

## Review

# Life Cycle Sustainability Assessment of Waste to Energy Systems in the Developing World: A Review

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**Abstract:** The global move towards a circular economy, as well as that of achieving the United Nations Sustainable Development Goals (SDGs), has necessitated the search for several sustainable solutions in various sectors. Given this, the provision of sustainable waste management and electricity systems constitute a significant part of the SDGs, and the waste-to-energy (WtE) concept has recently become a key topic given that it can potentially help reduce the dependence on fossil fuels for energy generation, as well as minimizing the need to dispose of waste in landfill. However, to date, the sustainability assessments of WtE generation technologies have been limited in scope concerning the three-dimensional sustainability framework (economic, environmental, and social). Life Cycle Sustainability Assessment (LCSA) has been proposed as a potential approach that could comprehensively address these three pillars of sustainability simultaneously based on life cycle thinking. LCSA, as a holistic method, could also potentially deal with the complexity associated with decision-making by allowing for the consideration of a full range of possible sustainability consequences. LCSA is an analytical tool that integrates the Life Cycle Assessment (LCA), Life Cycle Costing (LCC), and Social Life Cycle Assessment (sLCA) methodologies, which already exist and continue to be developed. Individually, these life-cycle approaches tend to be used to point out particular ‘hotspots’ in product or service systems, and hence focus on direct impacts in a given sustainability domain, neglecting the indirect ones. LCSA aims for a more holistic sustainability perspective and seeks to address the associated challenge of integrating these three pillars of sustainability into an overall and more comprehensive sustainability assessment. This need for harmonization within the LCSA methodology is a major challenge in its operationalization. In recent years there has been steady progress towards developing and applying LCSA, including for WtE. The aim of this paper is to review the most recent trends and perspectives in developing countries, especially regarding how LCSA could help inform decision-making. The paper also analyses the LCSA literature to set out the theoretical and practical challenges behind integrating the three methods (LCA, LCC, and sLCA). The review was conducted via a search of keywords such as LCSA, waste, and energy in the Web of Science databases, resulting in the selection of 187 publications written in English. Of those, 13 articles operationalized LCSA in specific waste and WtE related case studies. The review provides a review of the application of LCSA for researchers, technological experts, and policymakers through published findings and identifies perspectives on new research. These include uncertainty, subjectivity in weighting, double-counting, the low maturity of sLCA, and the integration of the interconnection between the three dimensions (environmental, economic, and social dimensions) of LCSA results in decision-making. In addition, gaps (such as the integration of the interconnection between the three dimensions) that need to be addressed via further research are highlighted to allow for a better understanding of methodological trade-offs that come from using the LCSA analytical approach to assess the sustainability of WtE generation technologies, especially in developing countries. It is hoped that this study will be a positive contribution to environmental and energy policy decisions in developing countries faced with the dual problems of waste management and electricity supply along with their sustainable development goals.



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**Keywords:** waste-to-energy (WtE); life cycle assessment (LCA); life cycle costing (LCC); social life cycle assessment (sLCA); life cycle sustainability assessment (LCSA)

## 1. Introduction

Municipal solid waste (MSW) management is a major challenge, particularly in the developing world, where it presents significant risks to the environment and society [1–6]. In 2018, global MSW generation was estimated to be 2.1 billion tons, and this is expected to increase to 3.4 billion tons by 2050 [7]. This significant rise in MSW generation is a consequence of economic growth, population increase, and industrial and urban development, as well as rural-urban migration [8]. This has led to the management of MSW becoming a serious challenge for many nations [9]. However, the situation has become especially pronounced in developing countries relative to developed ones [10] (Reviewer 3-comment 6). These challenges are due to problems associated with waste not being sorted at the source, limited access to a system that collects and separates waste, overuse of landfills, and unregulated leachate release and gas emissions from landfills [11,12]. Also, the limited space available, higher landfill costs, and increased societal pressure associated with landfills have made landfills as an option less attractive.

On the other hand, there is also the challenge of inadequate energy supply related to accelerated urbanization, particularly in developing countries [13,14]. This is coupled with an energy landscape dominated by fossil fuels, leading to emissions of greenhouse gas (GHG). Hence, these issues have generated much interest in terms of researching how solid waste can serve as a feedstock for renewable energy sources [9]. The solid waste management sector has gone from being one that focuses mainly on treatment and disposal to one that places a strong emphasis on the optimal conversion of MSW into a wide range of valuable products [15–20]. Waste to Energy (WtE) has attracted attention globally, especially for its capacity to address various urgent issues concurrently, such as waste management, energy production, and environmental pollution [21]. In addition to this, the adoption of WtE will assist governments to achieve a number of the sustainable development goals (SDGs), such as clean energy (SDG 7) and sustainable cities and communities (SDG 11) [22].

Various technological options, such as incineration, gasification, pyrolysis, anaerobic digestion, and landfill gas to energy, etc., have been suggested as means of energy recovery from waste. However, each has its own environmental, social, and economic impacts. Indeed, these various options can make it challenging for decision-makers to select a suitable WtE technology, due to the absence of complete and consistent methods that facilitate comparison. Therefore, an approach that allows sustainability assessment of the different proposed WtE technologies could provide decision makers with a logical framework for comparison.

One framework showing much promise for all comparisons of the sustainability of technologies is the Life Cycle Sustainability Assessment (LCSA). LCSA has three components: environmental life cycle assessment (LCA), life cycle costing (LCC), and social LCA (sLCA). However, there is still much diversity in the literature regarding how these components, most notably sLCA and LCC, are assessed and, critically, how they need to be integrated into an overall picture of sustainability. Therefore, while LCSA has much potential as both a research framework and a guide for decision-makers, some of its key operational elements, such as the integration of its three components, have been subject to a diverse range of approaches. This diversity in approach, and the reasons for it, has been explored in reviews of LCSA, but to date, there has been no published review of LCSA within the context of WtE or indeed waste treatment in general, especially in the developing world. WtE presents an intriguing and challenging context for such an analysis given that it is intended to address two major issues in the developing world: the supply of energy (typically in the form of electricity) and the management of waste. In addition, the problem associated with the integration of the results of the three dimensions

(environmental, economic, and social) into the LCSA framework is one of the main gaps that needs to be addressed. Hence, this paper aims to review the current trends and viewpoints connected with using LCSA to assess the sustainability of WtE, with a key focus on developing countries, and to identify the trends and approaches taken to date with the use of LCSA in this context. It will identify critical gaps for future research to address, as well as recommendations for decision-makers. Addressing the methodological options for prospective LCSA studies makes this review highly relevant, as this could potentially provide insights to aid decision- and policy-making for the adoption of WtE, and indeed other technological innovations, in developing countries, based on the assessment of their sustainability performances and potential contributions. Therefore, this paper's objective is to perform a methodical review of articles on LCSA published in research journals, and to identify the types of studies that have been developed recently and the way practitioners are using LCSA. The paper begins with a brief outline of the methodology adopted for the literature review, followed by a summary of the functioning of WtE technologies and their application throughout the developing world and approaches taken to date to assess their sustainability, with emphasis on the use of LCSA and the results that have been achieved. For convenience, this section has been divided into the assessment of environmental, social, and economic impacts, along with an exploration of the range of approaches taken to date to integrate them. The paper concludes with a discussion of the findings, a set of suggestions for future research, and recommendations for decision-makers.

## 2. Materials and Methods

The rationale behind this paper is to use a methodical literature review of various LCSA studies to propose novel recommendations that will provide innovation for more comprehensive and integrative approaches to LCSA. This, along with the identification and addressing of significant gaps in the literature, will contribute to improving the methodology and operationalization of LCSA. As previously stated, addressing the methodological choices of prospective LCSA studies makes this review important, as this could potentially provide insights required to support decision-making regarding the adoption of technological innovations such as WtE in developing countries based on the evaluation of their sustainability performances and the potential contribution of these technologies to these countries. The term 'developing countries' in the context of this study is used to describe countries that have not attained a significant level of industrialization relative to their populations, and have, in many cases, a medium to low standard of living [23].

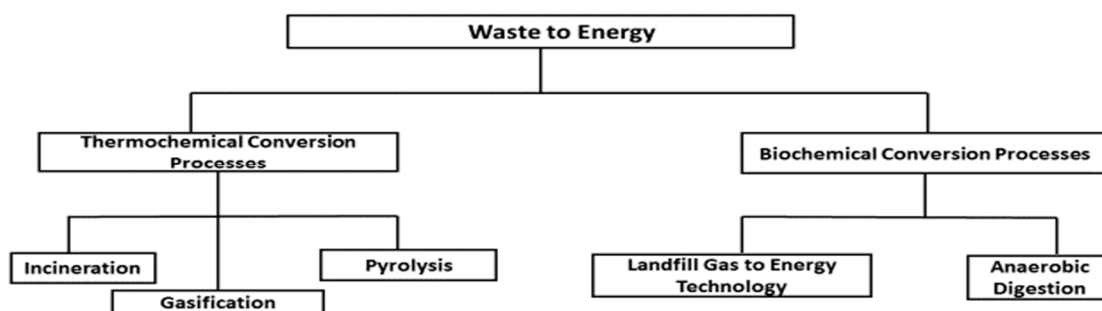
In order to achieve this aim, a methodical review of the literature was undertaken (in early 2024) using the search engine Web of Science, as it provides access to a wide range of peer-reviewed literature. The initial search was conducted using "LCSA" as the search term. This was followed by a search using combinations of search terms, such as "LCSA", "waste", and "energy", for publications related to waste to energy in the second step. A total of 187 publications were found in the Web of Science database using LCSA as a search term. While these 187 publications spanned a range of applications, including, for example, transport, they nonetheless provided some points of interest, such as the approaches taken to integration. Out of these 187 publications only 13 publications were found when using "LCSA", "waste", and "energy" as keywords, and these provided the core of the analysis provided here.

The systematic literature review involved using the search engine Web of Science, given that it is possible to obtain a large amount of peer-reviewed, credible, and high-quality literature from this search engine. Also, it was taken into account that the duplicity of articles found in this database showed a significant level of acceptance by the scientific community [24]. In addition, only publications written in the English language were selected during the search for relevant LCSA publications. Given this, the analysis of the literature based on the Web of Science databases provided the publication results using keywords such as LCSA as a first step in the search. This was followed up by collectively using keywords, such as "LCSA", "waste", and "energy", for publications related to waste

to energy in the second step. From here, there were a total of 187 publications in the Web of Science database relating to using LCSA as a keyword in the search, out of which only 13 publications were found when using “LCSA”, “waste”, and “energy” as keywords. The details regarding these 13 publications are further explained in Section 4.1 of this paper.

