doi: 10.2166/9781789063479_00293

Chapter 11

Life-cycle assessment, carbon-footprint and techno-economic analysis

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ABSTRACT

Anaerobic treatment of domestic wastewater has been recognized as a feasible technology to reduce energy demand from the valorization of streams with high-organic load. However, there are many factors that influence the environmental and economic profiles of the process, such as the effluent pollutant load, operating temperature and methane emissions in the treated effluent and in the air. Therefore, all these parameters must be optimized taking into account not only the technical performance (i.e. biogas production), but also environmental and economic implications. This chapter presents a dual approach to highlight the characteristics of anaerobic treatment of domestic wastewater with life-cycle assessment and techno-economic analysis and how to approach these methodologies in diverse case studies.

Keywords: domestic anaerobic treatment, life-cycle assessment (LCA), techno-economic analysis (TEA), wastewater treatment (WWT).

11.1 SUSTAINABILITY AND CIRCULARITY IN WASTEWATER TREATMENT PLANTS

The 2030 Agenda for Sustainable Development sets out a series of ambitious objectives for the global community. These sustainable development goals include targets for access to safe water and sanitation and improved water management. To date, these are challenges that have proven difficult to meet, partly not only because they are complex, but also due to the global context, with political, social, economic uncertainty and environmental adversities. This calls for redoubled efforts and carefully selected approaches to achieve transformational change. Especially, drinking water scarcity, climate change and resource depletion are driving a paradigm shift in the wastewater treatment sector, reinforced by social awareness and new legislative changes. Because of this, there has been a more proactive attitude regarding the transformation of public perception of wastewater treatment plants from disposal to waste valorization facilities or 'biorefineries'. In Europe, the goal is not only compliance with the already outdated Directive 91/271/EEC, but also try to incorporate the precepts supported by the Circular Economy Action Plan published in 2020, the Regulation (EU) 2020/852 (also known as 'EU Taxonomy') in relation to sustainable development, the Regulation (EU) 2020/741 for minimum quality in water reuse and with the forthcoming wastewater treatment directive (revised proposal in October 2022). Quality thresholds for wastewater reuse have also been contemplated by non-European countries and worldwide organizations, such as the World Health Organization, Environmental Protection Agency (EPA), Australian Guideline for Water Recycling (AGWR) and International Organization for Standardization (ISO). In terms of the implementation of circular economy, other initiatives, platforms and organizations around the world can also be cited, such as African Circular economy Network, Economy and Social Commission for Asian and the Pacific, Asia Circular Economy Association and the Platform for Accelerating the Circular Economy.

In this context of sustainability and circularity, the strategic direction of the sector is moving towards the design of energy-neutral facilities, the application of more restrictive criteria for the removal of micropollutants, the improvement of reclaimed water quality and the improvement of sanitation in areas that had been left behind. Better access to water and sanitation, water management and governance and the multiple benefits they bring, can contribute significantly to positive transformation in these environments.

Because of these, strategies and technologies have been put forward to reduce, reclaim, reuse, recycle, recover and rethink and thus maintain material flows within the production–consumption chain. The most prominent resource of a wastewater treatment plant (WWTP) is wastewater, which could be transformed into a multitude of co-products such as liquid or solid fertilizers from digestate or as struvite, hydroxyapatite, k-struvite or ammonium sulphate, polymers (polyhydroxyalkanoates (PHAs), celluloses and polyhydroxybutyrates (PHBs)), biomass (microalgae and biochar) and energy (hydrogen, biomethane, biodiesel, bioethanol and biogas) and also reclaimed water to be reused (Singh *et al.*, 2022). Among them, one of the most recurrent products is biogas given the large technological maturity of anaerobic digestion (AD) processes and its well-known function as storable energy vector. Although AD has been typically supporting sewage sludge management in developed countries because the continuous challenges in the search for politically and economically acceptable treatment routes, its applicability goes beyond, as algal biomass, food waste, sewage, industrial feedstocks and other high-organic strength streams have been widely used in the co-digestion of organic waste (European Environment Agency, 2022). Anaerobic treatment offers an opportunity for the transformation of organic matter present in industrial and domestic wastewaters and reduced sludge production compared to a conventional activated sludge (CAS) system. Apart from this, anaerobic treatment improves the profile of facilities in relation to energy self-sufficiency by converting the produced biogas into heat and/or electricity (Stazi & Tomei, 2018). The system also responds more quickly to the addition of substrates after shutdown periods. In line with the above benefits, it entails a reduction in reactor size, a feature associated with its higher loading rate (Zieliński *et al.*, 2023). However, these are only some of the reported advantages, as there are also environmental (avoidance of carbon dioxide emissions) and economic (sale of energy) benefits to consider. To obtain a broader view of the performance of this process across these two pillars of sustainability, this chapter presents both a critical review of the literature to highlight the potential environmental and economic benefits of anaerobic domestic treatment and a practical case study. Given the limited publication rate of research articles in the fields of life-cycle assessment (LCA) and techno-economic analysis for anaerobic secondary treatment, prospective challenges for technologies were also analysed by addressing a broad systematic revision of the literature on the topic of anaerobic treatment of domestic wastewater.

