associated with energy use and production (60.5 kg CO_2 Mg OFMSW[1] and 0.66 kg VOC Mg $OFMSW^{-1}$) in closed composting reactors with controlled aeration, gas collection, and treatment through a wet scrubber and a biofilter. The maturation phase took place in forced-aerated windrows open to the atmosphere).

d) Banias et al. (2020) reported that the energy used for the operation of the composting facility was 19.67 kWh of electricity, and 0.36 L of diesel, per ton of treated waste (i.e., technology not specified).

e) Di Maria and Sisani (2017) reported the electrical energy consumption was assumed to be 40 kWh/Mg OFMSW at the plant inlet. In contrast, diesel fuel consumption was assumed to be 0.134 kg/Mg OFMSW (i.e., technology not specified).

f) Colon et al. (2012) reported the electrical energy consumption was 740 MJ/Mg OFMSW, 235.8 MJ/Mg OFMSW, 33 MJ/Mg OFMSW, and 33 MJ/Mg OFMSW for composting in-vessel, composting in confined windrows, turned windrow composting, and home composting, respectively (7.435, 91, 3.000 and 0.43 t/year of BW processed, respectively). The authors indicate that the energy consumption was highly dependent on the machinery and the composting technology employed. The greatest energy consumption was for in-vessel and windrows aeration. These two technologies included additional processes to treat gas emissions (scrubbers and biofilters) that accounted for 45% of the total energy consumption in these facilities.

3.2.4. Life cycle impact assessment (LCIA)

3.2.4.1. Impact categories. Fig. 7 shows the impact categories considered in LCA studies on BW or GW composting. The studies used different impact categories depending on the goal and scope definition, system boundaries, and impact assessment method. According to Iqbal et al. (2020), the coverage of a greater number of impact categories represents a more detailed analysis. Despite a high diversity of impact categories considered in the studies, some categories are generally evaluated popular in the LCAs of composting systems. Among the most frequently selected categories are climate change (24/24 studies), terrestrial acidification (23/24 studies), freshwater eutrophication (21/24 studies), ozone depletion (19/24 studies), marine eutrophication (19/24 studies), human toxicity (18/24 studies), photochemical oxidant formation (17/24 studies), fossil fuel depletion (14/24 studies), and terrestrial ecotoxicity (11/24 studies). As observed, the number of categories analyzed in these studies is different. In some cases, this is due to the specific objectives of the work (some of them are only focused on climate change or carbon footprint), but in the case of a complete LCA there are two main groups: articles published before the implementation of ReCiPe2016 from the National Institute for Public Health and the Environment (The Netherlands) and other more detailed (Serafini et al., 2023) and recent articles using these modern methodologies (ReCiPe 2016, CML latest versions, IPCC values, Eco-indicator, etc.). Modern and previous methodologies follow upon the same strategy as ReCiPe2008, where different sources of uncertainty and different choices are grouped into a limited number of perspectives or scenarios, although the number