### 3. Overview of Waste to Energy Technologies

WtE can be defined as the transformation of waste into energy (electricity), heat, or fuel through thermo-chemical conversion processes as well as biochemical conversion processes [25]. It refers to how energy in the form of usable heat and electricity is extracted by running gas or steam via a turbine or fuel recovered from waste [8,26]. In the waste management hierarchy, WtE or energy recovery from waste is the strategy that comes before final disposal. WtE is more widely used in developed nations than in developing countries, where waste collection, transportation, and proper disposal are still major problems [20,21]. As a result, there are more than 1700 WtE plants worldwide which utilize various technologies to generate electricity from MSW [27,28], of which 62% are in Asia, followed in share by Europe and North America with 33% and 4.5%, respectively. Figure 1 depicts how energy production from MSW can be performed via thermochemical or biological processes [28,29].



**Figure 1.** A schematic diagram showing the different WtE processes.

Thermochemical conversion processes (incineration, gasification, and pyrolysis) involve thermally treating MSW to produce heat energy, fuel oil, or gas. Incineration is a broadly used WtE technology that recovers high-temperature heat from the combustion of MSW, which can then be used to generate electricity [25]. Gasification is another thermochemical conversion technology that involves the partial oxidation process, and usually occurs at a temperature range between 750 and 1100 °C [28,30].

The process involves the breakdown of MSW into ‘synthetic gas’ (syngas) which consists primarily of hydrogen (H<sub>2</sub>), carbon monoxide (CO), carbon dioxide (CO<sub>2</sub>), and lesser quantities of methane (CH<sub>4</sub>). After cleaning, the syngas can be used to generate electricity in fuel cells or as fuel in engines and turbines [31,32]. Pyrolysis is a thermochemical conversion process that involves degrading organic waste at high temperatures (300 °C to 800 °C) without oxygen [31,33]. The main end-products of pyrolysis of organic waste include syngas, pyrolysis oil, and biochar [31,32].

Biological conversion processes consist of anaerobic digestion (AD) and landfill gas to energy technology (LFGTE) involve the breakdown of the organic content of MSW by microorganisms to produce gaseous or liquid biofuels, which can generate energy [32,34]. AD uses microorganisms to convert MSW into biogas, whose main constituents are CH<sub>4</sub> and CO<sub>2</sub> [34,35]. The process involves the degradation of biodegradable organic matter by microorganisms without oxygen and leads to biogas production [35,36]. LFGTE involves producing landfill gas (LFG) from the decay of organic matter in landfills. In this technology, the MSW is deposited into a landfill, compacted, and covered, with vents provided for the gas to escape and be collected [36,37]. Of the options in Figure 1, thermochemical conversion processes are more rapid and stable, do not produce foul smells, and are pathogen-free. The biological processes take longer as they rely on the activity of microorganisms, and

they also require a suitable and stable environment for the microbes to function [37,38]. In terms of economic feasibility, biological processes tend to be more economical than thermochemical processes as they require less capital expenditure [32,34]. WtE processes also differ in environmental impacts, with thermochemical processes generally having more significant impacts relative to biochemical conversion [37,38].

For instance, the incineration of MSW can result in the emission of various pollutants such as CO, CO<sub>2</sub>, sulfur oxides (SO<sub>x</sub>), NO<sub>x</sub>, particulate matter, dioxins, furans, benzene, ethylbenzene, 1-hexene, toluene SO<sub>2</sub>, hydrofluoric acid, hydrochloric acid, and other hazardous compounds into the atmosphere. These pollutants and GHGs cause global warming and result in other forms of environmental degradation, such as acidification, which may threaten vegetation and lead to the corrosion of buildings. Pyrolysis, on the other hand, has received much attention because it is highly energy efficient and it has recently been considered an alternative to the incineration process [27]. This is attributed to the fact that the process produces fewer quantities of harmful gases [39]. There are still numerous problems encountered from the conversion of MSW to energy during pyrolysis, such as air pollution due to HCl, H<sub>2</sub>S, NH<sub>3</sub>, SO<sub>x</sub>, and NO<sub>x</sub> emissions, along with odor-related impacts [40]. However, this can be minimized using emission control strategies and regulations alongside the development of improvement measures for the quality of gas, liquid, and char produced to make pyrolysis a more environmentally favorable process, as obtained in many developed countries where emission standards for gases such as CO, NO<sub>x</sub>, and SO<sub>2</sub> are set at around 10.6 ppm, 16.8 ppm, and 2.3 ppm, respectively [41]. AD, however, has less effect on air quality and contributes to reducing CO<sub>2</sub> emissions by producing energy to replace fossil fuels [42]. In terms of economic feasibility, a study performed by [38] on the economic feasibility of incineration, gasification, AD, and LFGTE showed that LFGTE and AD were more economically feasible than gasification and incineration because the latter both required significant capital investment.

In terms of the social implications of adopting WtE technologies in developing countries, public health and safety management is one social issue that needs to be considered, as there have been numerous studies which have confirmed significant health effects. For instance, Giusti [43] revealed that the flue gases from incinerators containing acidic gases such as SO<sub>2</sub>, NO<sub>x</sub>, N<sub>2</sub>O, HCl, and HF, certain metals like arsenic, beryllium, chromium, cadmium, lead, and mercury, CO, CO<sub>2</sub>, dioxins, furans, polyaromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and volatile organic carbons (VOCs), increases the risk of people being prone to health problems such as lymphomas and sarcoma. There is also the case of government initiatives and political will.

For example, the 11th Malaysia Plan (2016–2020) encourages the idea of responsible consumption and production, which is the UN SDG 12, and focuses on treating solid waste as a new resource which can be recovered materially via biochemical or thermochemical routes for electricity generation. This plan aims to transform Malaysian society into a more energy and resource efficient entity by managing waste and raising the percentage of renewables in the energy sector and in the national electrical generation mix [44].

In addition, the feasibility of implementing these technologies in developing countries is further enhanced by studies such as those of [45], which assessed WtE in twelve selected cities in Nigeria using LFGTE, incineration, and AD technologies with the aim of assessing the electricity potential for each location, as well as evaluating the economic viability of possible WtE projects. In this study, the waste generation profile of each location was estimated using population data and the per capita waste generation. The economic viability of the projects was assessed using levelized cost of energy (LCOE) and net present value (NPV) methods. Some of the key findings showed that incineration and LFGTE technologies proved to be the most favorable technological options in terms of electricity generation potential in the northern part of Nigeria. This was based on the kind of waste produced at these locations. AD was found to be the best technological option in the Southern region. However, from an economic viability perspective, AD was found to be the best option in all the locations, followed by LFGTE. Incineration had the highest LCOE,

and hence was considered to be less economical for electricity generation in all locations. This paper proved beneficial to the stakeholders and policy-makers, as it provided valuable scientific information to guide optimum investment in WtE in Nigeria.

Pheakdey et al. [46] also conducted a study assessing the electricity generating potential, economic feasibility, and environmental impacts of LFGTE, incineration, and AD technologies in the Phnom Penh municipality in Cambodia. This study also evaluated the feasibility of each technology using the LCOE, payback period (PBP), and NPV during the economic analysis. The study went a step further to assess the environmental performance. The findings revealed that incineration produced the highest electricity generation output, ranging from 793 to 1626 GWh/annum, while those of LFG and AD technologies were 115 to 272 GWh/annum and 163 to 333 GWh/annum, respectively. The economic assessment indicated that the LCOE for LFG was about US\$0.070/kWh, while it was US\$0.053/kWh for incineration, and US\$0.093/kWh for AD. Thus, incineration and LFG recovery were found to be economically feasible, with positive NPV and a PBP of 8.36 years for incineration and 7.13 years for LFG. However, AD technology had a negative NPV and required over 20 years to generate a return on investment. On the other hand, AD had the best environmental performance; it was found that it could save an estimated 133,784 tCO<sub>2</sub>-eq/year. This study presented useful information for decision-makers and prospective investors to utilize in the optimization of WtE investment in Cambodia.

In developing nations, implementing WtE has faced several environmental, economic, and social challenges. This comes from considering several additional factors, such as the identification of waste characteristics and volume, suitable tariff system regulation, sufficient human resources, the capacity to cover large investments and operational costs, and appropriate regulatory support from the government [47]. Nevertheless, implementing WtE technologies in developing countries can serve as a potential solution for waste management and energy supply [48], as well as serving as a means of job creation [49]. However, investment in solid waste management practices such as WtE in developing countries can only obtain optimal results if the necessary background information on the energy-generating potential from the waste is available, alongside social and economic costs and benefits [50]. It should be noted that WtE is not the only solution available for dealing with MSW, and should thus be integrated with established 4R (Reduce, Reuse, Recycle, and Recover) programs [51]. In developed countries, the typical range of electricity generation from WtE is around 500 to 600 kWh per ton, while for developing countries, the range is from 300 to 400 kWh per ton [52]. This difference is largely due to variations in MSW quality and heating value [53]. For instance, the heating value of MSW in developing countries is relatively low due to the high percentage of food waste, high moisture content, and the paucity of MSW sorting systems [54]. Despite this, there has been significant improvement in the efficiency of WtE technology over the past decade, although it has been noted that the business model, management, and regulations/emission standards need to be modified to help facilitate adoption [12]. In many developing countries, such as India, Vietnam, and Malaysia, WtE has mainly been implemented at a small scale to date [36].

Yan et al. [55] approximated that food waste alone could meet up to 4.1% of Vietnam's electricity demand if converted into biogas using an AD process. Also, in emerging economies such as China, where the national government has brought in policies to encourage the adoption of WtE, the country witnessed the construction of 510 WtE plants with an annual WtE capacity of 193 million tons in 2020 [56]. Indeed, given this supportive background, China could potentially generate approximately 31,346 and 77,748 GWh of electricity from MSW by 2030 and 2060 respectively [57].

In Saudi Arabia, Ouda et al. [58] investigated the potential of WtE in the country by considering technologies such as incineration, refuse derived fuel (RDF), and biomethanation by AD from 2012 to 2035. It was found that biomethanation technology was the best WtE technology for Saudi Arabia due to the high volume of food waste available as feedstock, the technology's high efficiency (25–30%), and its low annual capital (US\$0.1–0.14/ton) and operational cost. However, the requirement for a large space to continually

operate such technology might increase the operational cost. The RDF proved to be a better option than incineration due to its lower annual capital (US\$7.5–11.3/ton) and lower operational cost (US\$0.3–0.55/ton), but had the constraints of requiring highly skilled labor. Incineration, on the other hand, had a higher efficiency (25%) and lower operational cost (US\$1.5–2.5/ton). However, the requirement to treat air and waterborne pollutants and ash within the incineration facility represented one of the key limitations for its development in Saudi Arabia. In 2012, the power generation potential for Saudi Arabia was approximately 671 MW and 319.4 MW from incineration and RDF with biomethanation scenarios, respectively, which was projected to be about 1447 MW and 699.76 MW for both cases, respectively, by 2035 [58]. Therefore, WtE technologies could contribute significantly to renewable energy production in Saudi Arabia, as well as reduce landfill costs and their associated environmental impacts.