11.2 ANAEROBIC TREATMENT FROM AN ENVIRONMENTAL PERSPECTIVE

Anaerobic treatment is a very versatile process whose application has been widely studied in the wastewater treatment sector, especially for the management of sludge. However, depending on the operating conditions the required energy can totally or partially offset the energy balance of the facility. Sustainable sewage treatment plants (STPs) operated with anaerobic processes must, then, rely on technologies capable of maintaining a positive balance between the desired quality of emissions and the recovered resources. Therefore, the design and operation of STPs must be framed within a context in which environmental impacts are comprehensively and continuously assessed. One of the most recognized methodologies for this purpose is LCA, which not only takes into account direct–indirect emissions, but it is also useful for the identification of internal process weaknesses and technological benchmarking through its life cycle.

It consists of four well-defined stages: definition of the scope and system boundaries, development of the life-cycle inventory (LCI), life-cycle impact assessment (LCIA) and interpretation of the results (ISO, 2006a, 2006b). The first stage comprises the formulation of the initial hypothesis, the level of detail of the system under study, the data search needs, the definition of system boundaries and functional unit (FU). The so-called 'system boundaries' include all elements of technology, process or product under assessment. On the contrary, the FU is a quantitative definition of the system, as it is used as a reference for inputs and outputs and allows comparison of processes (ISO, 2006a). The second stage or LCI involves the quantification of the quantities of materials and energy, emissions and waste calculated with reference to the previously selected FU. All the data collected is then used and translated into environmental burdens during the impact assessment (third stage), which classifies emissions into their respective impact categories. These categories are global indicators (e.g. climate change, eutrophication or resource depletion), which provide insight into the status of an environmental aspect. Finally, in the last stage, the results of the LCI and the LCIA are interpreted, and conclusions of the study are provided.

In this sense, the first question to be answered is whether the anaerobic treatment constitutes a good technological alternative compatible with a good quality state of ecosystems, wildlife and humans when installed in domestic wastewater treatment plants. As mentioned in Section 11.1, the technology can be implemented both in water and sludge lines. In the first case, the most powerful strategy is the direct reduction of energy demand by replacing typical aeration systems with mixing devices. It is worth noting that aeration electricity accounts for 45–75% of the overall cost of wastewater treatment plants (Kong *et al*., 2021). In the second case, the use of anaerobic treatment in the sludge line allows higher methane yields as a source of bioenergy while allowing the concentration of phosphorus or nitrogen in a solid digestate that can be used in agriculture. In terms of environmental performance, anaerobic treatment, if all the produced methane is recovered, has the potential to reduce greenhouse gas (GHG) emissions. The best performance of an anaerobic process is directly related to the concentration of wastewater. Considering only direct emissions, a conventional aerated sludge process is only outperformed when the influent has a biochemical oxygen demand (BOD) concentration above 500 mg/L. This is related to emissions to the environment of dissolved methane contained in the anaerobically treated effluents, which can be as high as the methane recovered (Cakir & Stenstrom, 2005). Methane is a strong GHG with a global warming potential (GWP) of 28 (IPCC, 2021). Thus, GHG emissions associated with the anaerobic treatment of diluted sewage could be higher that the aerobic counterpart. Problems associated with the presence of dissolved methane in the anaerobically treated effluents were discussed in Chapter 7. For indirect emissions, the selection of post-treatment technologies, reactor configuration and wastewater salinity play key roles. The environmental profile is expected to be positive for anaerobic systems, at least for medium-strength influents (Smith *et al*., 2014).

Concerning the environmental benefits of anaerobic treatment for sewage sludge three publications can be highlighted claiming its best environmental performance. Blanco *et al*. (2016) and Arias *et al*. (2020a) tried to quantify it with an LCA focused on the introduction of a digester within the sludge line of a domestic WWTP. The implementation of this technology led to a noteworthy impact reduction in the environmental profile of the facility of ~85 and 10% for each of the previously named studies, respectively. Awad *et al*. (2019) also supported the conclusions found in both publications, although the benefits achieved for the climate change category were much smaller (0.3% difference). It should be noted that, when an AD process is implemented, the dewatering characteristics of the sludge improve and, therefore, a lower amount of chemicals (polyelectrolytes) is required in the following dewatering stages. In addition, the energy received from the biogas transformation provides the plant with a certain degree of autonomy from the national grid. However, and as it was underlined for domestic wastewater, the results achieved would depend on the characteristics of influent and technologies.