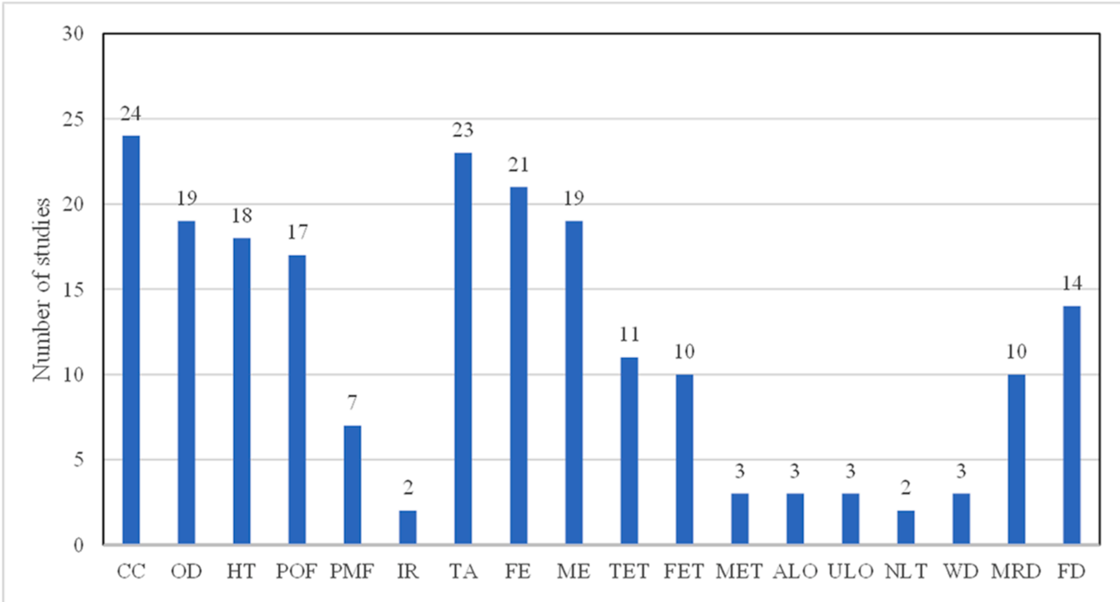

Fig. 7. Impact categories used in LCA of biowaste or green waste composting (n = 24 studies)*. **Note:** CC: Climate change; OD: Ozone depletion; HT: Human toxicity; POF: Photochemical oxidant formation; PMF: particulate matter formation; IR: ionizing radiation; TA: terrestrial acidification; FE: Freshwater eutrophication; ME: Marine eutrophication; TET: Terrestrial ecotoxicity; FET: Freshwater ecotoxicity; MET: Marine ecotoxicity; ALO: agricultural land occupation; ULO: urban land occupation; NLT: Natural land transformation; WD: water depletion; MRD: metal depletion; FD: fossil fuel depletion. * One article only evaluated endpoint indicators. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

of categories is different, being more detailed in the case of current methodologies, which are widely applied in most recent papers (Catalán et al, 2019). Regarding end-point indicators (three damage impacts: human health, natural resources, and biodiversity), it is the opinion of this review that they are a critical future trend, but no concluding results can be obtained nowadays in the composting field.

For example, Zhang et al. (2021) compared 45 LCA studies on organic solid waste management. They found GWP, also called 'climate change impact', was commonly considered in all the analyzed studies. The second most common impact was acidification potential and human toxicity potential. Similarly, Morris et al. (2013) reviewed 82 studies on end-of-life management methods for source-separated organic waste; they found that climate change was the environmental impact most frequently investigated. On the other hand, only a few studies (Oldfield et al., 2018; Abduli et al., 2011) justify their selection of the impact categories.

3.3. Perspectives and challenges for LCA applications to biowaste and green waste composting processes

The credibility and validity of the results of an LCA of biowaste and green waste composting face several challenges, for instance, data inconsistency, frequent neglect of the goal definition, variation in system boundaries, multi-input allocation, difficulties in capturing influential local specificities (e.g., the inclusion of representative waste compositions into the inventory), and unclear user assumptions. These challenges create complexity and limit the ability to compare several studies as the basis of system definition may differ. Similarly, the results of an LCA study are closely related to the specific conditions of each context, such as lifestyles, climatic conditions, composition, and waste characteristics. Based on this review, the challenges to address for improving the application of LCA in biowaste and/or green waste composting are described as follows.

3.3.1. Determination of system boundaries

The methodological framework for LCA application to biowaste or green waste composting needs standardization. This review found that current limitations to this standardization include data collection and its quality, system definition, system and time boundaries, and process modeling. Three approaches for the determination of system boundaries when conducting LCAs regarding the composting of biowaste or green waste were identified:

1) Pre-composting, composting, and post-composting can be included when evaluating different solid waste management options (e.g., incineration, anaerobic digestion, landfill, composting). Comparisons between industrial composting and home composting are also included in this approach.

2) Composting and post-composting can be included when evaluating different organic solid waste management options (e.g., anaerobic digestion and composting).

3) Composting can be included when evaluating different composting technologies or when the effect of some process improvement strategies is being investigated (e.g., adding inoculant, bulking or amendment materials, and aeration systems). Depending on the study scope, post-composting could also be included to assess the environmental impacts of end-product use.

Defining system boundaries is essential since this influences aspects such as the functional unit, including or excluding waste and end-product transport, allocation of environmental burdens, and avoiding environmental loads. Particularly the specification of system boundaries is crucial to reduce inconsistencies between different studies (Komilis and Sanchez, 2017).

On the other hand, a relative consensus exists around ignoring the infrastructure construction and dismantling stages. In this review, the studies that included these two stages found their contribution to environmental impacts to be lower compared to those generated in the operation stage.

3.3.2. Data collection

This review shows that a large amount of data is necessary to determine the environmental impacts of a composting system. Original data still need to be carefully collected (i.e., primary sources such as measured or estimated data after field investigation, official data from statistical reports, interviews, surveys, and site visits) and should not be solely based on the data contained in the “black box” of commercial software (Komilis and Sánchez, 2017). However, this does not exclude consolidating a database with information from rigorous scientific studies. This database would make it possible to compare the data obtained in different studies and improve the inventory phase of the LCA.

The database could be organized according to country-specificities, technology-specificities, waste characteristics, end-product characteristics, and other input and output data (e.g., emissions), considering that the assumptions or data from other studies may be specific to the locations where the studies were conducted.

3.3.3. Gaseous emissions

This review highlights the importance of considering all the gaseous compounds emitted from the process (CO_2, NH_3, N_2O, COV, and CH_4) and analyzing their contribution. Estimating actual gaseous emissions data is essential, considering that these depend on the process conditions, the technology used, the type of substrate (or a mixture of substrates), and the treatment processes applied. Since this is not always possible, especially in developing countries, information from databases can be used. Still, validating the data source (i.e., conditions, origin, process characteristics) is necessary. Additionally, the incorporation of indicators of odors generation in biowaste composting needs further study to allow its inclusion in LCAs. Recent studies on odor estimation (Cadena et al., 2018) or mitigation (Nguyen et al., 2023; Gao et al., 2022) offer important information for environmental impact assessment.