In Brazil, despite the country possessing tools that could promote energy recovery from waste as a treatment method, such as the National Solid Waste Policy (PNRS) [59] and the preliminary version of the National Solid Waste Plan (PLANARES) updated in 2020 [60], no energy recovery plants have been established yet. They are still in the design phase, or have pending environmental licenses. There exist only two energy recovery unit (ERU) projects in Barueri and Mauá, both in São Paulo State, which contributes nearly 29% of the MSW generated in the country [59]. At the first ERU, 825 tons of MSW with 25 MW of installed capacity are expected to be processed per day. At the second, it is estimated that 3000 tons of waste are to be transformed into 80 MW of power per day [61]. Although there are currently no WtE plants in operation in the country [62], various research projects have been conducted in Brazil to assess the potential of energy generation from waste in the country. For instance, Dalmo et al. [63] carried out a study on the potential for energy recovery from MSW in the state of São Paulo, Brazil. In this study, incineration and gasification (thermochemical routes) and LFGTE and AD (biochemical pathways) were considered, along with two-hybrid combinations. The study revealed that the combination of incineration and AD exhibited the highest theoretical potential of electricity generation (8,051,623 MWh/year). Another study performed by [64] estimated the electricity generation potential of the MSW management consortiums in Minas Gerais from LFG and anaerobic reactors for organic fractions of this waste. The results indicated that electricity generation from landfill biogas had the potential for economic viability at the Cias Centro Oeste, Cides, Cisab Zona da Mata, Convale, and Coresab Boa Vista MSW consortiums. The estimated electricity that could potentially be generated for the state of Minas Gerais would be approximately 3,900,000 MWh/year, and the total installed power capacity would be about 4500 kW for a thermal power plant (TPP) using LFG for all consortiums. These values were almost 15,400 MWh/year and 3200 kW for biogas energy derived from AD.

Lino and Ismail [65] assessed two scenarios for energy recovery from waste and social inclusion in the city of Campinas in the State of São Paulo, Brazil using incineration and biodigestion. The analysis showed that the first scenario, where the MSW was landfilled for biogas production and reused energy, was more beneficial environmentally compared to the energy potential. The CO<sub>2</sub> emissions corresponded to about 50% the value of the second scenario, while the electric energy generated by the combustion of biogas (about 817.2 MWh) was sufficient for 3783 residences with a consumption of 216 kWh/month. The market cost of this amount of electric energy is about US\$163,440 monthly. The results showed that this solution was not suitable for Campinas for several reasons: first, because the amount of electricity generated was relatively modest; second, because good locations for constructing new landfills are not available; and thirdly, because of the need to monitor the landfill for many years following its closure due to risk of explosions. In the second scenario, which considered the incineration of MSW, it was discovered that the produced energy was more than 30 times that from biodigestion. In addition, the electricity generated (28.991 MWh) was sufficient for 134,217 homes. This amount of energy equates to about US\$5798.2/month. The emissions were higher for incineration, but could be minimized to

acceptable limits by using a suitable pollution control system. The solid ash could be used in road construction, in the cement industry, and as aggregate for construction materials. Based on the results, it was concluded that this would be a more favorable option for Campinas, since it is a permanent solution representing the new global trends in MSW management.

Given this, there is no doubt that the potential for WtE in Brazil, using technologies such as incineration, LFGTE, and AD, is significant. Despite several positive studies on WtE, including environmental, social, and economic feasibility, in a number of developing countries, the potential of WtE technologies has been constrained in these countries. Thus, the sustainability of WtE in developing countries can only be achieved if several key issues, such as regulation and technical standards, business models, technical localization and development, not in my back yard (NIMBY), and labor training are considered and addressed [66].

Clear and compelling analysis of the overall sustainability benefits, such as by LCSA, plus the overcoming of various other constraints will be needed to enable projects to be implemented for WtE systems in developing countries to add to the electricity generation mix and reduce environmental impact, minimize health risks, and provide jobs for the local populace.

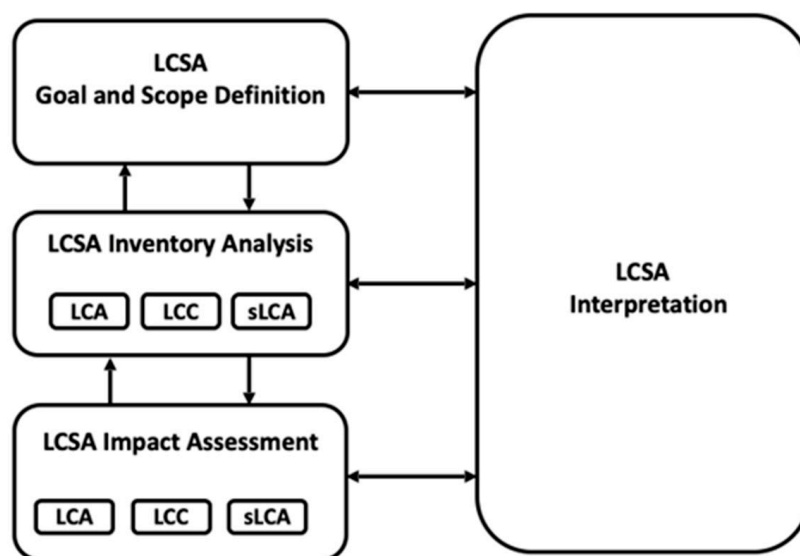
#### 4. Sustainability Assessment of Waste to Energy Technologies

Given the growing interest in adopting WtE in developing countries, the need to assess the economic, environmental, and social impacts of various WtE options as a basis for informed decision-making has been of interest to policy- and decision-makers. The sustainability assessment for any system needs to be framed within the context of an intersection between the three sustainability pillars: environmental, economic, and social [67]. Hence, a holistic approach to assessing the sustainability impacts of WtE systems needs to include indicators that extend across the three pillars of sustainability, with overlaps in certain areas [68]. Some recent studies have explored the sustainability of various WtE systems, but they have focused more on indicators from a single pillar (primarily environmental), while others have included indicators spanning two of the pillars [69]. However, including indicators across all three pillars is far less prevalent in the literature, as are critical discussions on how these can be integrated to provide a more comprehensive assessment of sustainability [70–72]. For example, a study of WtE options in Serbia used only eight indicators for the sustainability assessment of waste managed via landfilling, incineration, and AD. Three of these indicators were environmental and economic, and just two were social [71]. Another study conducted by Antonopoulos et al. [72] on the sustainability assessment of three WtE technologies used five, two, and two environmental, economic, and social indicators, respectively, and concluded that incineration had the best sustainability performance.

##### 4.1. Life Cycle Sustainability Assessment for Waste to Energy Technologies

LCSA as a concept was first introduced in 2007 by Finkbeiner [73] and developed further by Klöpffer [74]. LCSA is an evaluation methodology used for the measurement of the sustainability performance of a product or service over its entire life cycle [75–77], and the approach can be applied based on a number of International Organization for Standardization guides, including ISO 14040, 2006, ISO 14041, 1998, ISO 14042, 2000, ISO 14043, 2000, and ISO 14044, 2006 [76,77].

Indeed, the increasing interest in LCSA has led to UNEP-SETAC publishing a document as an introduction and a guideline [23]. Since LCSA is a relatively new approach, there is no agreed assessment methodology, and several researchers have introduced and used various frameworks to implement LCSA. In general, LCSA consists of four main steps consistent with the three tools: goal and scope, inventory, impact assessment, and interpretation [78], as depicted in Figure 2.



**Figure 2.** A diagram of the LCSA framework (based on [74]).

There are, broadly, two main formulations of LCSA. The first was suggested by Kloepffer [74], and comprises separate assessments based on LCA, LCC, and sLCA, which are then integrated. On this basis, Kloepffer [79] developed an equation for the sustainability assessment of an entire life cycle as follows:

$$\text{LCSA} = \text{LCA} + \text{LCC} + \text{sLCA} \quad (1)$$

Equation (1) above has a basis that rests on three separate life cycle-based assessments, where the LCA is the environmental assessment of a product/system's life cycle [80], LCC represents the assessments of costs and revenues of a product/system's life cycle [79], and sLCA assesses the positive and negative social impacts [81]. These three life cycle-based analyses are then summed to generate the LCSA. At its heart, the LCSA equation of Kloepffer [74] describes the concept of adopting the three assessments in a complementary way to the same functional unit (FU) and system boundary, a point that will be returned to later. However, while the LCSA equation above is a simple one and implies that the components can be summed, this should not mean that poor performance in any one component can be compensated by a better performance in another [82]; in effect, all three must be weighted equally in terms of importance. Also, the aggregation of the results is not because of the risk of missing transparency in the early stages of LCSA [83].

A second approach to LCSA has been proposed by Guinée et al. [84], whereby LCSA acts more as an interdisciplinary integration framework instead of a model, sharing the same concept as first proposed, but including a larger and more comprehensive scope of application. In this second approach, there is an expansion and broadening of the current LCA scope from primarily environmental impacts to covering all three dimensions of sustainability. This broadening of scope goes from mainly product level to sector level or even economy level [83]. Indeed, some methods have been developed to extend LCA within its basic framework to include economic and social elements [85]. Thus, the combination of LCA with other methods, such as material flow analysis (MFA) and cost-benefit analysis (CBA), is possible [86]. One major difference in this approach is that normative aspects such as discounting, weighting, and the concept of weak versus strong sustainability could be easily integrated as part of a broadening of scope [86]. Furthermore, this second approach differs from the first approach by having a single product system inventory (Life Cycle Inventory) [80] and a single integrated Life Cycle Impact Assessment (LCIA) [87]. For this reason, Kloepffer [74] originally called this approach 'LCA new' rather than LCSA. Guinée et al. [84] and Heijungs et al. [88] defined the approach as the 'integrated sustainability assessment with life cycle perspective'. Both naming methods follow the same goal of a

single product system inventory, and, in effect, can be considered an LCSA. This approach to LCSA (Figure 3) comprises three phases: goal and scope definition, modelling, and interpretation, and shows that the use of an LCA practice can serve as a basis for conducting LCSA.

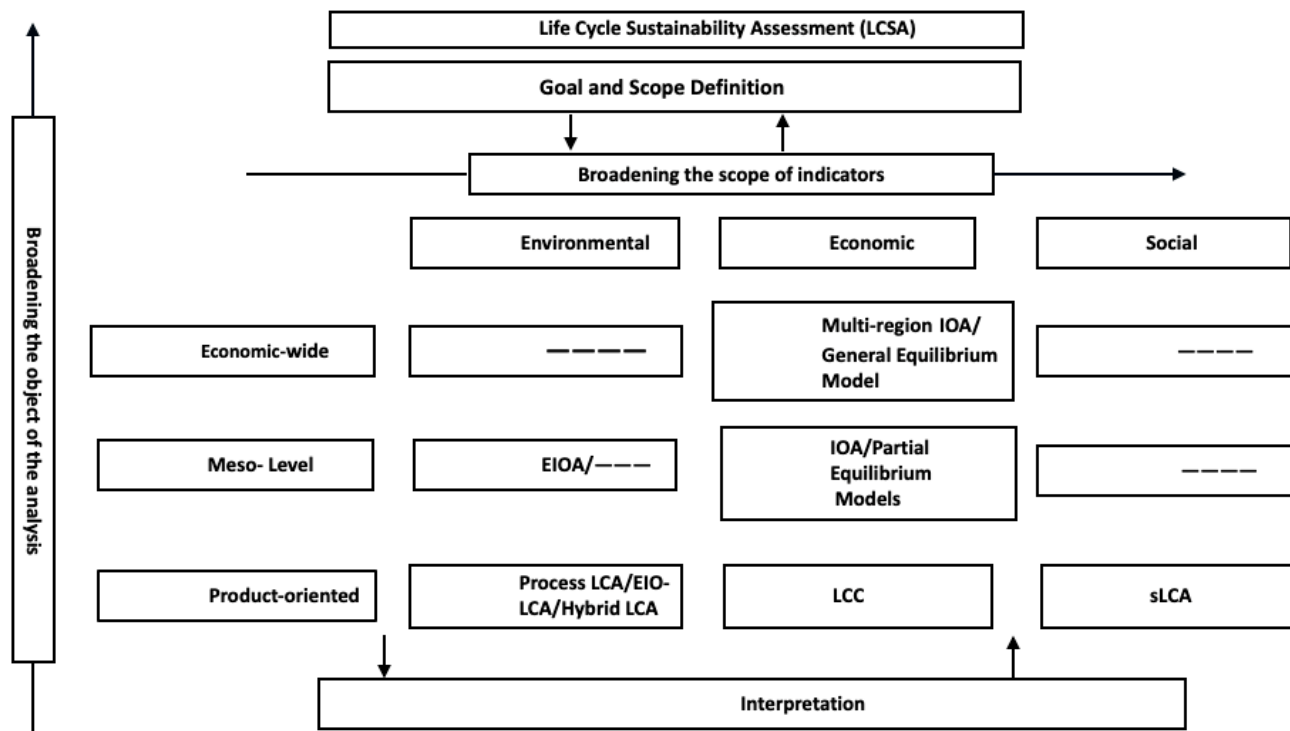


Figure 3. Transdisciplinary integration framework for LCSA (after [82]).

Figure 3 provides an overview of models, methodologies, and tools that can be used for assessing certain sustainability impacts. The selection and application of the appropriate model to a certain sustainability framework can be a major challenge when using this approach [84]; however, it can be addressed using three modelling considerations. The first is ‘broadening the scope of indicators’, where LCSA not only includes environmental impacts, but also economic and social impacts, to cover all dimensions of sustainability. The second modelling consideration is ‘broadening the object of analysis’, where the analysis of sustainability goes beyond products and services and entails perspectives encompassing technologies, product groups, or industries (meso-level), as well as state economies or geographical or political bodies. This is based on the observation that LCA mainly focuses on product-level assessments and the modelling of impacts beyond product-level can gain from wider experiences in economic modelling and environmentally extended input–output assessment (E-IOA). Thirdly, ‘deepening’ is another modelling consideration which highlights the rationale behind LCSA in a way that reveals how sustainability impacts in LCSA are not only physically related to processes or flows of commodities, but are also linked to a system or process through economic, social, political, and cultural relations and mechanisms [85].

While the two approaches to LCSA noted above are different, one aspect they share is the transparency and identification of potential trade-offs between the three sustainability pillars, each affected by varying degrees of maturity and various target roles of each sustainability dimension [89]. However, both approaches share challenges, such as maintaining consistency within the system boundaries and the functional unit, and the weightings attributed to the three sustainability criteria when combining them into a holistic picture of sustainability [86]. The weighting to be given to the three components of LCSA is especially complex and is a challenge that still requires addressing [86], especially as there needs

to be an avoidance of compensation for poor performance in some components by good performance in others. Nevertheless, some promising approaches, such as multi-criteria dimension analysis (MCDA) [84], can be used, and this is a point that will be returned to later in the review.

There have been many published studies on applying LCSA to waste management and energy generation systems, but the number of LCSA studies involving WtE is still minimal. After a review of the literature using Web of Science, only 13 publications could be found when the three keywords “LCSA”, “waste”, and “energy” were used together. Furthermore, only 4 of these 13 papers were review papers that discussed the LCSA methodology in relation to waste management and energy generation [90]. Two of the papers referred to LCSA applied to electricity and power generation, but neither used waste as an energy source [90]. The other two papers discussed waste recycling, with one focusing on the different packaging waste collection systems by only presenting the environmental and social impacts [91], and the other paper focusing on the reuse of solid waste [87], but without focusing on utilizing waste to generate energy. Only one out of the thirteen publications involved vegetable oil-based biodiesel systems [87], which is related to WtE. In this study, the authors integrated LCA and LCC results into one inclusive impact index to make communication with decision-makers easier. However, further research on sLCA was required in this proposed index.

Also, several other research focusing on the application of LCSA for waste treatment exists. An example is a study comparing three collecting systems of used cooking oil, which required 11 environmental indicators, 7 economic indicators, and 11 social indicators to perform the LCSA [92]. Wang et al. [93] also attempted to use LCSA as an optimization model to identify the most sustainable technology for MSW, wastewater, and agriculture waste treatment in Ghana. However, the focus of that study was primarily on the emissions of GHGs as the environmental impact, and job creation as the social impact.

In addition, there are several other issues linked with these tools and their application within the LCSA framework. One is the choice of indicators, which is still a challenge in the LCSA framework, and economic and social impacts are often limited to a few indicators. For example, economic impacts are primarily assessed using the LCC analysis, which does not cover the full dimension of economic sustainability [94]. However, other economic indicators, such as gross value added, profit, import dependency, levelized cost, and profit, etc., can be used to expand the depth of the economic assessment [95]. Presently, research on the applications of social indicators have also not been conducted adequately. Due to the relatively early stage of application, sLCA is limited because of the availability of data, difficulties in data quantification, and subjectivity surrounding social indicators. However, recent works have quantitatively used some key social indicators, such as human health impacts, employment, accidents and safety, public acceptance, life expectancy, public welfare, and equity, in the sLCA of energy and transportation systems [96]. Given this, these quantitative indicators can be developed and made applicable to other LCSA studies, as well. In addition, SETAC and UNEP are continuously working on creating a framework that will include socio-economic impacts and transform the current LCA into a triple-bottom-line sustainability assessment model. Given that the applications of sLCA face some limitations in data availability and absence of standard methodology and tools to obtain product-specific social impact data [97], the social hotspots database of country- and sector-based statistical data has been developed for the screening of potential hotspots at a macro level and to provide social assessments about value chains [98].

#### *4.2. Life Cycle Assessment for Waste to Energy Technologies*

LCA considers all inputs and outputs of materials and energy throughout the life cycle and identifies their environmental impacts and energy consumption [99]. This allows for examining all aspects of a process ‘from the cradle to the grave’ to highlight and evaluate potential environmental impacts from the entire production line [100]. The general LCA framework comprises four main phases as set out in ISO 14040:2006, namely goal and

scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA), and interpretation [68,101,102]. In the case of WtE, the functional unit is typically defined either in terms of input (the amount of the treated waste) or output (the produced energy) of the assessed system; hence, the treatment of 1 ton of waste or the production of 1 kWh of electricity [103].

LCA has long been recognized as a useful tool to support structured and accurate decision-making related to waste management systems such as WtE [104,105]. LCA has also been used to compare different treatments and scenarios in waste management to help select the most appropriate waste management strategy [105–107]. Khandelwal et al. [106] pointed out the development and application of different LCA-based models in waste management since the early 1990s, and many researchers have also used LCA to compare the efficiency of different WtE treatments, including LFGTE, gasification, and incineration in cities. For instance, Malinauskaite et al. [107] examined waste in the energy context of MSW management systems for 10 EU countries, and concluded that WtE processes should be adopted as part of a comprehensive approach towards a circular economy. Another study by Nabavi-Pelesaraei et al. [108] involved the comparison of landfill and incineration for MSW management in Tehran, Iran, and concluded that transportation was the most important energy-consuming process in both scenarios, with incineration reducing toxicity by generating electricity and phosphate fertilizers.

Noya et al. [109] assessed the environmental advantages of alternative management schemes based on low waste generation and renewable energy production in Kazakhstan. They noted that material recovery and waste reduction both had significant environmental advantages. However, it must be noted that site-specific assessments are important for evaluating waste management systems. As a result, there are differences between various LCA models applied for waste management as environmental impacts of MSW differ from region to region, primarily because of differences in waste composition [110]. This variation has been observed in many published studies for different countries, e.g., [110] for India, [111] for China, [112] for Germany, [113] for Japan, and [114] for England. Similarly, Zaman [115] conducted an LCA to investigate sanitary landfill, incineration, and gasification-pyrolysis technologies based on input–output materials flow, and concluded that a sanitary landfill with energy generation had the lowest environmental impact among the three examined. Nubi et al. [116] also used LCA as the primary analytical approach to assess and compare the environmental impact of four distinct WtE technologies, diesel backup generators (DBGs) and grid electricity for two case study cities in Nigeria. The study revealed that AD has the capacity to provide additional electricity supply to the grid and at the same time minimise the dependence on DBGs, which potentially have a positive impact on the environment as shown in Table 1. However, it should be noted that in the case of WtE, environmental impacts alone cannot comprehensively reflect all aspects of the system. For instance, some advanced WtE techniques can better control environmental impacts, but can also generate economic or social burdens [110].