3.3.4. Environmental impacts from compost utilization

There is a lack of information related to the environmental impacts of compost utilization as soil amendment to determine the replacement capability of compost. Further studies are required that relate the quality of the end-product with the resulting environmental impacts for different uses (e.g., agriculture, horticulture, soil amendment, landscaping). This relation depends on the quality and physicochemical characteristics of the compost and on the environmental conditions of the system where the product is applied. Recent studies show that using compost generates more benefits than simply adding nutrients (for instance, organic compost has significantly higher antioxidant and anticarcinogenic properties than chemical fertilizers) (Komilis and Sánchez, 2017). Future studies should determine: i) the avoidance of conventional soil improvers or amendments, ii) long-term carbon sequestration, iii) improvement of soil properties, iv) contribution to climate change mitigation and v) other uses of compost such as biofilter media, soil bioremediation or landfill cover.

3.3.5. Impact categories

Only a few studies (Oldfield et al., 2018; Abduli et al., 2011) justify their selection of impact categories. These studies allow for standardizing the categories used in a composting system according to the system goal and scope and depend on the impact assessment methodology selected. These arbitrary choices seriously influence the results and can restrict the completeness and reliability of an LCA. Although coverage of a greater number of impact categories allows a more detailed analysis, in this review, few studies analyzed the following impact categories: ionizing radiation, agricultural land occupation, urban land occupation, natural land transformation, water depletion, and marine ecotoxicity. All LCA in BW and/or GW composting should justify the choice of the impact categories evaluated in agreement with the studýs goal and scope.

3.3.6. Environmental impacts of strategies for improving composting technologies

Generally, the strategies for improving the composting of BW and GW are addressed from a technological perspective (e.g., reduction of composting time and improvement of product quality). These strategies may include amendment or bulking materials, inoculum addition, and forced aeration systems. However, there is a lack of studies assessing the effect of these strategies on process emissions (e.g., gaseous emissions such as CH_4 and N_2O) and, therefore, on the impacts in terms of environmental impacts. Research is needed to compare facilities of different sizes, operating systems, and changes in substrate quality to draw more convincing conclusions on potential mitigation strategies.

Other technological aspects of BW or GW composting that can be studied through LCA were also identified in this review, for example, (i) the effect of feedstock porosity in the composting process using different technological systems (i.e., porosity affects process conditions and gaseous emissions, and therefore, environmental impacts); (ii) the effect of biofilters in reducing greenhouse gas emissions and, therefore, environmental impacts; (iii) the environmental impacts of decentralized management approaches, such as community composting (hotels, hospitals, universities, and schools); (iv) the environmental impacts of the residues generated during the operation of the BW composting plants (e.g. greater area requirement and extra energy use). Works have advanced on the environmental impacts of residues generated in the organic fraction of municipal solid waste anaerobic digestion plants (Colazo et al., 2015). Still, research is scarce with regard to residues from BW or GW composting systems. Finally, although this study did not include vermicomposting, future studies could employ LCA to consider aspects associated with the emissions of this technology. Some works have advanced in estimating gaseous and other emissions from vermicomposting (Lleó et al., 2013).

3.3.7. Use of software

There is an increasing need for predictive models to support environmental policy and decision-making. The validity of these models depends on the availability of reliable, rigorously generated data relevant to the study context. This work found several LCA software tools that evaluate the environmental impacts of BW and/or GW composting. Even though all of these tools contain databases and information, very few users are aware of certain software features, specific terms, and underlying computational processes. In addition, the data contained in the databases are restricted to limited scenarios (Komilis and Sanchez, 2017). Therefore, it is necessary to train users to improve the application of these tools. The training may include aspects that increase the understanding of: (i) composting processes, (ii) emissions generated in the process (e.g., mass and energy balances), (iii) basics of LCA, (iv) input data to the software, (v) use of specific software, (vi) interpretation of results.

4. Conclusions

In this study, a review of LCA articles assessing the environmental impacts of biowaste and/or green waste composting was systematically performed. This work demonstrates the complexity of conducting credible and valid LCA studies on biowaste composting. Future research should be carried out with the aims of: i) improving input data, prioritizing the collection of primary information, and the creation of regional databases in the context of data scarcity mainly in developing countries (e.g., waste characteristics, end-product characteristics, emissions, and other input and output data); ii) improving the definition of the system boundaries in agreement with the objectives of the LCA study (pre-composting, composting, post-composting); iii) achieving a standardization of the environmental impact categories (i.e., midpoint and endpoint) used in accordance with the goal and scope of the studies; iv) conducting studies on the environmental loads avoided by applying compost for different purposes; v) assessment of the environmental impacts of specific strategies for improving composting technologies (compare facilities of different operating systems and technologies); and vi) training practitioners in the application of LCA models, considering criteria for data selection, understanding process material and energy balances, and interpreting results.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

It is a review, data is from other studies

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2023.09.004.

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