**Table 1.** Life cycle environmental impact assessment characterization results for WtE systems, diesel back-up generators, and national grid supply per 1 kWh of electricity for Abuja, Nigeria [116].

| Impact Category                                  | Unit                                  | Anaerobic Digestion | Incineration | Gasification | Landfill Gas to Energy | Diesel Back Up Generators | Grid Electricity |
|--------------------------------------------------|---------------------------------------|---------------------|--------------|--------------|------------------------|---------------------------|------------------|
| Abiotic Depletion Potential (Fossil Fuels) (ADP) | (MJ)                                  | 0.618               | 3.17         | 6.4          | 4.59                   | 14.1                      | 8.69             |
| Global Warming Potential (GWP)                   | (kg CO <sub>2</sub> eq)               | 0.507               | 0.804        | 0.858        | 4.88                   | 1.02                      | 0.497            |
| Human Toxicity Potential (HTP)                   | (kg 1,4 DB eq)                        | 0.00548             | 0.0102       | 0.0195       | 0.019                  | 0.0732                    | 0.0117           |
| Photochemical Oxidation Potential (POCP)         | (kg C <sub>2</sub> H <sub>4</sub> eq) | 0.000106            | 0.0000396    | 0.0000464    | 0.00103                | 0.000198                  | 0.0000406        |
| Acidification Potential (AP)                     | (kg SO <sub>2</sub> eq)               | 0.000564            | 0.000889     | 0.000974     | 0.00299                | 0.0129                    | 0.000296         |
| Eutrophication Potential (EP)                    | (kg PO <sub>4</sub> eq)               | 0.00144             | 0.000192     | 0.000209     | 0.000717               | 0.00313                   | 0.000061         |

Table 1 shows that the most environmental impact in all categories except GWP and POCP, were from DBGs while LFGTE had the most environmental impact in terms of GWP and POCP due to CO<sub>2</sub> and CH<sub>4</sub> emissions respectively with incineration having the least environmental impact for POCP. National Grid electricity had the least environmental impact in 3 out of 6 selected categories (GWP, AP and EP) while AD had the least impact in ADP and HTP.

For the LCA, the results provide vital information required by decision-makers, particularly with regards to creating a strategy that is environment-friendly as well as in identifying strengths and weaknesses when optimizing technologies such as those of WtE. For this reason, the selection of the system boundary should include all significant processes, starting with the collection and transportation of MSW and ending with electricity generation. Here, the MSW, energy, and mass are the input to the WtE system, and the outputs considered are the emissions (air and water) and electricity generation from the processes. The system boundaries selected for this study include direct emissions such as emissions associated with different WtE systems (AD, incineration, gasification, and landfill gas to energy) and the indirect emissions, like fuel requirement and electricity supply. The system boundary makes the study simpler, helps in making comparisons among options, and makes decision-making easier [110]. The choice of the impact categories used is based on their relevance, applicability, and data availability. Adeleke et al. [117] adopted this approach during the LCA study of the current, fledging, and alternative waste management systems in Johannesburg, South Africa, using GWP, ozone depletion potential (ODP), AP, EP, and PODP as impact categories based on their importance to the case study. Zaman [115], on the other hand, used all of the impact categories to conduct an LCA on MSW treatment technologies in Sweden. It should be noted that the characterization analysis of these impact categories should be carried out based on the LCI result using relevant characterization factors with the results exhibiting a reasonable level of accuracy to differentiate certain impact categories that have closely matched values. However, there can be some methodological limitations, such as the possibility of inconsistencies and discrepancies in inventory data due to variations in technologies and the geographical and temporal scope of the studies, leading to several uncertainties in results. This could limit the value of the LCA study for the future application of WtE technologies in society according to [115].

Thus, future LCA studies on WtE should address the issue of uncertainties by using valid mathematical models in the LCA calculations as part of evaluating the robustness of the LCA results, considering that there exist various methods for assessing uncertainties within the LCA, such as the Monte Carlo method [118]. Environmental costs should also be considered in future LCA studies for WtE. Additionally, when conducting comparative

LCA studies to assess the environmental impacts that could potentially arise from adopting WtE, it is important to consider the implication of any similarities or differences in terms of waste composition, geographical location, and socio-economic status and their effects on the environmental impacts. This is important in terms of the scalability and transferability of the LCA findings across different geographic and socio-economic contexts within developing countries. This is because any similarities or differences in findings may or may not affect the WtE scenario outcomes in terms environmental impacts. Rana et al. [119] demonstrated a similar finding when performing the LCA of MSWM strategies in the Tricity region of India. In this study, all of the scenarios that took into account the combination of recycling, composting, and sanitary landfill had the lowest environmental impacts for three different cities in India. Regarding policies, the lack of accurate findings from a proper environmental impact assessment of the different proposed scenarios can make policies and decisions surrounding selecting the most suitable technologies a challenge. Furthermore, the development of policies based on methods that are not ideal can cause serious problems. Because of this, LCA assessment is a useful tool in assessing the environmental impacts of selected or proposed WtE systems [120].

However, it should be noted that in the case of WtE, environmental impacts alone cannot comprehensively reflect all aspects of the system. For instance, some advanced WtE techniques can better control environmental impacts, but can also generate economic or social burdens [121]. Hence, taking into account the intricate balance of optimizing the three pillars of sustainability for WtE should be part of future frameworks. This will form a key part of the evolution of a more holistic LCSA framework required to help the governments of developing countries create suitable policies for sustainable waste management and electricity supply.

#### 4.3. Life Cycle Costing for Waste to Energy Technologies

LCC is an economic assessment method covering all costs from project inception to final disposal. This consists of all costs of operation, repairs, maintenance, and energy consumption, in addition to the initial costs of development, and all of the costs are discounted to the same point in time [122]. Kloeppfer [74] stated that this economic counterparty to environmental LCA has a methodology that includes and assesses all costs relating to a system over the entire life cycle [123,124]. Generally, LCC follows the ISO 14040/44 framework, and the objectives and scope are similar to those of LCA. LCC is usually based on the same functional unit as LCA, especially when these two aspects are investigated in the same study. However, unlike LCA, there is no comparable impact assessment phase in the LCC. This is because all inventory data consist of a single unit of measurement, and as a result there is no requirement to characterize LCC inventory data. However, one difficult aspect of LCC is the proposed reporting of all costs over the entire life cycle, where different actors bear those costs [125].

According to Hunkeler et al. [126], there are three distinct types of LCC: conventional LCC (type I), environmental LCC (type II), and societal LCC (type III). However, the application of LCC in the management of MSW is rare, although there has been an increase in recent years. For WtE projects, LCC potentially provides a systematic method for assessing economic feasibility over their life span [127], and studies have shown that it can be a powerful tool for generating an economic analysis of waste management systems [128]. For instance, Dong et al. [129] applied LCC methodology to assess and compare three MSW treatment technological options in China: landfill, landfill with biogas conversion to electricity, and fluidized bed incineration with energy recovery. They showed that landfill with biogas conversion to electricity offered the best LCC was a more sustainable option from economic perspective with a net LCC of 16.57 yuan per ton of MSW, in comparison to 34.78 yuan per ton for incineration and 45.87 yuan per ton for landfill. Li et al. [130] also used LCC to investigate the cost of a 600 ton per day MSW incineration power plant project in China. This study used an LCC model that considered the time value of acquisition, operating, maintenance, and fault costs.

The result revealed that LCC and total revenue from the MSW project were  $587.7 \times 10^6$  yuan and  $746.4 \times 10^6$  yuan, respectively, for operations over a 20 year service period. Slorach et al. [131] used LCC to four waste treatment alternatives in the UK, namely composting, incineration with energy recovery, AD, and LFGTE. The results indicated that incineration had the lowest LCC (£71/ton), while LFGTE had the highest LCC (£123/ton). Al-Wahaibi et al. [132] conducted an LCC on WtE systems in Oman, revealing that AD systems were not economically feasible due to the low selling price of the produced biomethane. These findings illustrate the substantial influence of local marketing conditions on the economic feasibility of WtE systems. In addition, Abdeljaber et al. [133] used LCC to examine the economic viability of including gasification and MBT as part of integrated solid waste management strategies. The findings revealed that in comparison to the conventional incineration and AD-based alternatives, the MBT- and gasification-based scenarios were the most economically viable, with NPVs of \$364 and \$284 million, respectively. Sharma and Chandel [122] used LCC to assess and compare the cost of MSW management systems under different scenarios. The LCC analysis was conducted for six integrated MSW management scenarios for Mumbai City, India, which generates more than 9000 metric tons of MSW a day and disposes of most of it in open dumps. The scenarios consisted of a combination of recycling, composting, anaerobic digestion, incineration with electricity generation, and landfill with biogas recovery. As part of performing the LCC analysis of scenarios, the present worth (net present value) method was applied.

The net present value (NPV) of operations and maintenance (O&M) cost and revenue generated was calculated using a discount rate of 11.25% for a 20 year life span. The results indicated that the incineration-based scenario was the least cost-effective option, with a net LCC of US\$38 per ton of MSW due to the high capital cost associated with incineration. Conversely, the scenario with a combination of recycling and sanitary landfill was the most economically viable option, with a net LCC of US\$19 per ton of MSW due to the comparatively lower operating cost.

Hadidi and Omer [134] analyzed the present situation of MSW management in Saudi Arabia by proposing an economic model to determine the viability of WtE investments in Saudi Arabia in order to solve its waste management problems and meet its projected energy demands. This model assessed the economic viability of WtE plants using gasification and AD conversion technologies. The model presented a cost estimate for the establishment of both gasification and AD WtE plants in Saudi Arabia through a set of economic indicators, such as NPV, internal rate of return (IRR), modified internal rate of return (MIRR), profitability index (PI), payback period, discounted payback period, levelized cost of electricity (LCOE), and levelized cost of waste (LCOW). Finally, the assessment of the model indicated that the main factors influencing the investment decision of gasification plants were the waste capacity, generated revenue, and the capacity factor. Likewise, the study also identified facility waste capacity and the capacity factor as the major influencing factors on the AD plants' investment decision.

However, there are challenges when using LCC in LCSA, and these arise from the fact that LCC does not fully cover all the economic concerns that are relevant within sustainability. This is because LCC and the summing up of costs require supplementary indicators to provide a more detailed analysis of economic sustainability in LCSA, including identification of possible trade-offs [135]. Hence, conventional LCC approaches have to be expanded to better connect the environmental and social aspects of sustainability. In addition, there is also the absence of a standardized weighting method, the absence of market price or other penalties for emissions, and the challenge of indicating complex environmental issues in monetary units [130]. For example, a variant on traditional LCC called 'environmental LCC' seeks to monetize environmental impacts within an LCC in order to identify hotspots in both cost and environmental impacts, and this can better serve as a suitable method for incorporation into LCSA. One distinct characteristic of environmental LCC is the internalization of external costs that are expected to build up in the future, along with the already monetized externalities [135].

Since it is complemented by LCA, measures and impacts are not converted from environmental to economic terms, and there is the avoidance of double-counting of any environmental impacts or externalities. This latter point is important as LCC, both environmental and social, needs to be lined up with LCA and sLCA to avoid double-counting effects, and both approaches can be used separately to monetize externalities [135]. At present, there is no standard guideline for environmental and social LCC, and it is not aimed at being an alternative for sustainability assessment that uses a macroeconomic system including all societal actors and monetizing all externalities [135].

In terms of the methodological approaches for modelling and calculating processes economic impacts, the LCC is one technique used to assess all relevant expenses of a product or system project over its lifetime [136]. It considers the initial costs, future costs, and any resale, salvage, or disposal costs, over the lifetime of the project or product [137]. In comparison to other techniques such as techno-economic analysis (TEA), LCC depends on a wider regulatory basis in terms of sets of legal regulations, standards, and guidelines than the TEA [137]. In addition, LCC tends to have a higher level of logic than TEA. As such, it supports the proper decision-making process by summing up and estimating costs into easily read figures, as well as showing and weighing in the effect of various factors, such as the time value of money and other uncertain economic factors, on decisions. The cost generally consists of all costs related to production, operation, maintenance, and retiring/disposing of a product from the cradle to the grave [136].

For example, Li et al. [130] performed a cost analysis of a MSW incineration power plant project in China using the LCC method. In this study, the calculation was such that the LCC was divided into the acquisition cost, which occurred during the construction period, the operational cost, which was made up of electricity generation cost and employee wages, the maintenance cost, comprising routine equipment maintenance costs, annual maintenance costs for waste cranes, and the cost to replace bag filters, and the fault cost, which was linked to overhauling malfunctioning equipment to restore normal function, and the disposal cost incurred after decommissioning.

Sensitivity analyses should be conducted on developed economic models to point out the key factors that the decision-makers and investors should focus on more when assessing the various technologies. For example, Ayodele et al. [138] showed that the NPV being highly sensitive towards changes in electricity generation efficiency was an indication that decision-makers and investors should investigate the possibility of employing innovative and enhanced technology in engine design that can increase electricity generation efficiency and make WtE technologies more profitable.

Additionally, these sensitivity analyses could prove useful, as changes in these types of parameters can allow decision-makers to investigate the various uncertainties that could emerge while considering the impact on the overall cost-effectiveness of the project during the planning phase. For instance, the waste capacity could be key in allowing decision-makers to decide how to ensure project profitability while taking into account the availability of waste [139]. The same applies to the impact arising from changes in the selling price of electricity. Given the current trend, increases in electricity prices could lead to higher returns from the sale of the electricity generated from WtE, and therefore higher NPV. However, the issue of volatility in prices should also be taken into account with volatile electricity prices implying delayed investment in WtE, as opposed to early investment at more predetermined prices of electricity. Given this, it is clear that higher uncertainty in prices may further hold up the investment, as most investors are opposed to accepting a high-risk project, with only the expectation of receiving higher returns making for the risk [140].

However, a major challenge concerning WtE in developing countries is from an investment perspective, considering the already high tariff paid for electricity. In developing countries like Nigeria, the government has recently taken steps to address this issue by reducing the tariff to further encourage foreign investors' participation in developing the energy sector of the country [141]. The introduction of feed-in-tariffs could play a ma-

jor part in attracting private investments in new installations of WtE/renewable energy capacity in Nigeria. Feed-in-tariffs (FITs) refer to the price per unit of electricity above the electricity market price, which are fixed in contracts that a government commits to for purchasing renewable energy by signing a long-term contract with renewable energy suppliers [141]. These help to lower the LCOE of energy generated from renewables such as WtE by offering cost-cutting based on compensation to the producers of renewable energy (RE) sources, along with guaranteed grid access and ensuring that investors are paid a cost-based price for providing renewable electricity to the grid in addition to long-term contract agreements [142]. Given this, there is a need for policies requiring consumers to buy all of the grid-connected electricity produced from renewable sources, thereby making FITs for WtE and other RE technologies favorable compared to conventional sources of energy, such as fossil-fuel based electricity supplies.

For WtE projects, the introduction of gate fees could also prove useful in making the price of electricity cheaper [143]. The collection of gate fees for the waste delivered to the plant as an additional source of revenue for investors could in turn make the WtE project more profitable [144]. In the UK, for example, it has been revealed that the average gate fee (including transportation) for WtE plants has a range of 65–145 £/t, which equates to about 79–177 US\$/t according to the WRAP gate fees report 2021/22 [144]. The government introducing a gate fee on WtE plants improves the economy of scale of the various WtE technologies, and as a result, investors will be able to achieve financial benefit from the collection and processing of substantially large amounts of waste. Nevertheless, this could change in future with the introduction of legislation and policies. Therefore, it is important to examine the effects of gate fees, as well as those of FITs, on the profitability of WtE plants for future research, in order to determine their effect on the overall economic viability of various WtE projects.

The impacts of introducing carbon credits as an additional source of revenue for such projects should also be considered. Unaegbu and Baker [145] discovered during their assessment of the potential of electricity generation from MSW in Nigeria that Lagos and Abuja, through the adoption of WtE, could experience carbon emission reductions of 2835 million kgCO<sub>2</sub>e and 324 million kgCO<sub>2</sub>e per year, respectively. This could potentially yield carbon emission reduction (CER) earnings of US\$113 million and US\$7 million. Given this, further research needs to also be performed to determine the effect of carbon credits on the economic viability of WtE projects in developing countries, alongside those of revenue from the sale of biogas for cooking and digestate used for fertilizer during AD, and the sale of the bottom ash generated during incineration, which could be used as a construction material.

#### 4.4. Social Life Cycle Assessment for Waste to Energy Technologies

sLCA is used to evaluate the social impacts of products and services throughout their life cycles. It does this by measuring impact categories and subcategories associated with products and services that could have a positive or negative impact on the stakeholders based on predetermined life cycle stages [146]. Compared with other social impact assessment (SIA) techniques, sLCA differs in terms of its objective, scope, and systematic nature [147]. For instance, while sLCA assesses products and services based on the predetermined life cycle stage and system model, social impact assessment (SIA) involves analyzing, observing, and handling the social outcomes of planned projects, policies, or programs before a project is initiated [148]. Likewise, sLCA differs from the recently developed social organizational life cycle assessment (SO-LCA) approach, which only focuses on organizations. sLCA can be conducted individually or in combination with LCA and LCC [149]. sLCA also uses the LCA framework and can be regarded as a social counterpart to the LCA [146]. The implication here is that sLCA follows the four steps of the LCA [150], and the UNEP-SETAC guidelines (UNEP-SETAC, 2009/2020) for sLCA have been provided as a methodological technique for assessing social impacts of products or services throughout their life cycle [151]. Since its establishment, the UNEP-SETAC's sLCA guidelines and

methodological sheets have been broadly used by researchers and others. However, there are challenges regarding the use of sLCA for quantitative analyses of social impacts, related to being unable to adequately sum this information per functional unit. In sLCA, it is usually challenging to relate social impacts to an exact functional unit as in LCA, because the functional unit in sLCA often relates to the project itself, rather than to a quantified unit.

A key difference in methodology between sLCA and LCA is the definition and selection of stakeholder categories. This selection can potentially influence the results of the sLCA. Additionally, for the sLCA, decision-makers must consider the social impacts of numerous potential social issues [152]. Indeed, one weakness of sLCA can be the absence of validity, with the judgment of those applying it playing a vital role in its assessment. For example, according to the UNEP-SETAC guidelines [152], the framework consists of six stakeholders (workers, local community, society, consumers, value chain actors, and children). These groups each have social subcategories and indicators. The UNEP-SETAC guidelines list more than 100 indicators to assess social impact, many of which are unclear [152]. A key aspect of sLCA in the context of LCSA is that it needs to be applied within the constraint of an identical functional unit and system boundary as the other two (LCA and LCC) assessment methods [152]. This perhaps explains why the UNEP-SETAC guidelines for sLCA have been provided as a methodological approach for assessing any social impacts of products or services throughout their life cycle [152], and thus have flexibility at their heart rather than seeking to be prescriptive. In the case of WtE processes, several social indicators can be of direct concern, such as contribution to local employment [152]. An example is seen in a study conducted by Manik et al. [153], which assessed palm oil biodiesel and involved 24 social criteria, subdivided into 5 social impact categories with weighting factors. These social indicators may be expressed in quantitative, semi-quantitative, or qualitative terms. Data collection for sLCA is usually undertaken via on-site observations and interviews with key stakeholders using questionnaires [154].

As part of facilitating an sLCA, some databases have been created for generic data, such as the Global Trade Analysis Project database and the Social Hotspots Database (SHDB) [155]. However, the results obtained using generic data from statistical databases tend to provide a broad (country-level) estimate of social factors such as child labor, unemployment, and income inequality; however, these can vary significantly within a single country.

In contrast, site and context-specific data can reflect social impacts more precisely [156]. Furthermore, although the UNEP-SETAC guidelines provide a framework for sLCA, there remains an absence of standardized methodology for evaluating social impacts from inventory data. Additionally, in the case of WtE in developing countries, the availability of adequate data due to the non-existence of WtE in these countries has been a key issue. Hence, in many cases, scoring methods have been used for social impact indicators, and an example is provided by [155] with their use of a scoring system with seven impact levels from 1 = minor and 7 = most important. Rather than a scale, a binary system, where a value of 1 represents yes, and a value of 0 means no, has also been used [155]. Other methods have been proposed for scoring and weighting impacts, and examples are the Performance Reference Point and Impact Pathways methods [155–158], but these scoring systems remain subjective and are full of uncertainty.

Some sLCA-based case studies have been published on waste treatment, but only a few of these have specifically been on WtE systems. For instance, Foolmaun and Ramjeea-won [159] conducted sLCA studies on the treatment of used plastic bottles in Mauritius, and identified three categories of stakeholders (workers, society, and local community) and eight subcategory indicators (child labor, fair salary, forced labor, health and safety, social benefit and security, discrimination (for worker stakeholders), contribution to economic development and job creation (for society stakeholders), and community engagement (for local community stakeholders)) were chosen to explore social impacts. The results indicated that recycled PET flake production obtained from used PET bottle wastes after

a series of procedures such as sorting, washing, grinding, and drying for recycling, can create better social impacts than incineration and landfill. Lu et al. [160] also used the idea of life cycle inventory analysis and sLCA by including four types of stakeholders (workers, society, local community, and value chain actors) and fifteen inventory indicators to assess the criteria, regulations, and management conditions for WtE incineration in Taiwan. The study aimed to first identify materiality issues and other possible issues that could enhance/improve GHG management of WtE incineration plants in Taiwan by using qualitative analysis to analyze domestic laws and regulations. These aims were based on previous management systems and intricate issues associated with GHG, energy, and solid waste treatment in the country.

Nubi et al. [161] also used sLCA to determine the social impacts that could emanate from prospectively generating electricity from MSW in Nigeria using two case study cities (Lagos and Abuja). Rather than utilizing pre-determined social impacts, Nubi et al. [161] employed a participatory approach to highlight the key social issues that could serve as their sLCA impact subcategories. The findings from the study showed that the impact subcategories, such as “Improved Electricity Supply” and “Income”, were ranked, respectively, as having the highest and the lowest social impacts that related to the potential adoption of WtE in the cities. The research revealed that the social impact for WtE electricity generation in Lagos was higher than the social impact for WtE electricity generation in Abuja. However, this study did not further differentiate or compare the respective WtE systems to ascertain which option would be the most socially acceptable in either city due to the methodological approach adopted for the sLCA. This was different from the study conducted by [162], which used social indicators such as public health risk, new job creation, and community acceptance to assess the social sustainability of WtE in Bangladesh, using comparative measures to quantify the social indicators. In this study, any overlap was avoided when selecting the social indicators, and they were completely different from the economic and environmental indicators. However, they were difficult to select, as they were mostly qualitative, and were selected on the basis that their impacts could be scaled as high, medium, moderate, or low in relation to the WtE generation technologies considered in this study.

Additionally, the impacts were assessed on the basis of being direct or indirect and being positive or negative. In this paper, only direct and relevant impacts were considered. In addition, the selection of social indicators was made so that they were functional, credible, value relevant, and justifiable [163].

For the sLCA, the methodology should be based on the UNEP sLCA guidelines so that key social impact issues and social impact subcategories pertinent to the potential implementation of WtE, such as health and safety, employment, gender equality, and social justice, can be identified. A set of relevant indicators, such as the level of expected accidents/injuries/fatalities and level of job creation opportunities relating to WtE, can be developed to calibrate the relative importance of the different social impact subcategories for WtE through a combination of interviews and questionnaire-based surveys conducted with different stakeholders (workers, local community, and consumers).

Additionally, sLCA studies for WtE should be conducted at the subcategory level for the conversion of both qualitative and quantitative indicators to allow the subcategories to be ranked and calibrated. The benefit of this is that it is possible to provide a complete perspective on the performance in an sLCA by identifying the subcategories with the highest or lowest impacts [164].

However, an issue that needs further research in future work is that of assuming equal weightings for the indicators within each sub-category. Such methodological aspects within sLCA are symptomatic of its emerging nature and the fact that assessing social impacts in sLCA is not yet based on a fully comprehensive and widely accepted theoretical knowledge [165]. This perhaps could be seen as a limitation in sLCA studies where the basis for using scoring (and weighting) to assess social issues/impact categories are subjective. However, this cannot be avoided, particularly due to the absence of adequate data and the

need for quantification. For prospective WtE studies, there is the issue of uncertainties that attend preferences and perception over a future condition. Given this, further research is required to improve characterization models that will enable results to be better aggregated and compared. This can therefore be used to explain the appropriate usage of various systems/processes. This should be incorporated with the development of standardized social impact categories that are sensitive to regional differences, along with the use of innovative data collection methods such as mobile technology or social media to gather social data.

From the above, it is clear sLCA is not as mature as LCA and LCC [166], and several challenges exist in applying this approach, such as identifying and selecting suitable indicators, impact categories, and characterization systems [167,168]. Hence, more sLCA studies are needed to provide a better understanding of this tool, along with clearer communication about existing or potential impacts identified in the life cycle of a product, process, or service [169].

#### 4.5. Integration of LCSA Components: LCA, LCC and sLCA

According to the UNEP-SETAC Guidelines, LCSA involves integrating the outcomes of LCA, LCC, and sLCA and the reflection of the system's performances in all its aspects [169], thus making LCSA an interdisciplinary integration model [170]. LCSA integration refers to how the results of LCA, LCC, and sLCA are combined and presented, and remains a major challenge despite the fact that the number of studies that have used operational research methods to help achieve it have increased [167]. The challenges include the maintenance of consistency within the system boundaries and the functional unit, and the application of any weightings attributed to the three sustainability domains when attempting to develop an overall quantitative integration 'score' for the LCSA outcomes [168].

Nevertheless, some promising approaches have been applied to this quantitative integration challenge. For example, Foolmaun et al. [171] developed an approach based on the analytical hierarchy process (AHP), which kept the consistency of system boundaries. Multi-criteria decision analysis (MCDA) approaches have also been used [170], following three main approaches:

1. Multi-attribute decision-making (MADM) methods, which are used to assess a finite set of options based on multiple criteria attributes;
2. Multi-objective decision-making (MODM) methods, which are used to identify and evaluate Pareto optimal solutions on the efficient frontier of a mathematically constrained solution space;
3. Data envelopment analysis (DEA), which is applied to analyze the efficiency of a sample of alternatives if the efficient frontier is not known [170].

In relation to the sustainability of non-waste management systems and non-WtE systems, there exist studies such as those of Guo et al. [172], who conducted an LCSA of pumped hydro-energy storage in China and used a MCDA in the form of a multi-attribute value theory (MAVT) approach to assess the complete performance of energy storage. The results showed that conventional pumped hydro-energy storage (CPHES) had a better performance in terms of the economic impact and less environmental impact than underground pumped hydro-energy storage (UPHES) because of the economies of scale. In contrast, the UPHES had a better social impact performance due to the absence of stages of excavation and backfilling.

Roinioti and Koroneos [173] also used MAVT in the LCSA study of the Greek interconnected electricity system for the sustainability assessment of the different electricity options. The trade-offs between environmental impacts, costs, and social implications in their study affected the choice of the most sustainable option. Wind energy was identified as the electricity option with the overall best sustainability performance, both when the same weighting was applied to all three sustainability dimensions and when environmental and economic criteria were weighted as the most relevant dimensions. However, photovoltaics

emerged as the most preferable option when the social aspect was a preference, due to the considerable employment effects associated with the industry.

All these studies focused on integrating the three life cycle assessment methods (LCA, LCC, and sLCA) within an LCSA framework through the representation of various case studies. However, these life cycle methods still have their distinctions, so they cannot be integrated and merged without some adaptation. Given this, Kabayo et al. [174] applied a ranking and scoring approach to obtain single scores for the LCSA of electricity generation systems by aggregating the LCA, LCC, and sLCA results. This enabled a conclusion that of the ones assessed, small hydro was the most sustainable system from an environmental, economic, and social perspective.

A few published LCSA studies have not made an attempt to quantitatively integrate the three life cycle assessment tools, and have instead adopted a 'multi-domain' narrative approach to integration and presentation of LCSA outcomes. For instance, Shrivastava and Unnikrishnan [175] performed an LCSA of crude oil in India using a framework that represented the results for all three life cycle assessment tools in a separate way, with no attempt to quantitatively integrate them. The study revealed that for the environmental dimension, the emissions leading to the environmental impacts were mostly from the oil refinement and transportation phases, according to LCA results while for the LCC, a new economic method needed to be developed for the study because of the more complex operations of the oil refineries. For the social dimension, it was found from the sLCA that the industries had established a strong relationship with their stakeholders, particularly with the workers and consumers, and emphasized the need to improve various subcategories such as working hours, equal opportunity, health and safety, feedback mechanism, consumer privacy, and end-of-life responsibility. These all led to recommendations for sustainability measures that could enhance the environmental, economic, and social domains, but these were treated separately.

Lu et al. [176] used a similar approach to conduct a sustainability on the reusability of electrical and electronic products and components using LCSA as part of efforts to improve the waste electrical and electronic equipment (WEEE) management policy in China. Their results showed that the reusability of components was high, showing that reuse is more preferable than basic materials recovery, particularly from an environmental and economic perspective, as noted in the LCA and LCC results. However, the sLCA results revealed there were challenges in ascertaining whether job creation was more important than health risks, thus making the aspect of social reusability ambiguous. Regardless, the study concluded that component reuse should be supported accordingly in the policy-making process for both government and recycling industries.

Another interesting example of this multi-domain narrative approach to LCSA presentation, albeit unrelated to waste management, was provided by Gulcimen et al. [177] in a study concerning the LCSA of a light rail transit system. Their results showed that for the LCA, the light rail system's global warming and abiotic depletion potential per passenger km were 0.024 kg CO<sub>2</sub>eq. and 0.27 MJ, respectively, with a service life of 50 years. The LCC of the light rail system was calculated as US\$0.046 per passenger km, with the main contributor to the total life cycle cost being energy (92% of the total cost; US\$2.88 × 10<sup>8</sup>). For the sLCA, it was found that the industry performed well for society, the local community, and workers, but performed poorly for the consumer because of the weak feedback mechanism.

Traverso and Finkbeiner [178] introduced another more visual approach to LCSA representation by adopting the life cycle sustainability dashboard (LCSD) during the LCSA of marble slabs production in Italy where four marble slabs—Marble A (Perlato di Sicilia 1), Marble B (Perlato di Sicilia 2), Marble C (Bianco Carrara) and Marble D (Bianco Carrara)—were used as case study. This approach involved the insertion of inventory data and indicator sets that were used for LCA, LCC, and sLCA into the LCSD database. The findings revealed that Marble C (Bianco Carrara) had the best performance based on the assessment of the three techniques. Furthermore, there were trade-offs in the cases

of Marbles A, B, and D, as Marble B had a better sLCA performance than Marble A, but showed a worse LCA and LCC performance than Marble A. On the other hand, Marble D had the best economic performance of the four marble slabs, but had poor social and environmental performance.

In terms of the sustainability of waste management systems and WtE systems, Iacovidou and Voulvoulis [179] used an MCDA as part of a sustainability assessment technique in developing a screening and decision support framework for the comparison of the sustainability performance of food waste management alternatives. Soltani et al. [180], focused on the waste management system and the WtE industry in Canada, developed a weighting scheme in a multi-criteria decision-making (MCDM) approach for the aggregation of the results of environmental and economic assessment. They then used game theory to engage multiple stakeholders in the decision-making and execution processes in the WtE industry. Nubi et al. [181] adopted two quantitative integration approaches. The first approach involved ranking and assigning scores to each WtE system within each impact category, where a score of four indicated the most favorable option, and a score of one indicated the least favorable option.

The overall aggregated score for each life cycle domain (LCA, LCC, and sLCA) was divided by the number of individual environmental impact categories per domain before summing each overall domain score into a single score for the LCSA. This representation aimed to provide a more straightforward identification of areas of significant impact and a platform for possible sustainability improvement. Nubi et al.'s second approach involved using MAVT, which considered all three aspects of sustainability and allowed for compensation between them. Such an allowance for compensation between the three aspects of sustainability is often required in policy applications [182]. Here, the first step was to calculate the scores for each sustainability dimension (environmental, economic, and social). This depended on the values of the related sustainability indicators determined in the sustainability assessment and their weights of importance. Using the scores for the sustainability dimension evaluated in the initial step and the weights of importance for each dimension, the sustainability dimensions were used to calculate the total sustainability score of the scenarios in the second step. The MCDA was performed by assuming equal importance for all of the aspects, considering that the same weights were also given to the indicators for each sustainability aspect to avoid bias. The first approach was similar to that of Kabayo et al. [174], while the second approach was in line with the approach of Guo et al. [172]. On the basis of both integration approaches, the sustainability score associated with adopting WtE in Lagos (2.99 and 2.97) would be approximately 8–9% higher than for Abuja (2.74). The results from both approaches showed that adopting WtE provided both cities with benefits from a sustainability perspective, with these benefits slightly more so for Lagos than Abuja. In another study conducted by Khan and Kabir [182], the use of a MCDA to integrate the LCA, LCC, and sLCA components was also employed to evaluate the sustainability of four different WtE technologies (AD, incineration, gasification, and pyrolysis) in Bangladesh. Here, the sustainability assessment results of each WtE system showed that AD, pyrolysis, and gasification were 111%, 65%, and 33% more sustainable than incineration in terms of their respective sustainability performances. Hence, it was concluded that AD and incineration were the most and least sustainable WtE systems, respectively. Other studies have taken a different approach regarding the quantitative integration of LCA, LCC, and sLCA results.

Foolmaun and Ramjeeawon's study [171], referred to in the previous section, integrated all three assessment tools into single scores by applying AHP to aid identification of the most sustainable option (75% flake production with 25% landfilling) for the management of post-consumer PET bottles in Mauritius. Tsambe et al. [183] used LCSA to assess the sustainability of two used lubricating oil (ULO) management systems in Brazil and employed sustainability indices that aggregated data from all indicators to define the best-case scenario.

The sustainability index was developed from eight environmental indicators, five economic indicators, and five social indicators, and the results indicated that the TsTR (transportation without trans-shipment and later re-refining) scenario had the most social and economic performance, while the TTR (transportation with trans-shipment and re-refining) scenario had the least environmental performance. Overall, the TsTR scenario had the best sustainability assessment values compared to the TTR scenario.

## 5. Discussion

LCSA has been utilized in several sectors, including transportation [167], building [184], energy [185], agriculture [186], manufacturing [187], and waste treatment [176]. The rising number of publications on this topic and the various case studies show that LCSA is increasingly being adopted as an assessment tool to support sustainability decision-making and can be applied to WtE systems [188]. It is also apparent that LCSA is in a very dynamic, emerging period of its development, and the complexity, uncertainty and inconsistency of different methods, particularly for the integration step, remain subject to ongoing refinement [189]. LCSA has also been applied in various geographical contexts, spanning both the developed and developing worlds [189,190]. The representation of results based on the objectives of any given study is a crucial phase to ensure that LCSA is effective [191]. However, LCSA's relatively early development stage reveals numerous challenges, particularly in its consistent operationalization, thus making it challenging to compare results between studies, even within the same sector and geographical context. It is a complex approach, and its three components (LCA, LCC, and sLCA) have different metrics and varying levels of maturity [190]. The inclusion of both qualitative (often for sLCA) and quantitative (primarily LCA and LCC) indicators makes clarity and equivalence difficult [191]. Additionally, the interactions and trade-offs among the three sustainability dimensions need to be taken into account in greater depth [190], and the findings must be communicated to decision-makers. It is intriguing that UNEP-SETAC (2020) do not suggest any specific approach to aggregating and weighting the results of the three elements of LCSA, and all the approaches noted above have their advantages and disadvantages. There is also the issue of managing interactions and changing relationship between indicators that could generate overlapping or double-counting of some effects, which still needs to be resolved. For example, Wulf et al. [192] identified the problem of indicators overlapping between sLCA and LCA, especially in human health and resource use. Aside from the fact that data collection can be challenging within sLCA when evaluating the social impacts, there is still a need to clarify how social impacts ought to be resolved and related to the functional unit.

A particular challenge in LCSA is the issue of weighting indicators. All quantitative indicators can be combined numerically into a single or a few scores through the use of weighting factors or rankings based on a weighting algorithm. However, these methods depend on normative perception. Consequently, various methods, such as expert panels or surveys, are available to give weighting factors based on normative judgement, which is required as an input for the aggregation step. However, the problem is that neither of these approaches is totally validated as a standardized method of assigning the weights to the indicators, which would be necessary for decision makers to implement the resulting recommendations. Trade-off scenarios are also not readily apparent, and decisions in such scenarios, which depend on weighting factors, are difficult to understand for decision-makers not involved in the study. This has led to the need for a final integration stage for the different components in LCSA [190], and further research needs to explore the use of complementary tools such as MCDA to aid combining, balancing, and potentially weighting life cycle sustainability data. That having been said, the majority of decision cases do not require absolute judgements, and can be adequately supported by a single score and a differentiated assessment [193]. Therefore, there is much scope for LCSA to add real value to decision-making in qualitative as well as quantitative contexts.

Finally, there is the case of the validation of many integration approaches, either by analyzing the results separately or by aggregation, as seen in studies such as those of Nubi et al. [180], where the presentation of sustainability for the WtE systems and the evaluated contexts involved a ‘twofold’ approach. For the first approach, the results evaluations for the respective LCA, LCC, and sLCA tools were conducted in a separate way, and thus, these dimensions of sustainability were considered independently of each other. In the second approach, an integrated sustainability assessment perspective was adopted by aggregating the findings of the three life cycle assessments into a complete LCSA context, involving the derivation of single life cycle sustainability scores. While there may be value in taking a flexible approach to integration, as each approach has its own merits and issues, there is nonetheless a need for a single standardized methodology for integrated sustainability assessment which would facilitate the comparison of findings [190], and opportunities exist for more studies to carry on with the development of LCSA approaches, particularly concerning the standardization of a general methodology for LCSA.

Despite the challenges noted above, the use of LCSA to evaluate the sustainability of WtE in developing countries should be recommended. This is because LCSA is a decision support tool that provides a review of the prospective sustainability performance of electricity generation from MSW and indicates areas of either significant negative impacts where improvements can be made, or positive impacts where opportunities can be utilized [188]. LCSA processes and outcomes are capable of accommodating trade-offs that occur when seeking optimal sustainability outcomes for systems such as WtE and which depend on the various priorities of key stakeholders and the environmental, social, and economic attributes of diverse technology options.

Each WtE technology has its benefits and detriments regarding sustainability, and one specific technology usually cannot deal with all of the waste. Thus, an integrated system combining different WtE and waste management technologies (recycling and composting) is expected to improve the overall energy and material recovery efficiency, particularly for waste with a high level of complexity. However, to date, published reports on the sustainability evaluation of such integrated systems in a developing world context are rare. Hence, future studies should be conducted to capitalize on the holistic nature of LCSA and its component individual domains: LCA, LCC, and sLCA [188].

In terms of policy implication and implementation, it is important that governments of developing countries avoid the problem of resolving one issue (energy security or waste management) at the expense of another (environmental or social impact, etc.) when planning sustainable strategies for the electricity or the waste sector. This will assist in making more sustainable decisions for the future [194]. Additionally, governments should use a life cycle approach in decision- and policy-making. This will help in identifying hot spots and opportunities for reducing the environmental, economic, and social impacts across the entire electricity supply chain. Governments should also support research into the environmental improvements and impacts of WtE technologies, as well as strengthen laws to limit environmental impacts from electricity production [194]. It is very important that policy- and decision-makers resolve and effect the implementation of WtE policies aimed to overcome challenges from financial, institutional, and technological perspectives [194].

For the effective optimization of WtE projects, a combination of two or more technological options could prove promising, as this would in theory guarantee maximum usage of feedstock, resulting in an optimized energetic yield and greater environmental benefits compared to applying the technologies separately. For example, integrating AD, which produces biogas from food waste, and gasification, which produces syngas from wood waste, agricultural residues, and recyclables such as paper and plastics, will ensure optimum usage of the waste fractions in the MSW stream. Research on the hybrid application of WtE technologies includes those of [195], involving the economic assessment of a hybrid of anaerobic digestion and gasification, where the findings revealed that integration of gasification and AD performed better economically compared to a separate system. Alao et al. [196] performed a study on the hybrid application of AD, LFGTE, and

pyrolysis in the city of Lagos, Nigeria and revealed that this hybrid application could reduce greenhouse gas emissions by 91.16%. Given that the hybrid application of WtE technologies tends to be more sustainable than standalone applications, further research is needed regarding the integrated application of waste-to-energy technologies.

Hence, the following could potentially serve as additional research focuses required to achieve this:

1. Further research work on the techno-economic, environmental, and social implications of a hybrid implementation of WtE systems is required.
2. More research work should be aimed towards the comprehensive study of new technologies, such as torrefaction, plasma arc gasification, fermentation (bio-ethanol production), bio-hydrogen production, use of microbial fuel cells, and esterification.
3. Given the different performance of WtE technologies from a technical, economic, environmental, and social perspective based on qualitative and quantitative standards, the adoption of multi-criteria-based approaches that can simultaneously consider qualitative and quantitative criteria should be a platform for future work.

Finally, the maximization of the potential of the MSW generated daily in developing nations can only be actualized through collaborative efforts, such as technology transfer and knowledge sharing among researchers and academics, equipment manufacturers, and all others concerned in developed and developing nations, which could assist developing countries in actualizing the implementation of various WtE projects.

## 6. Conclusions and Further Research

The analysis of the literature set out in this paper indicates that the LCSA approach can be used in revealing the life cycle impacts and sustainability of WtE systems in an integrated way across the boundaries of the three traditional domains of sustainability. This makes LCSA a potentially useful emerging decision support tool that can provide valuable insights into the level of sustainability of present and future waste management and electricity generation systems used in developing countries.

It also highlights domains of either negative impact that need to be improved, or those of positive impact where opportunities can be utilized. Ongoing work is required to develop LCSA, particularly to address some of the challenges outlined in this paper.

The following are suggested as priority areas for future research:

1. Continuing research and methodological development to ensure consistency, equity, and balance among the environmental, economic, and social domains of LCSA.
2. Further enhancement of the data and stakeholder engagement processes to develop more regionally specific, recent, and relevant economic and social data. This should include further research on approaches to identify, prioritize, and calibrate social indicators in the sLCA component of LCSA.
3. Methods to represent and calibrate the uncertainties in LCSA and its components should be further improved [68].
4. Additional research on the combination and integration of LCSA results using multi-criteria decision-making frameworks and/or optimization models.

In addition, this review has shown that the potential use of sustainability indicators can provide valuable and far-reaching insights into the benefits and detriments of different technological options, both present and future. This requires further research into the potential of having an optimized WtE system incorporating other technologies, such as recycling, that could be environmentally beneficial, economically viable, and socially acceptable. This could potentially contribute to development of waste management and the alleviation of inadequate electricity supply in developing countries. This study has also provided an opportunity to explore several ways to develop the use of LCSA as a tool for analysis, particularly with regards to the representation of integrated sustainability results to enhance their better understanding and interpretation. This will invariably aid in making better informed decisions on various strategies or processes to be undertaken by decision-

and policy-makers. Given this, it is anticipated that the use of the LCSA methodology and its findings can make a substantial contribution to environmental and energy policy decisions in developing countries to solve the dual problem of waste management and electricity supply.

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