11.2.1 Is technology selection affecting the environmental profile?

The environmental benchmarking between anaerobic treatment technologies and other technologies cannot yet be offered in its entirety. The reason is the difficulties encountered in the following two aspects: the implementation of the technology within wastewater treatment plants and the lack of comparability between LCA studies, as they have different objectives, scopes, FUs, system boundaries and so on. Due to different methodological decisions made in each publication related to the topic, a coherent comparison between technologies cannot be made among research studies published by different authors. Accordingly, the comparison is restricted to scenarios proposed within the same published manuscript. In this sense, currently the maximum number of scenarios evaluated in the same manuscript is around five–six, although it is usual to study three–four scenarios in a comparative way. In addition, the diversity of studies is greater when anaerobic treatment is applied for sludge valorization compared to when it is referring to the water line. Consequently, research for domestic treatment has been directed towards the comparison of biological technologies, whereas the ongoing study of the scientific community for sewage sludge has resulted in some interesting findings for incineration, PHA production, hydrothermal carbonization (HTC) systems, pyrolysis and AD with microalgae processes for side-streams and lagoon biodigesters.

Chronologically, the first research addressing the aerobic versus anaerobic debate as secondary treatment from an LCA perspective was published by Smith *et al.* (2014) trying to compare an anaerobic membrane bioreactor (AnMBR) with a -rate activated sludge (HRAS) system. Other technologies were evaluated, not only in terms of technological comparison, but also taking into account different domestic wastewater composition, temperature and sludge-handling practices. An AnMBR produced the highest environmental impact in all categories analysed for medium-strength wastewater (430 mg/L chemical oxygen demand (COD)) and was related to two aspects: high-energy requirements for membrane scouring by biogas sparging and dissolved methane emissions. At higher strength of wastewater (800 mg/L COD), an AnMBR was able to outperform a CAS system due to higher energy production. Two years later, Pretel *et al.* (2016) also share insights on the environmental performance of AnMBRs. Due to the problematic encountered with respect to diffuse methane emissions in the previous study, they proposed an improvement of the system with a degassing unit in order to capture the methane dissolved in effluents for additional energy production or to be used as organic matter for denitrification in the downstream post-treatment unit. As a result, they obtained the best profile for AnMBRs for moderate–high-strength domestic wastewater in all categories analysed except for eutrophication. However, many more parameters affect the environmental profile. Pretel *et al.* (2013) attempted to provide insight into energy demand, biogas recovery, nutrient recovery and sludge removal factors for an AnMBR. Although it is true that higher operating temperatures resulted in higher methane production, this energy was not sufficient to tackle the heating requirements to operate. To address the environmentally needed improvements for AnMBR technologies in domestic wastewater treatment, Harclerode *et al.* (2020) evaluated eight new scenarios. They considered primary treatment, membrane fouling, sulphide removal, phosphorus removal, dissolved methane management and waste management. Oxidative biological sulphide removal was shown to be more sustainable than coagulation, vacuum degassing tanks were shown to be better than hollow fibre contactors for methane removal and primary sedimentation was shown to be better than simple fine screening. Among these, sulphide and phosphorus removal offered the greatest improvements, with reductions of up to 70% in all impact categories.

The study of Sills *et al.* (2016) was no longer related to AnMBRs. Their environmental assessment was rather carried out for an anaerobic baffled reactor (ABR), for a combination of a tricking filter with ABR, tricking filter and ABR with constructed wetland (CW) and for an ABR with CW. Of the four scenarios, the ABR followed by tricking filter resulted in the best profile. However, the benefits of bioelectricity production in the climate change category for ABRs were again hampered by dissolved methane in anaerobic effluents. This is because ~95% of GWP was affected by this specific direct emission. It appears that, regardless of the technology used, the environmental competitiveness of anaerobically operated technologies as secondary treatment is related to the degassing of methane in effluents and the concentration of nitrogenous compounds. Upflow anaerobic sludge blanket (UASB) with wetland post-treatment and CAS were also compared in a study by Laitinen *et al.* (2017), who found a strong relationship between energy demand and climate change. The clear winner of their analysis was the CW-based treatment. UASB + CW was also the core topic of the study by de Sampaio Lopes *et al.* (2014) but the interpretation of the results did not include a comparative assessment of technologies. However, the contribution analysis provided pointed out the importance of sodium hypochlorite solution control during the disinfection stage. There are no longer restrictions for the joint application of UASB with CWs as a post-treatment step, Patel and Singh (2022) carried out a comparative LCA of the stand-alone performance of a UASB reactor with activated sludge processes, sequential batch reactors and CWs separately. Overall, UASBs show a better profile in the categories analysed compared to CAS and SBR (sequencing batch reactor) processes, except in eutrophication. This is due to the negligible nutrient removal from the anaerobic treatment, whereas SBR, despite being the highest contributor in the other categories, shows the lowest eutrophication potential (Table 11.1).

Considering the AD of domestic sewage sludge, two groups of technologies were studied: non-biological and biological-based. Within the first family, Chen *et al.* (2022) attempted to provide a comparative view for sewage sludge management of direct incineration and a combination of AD with incineration, whereas Medina-Martos *et al.* (2020) focused on the integration of the above-mentioned technology with a newly developed HTC system. The combination of HTC and AD resulted in a reduction of global warming potential by 75%. The improved performance compared to stand-alone AD was due to the recovery of hydrochar for heat production, which has a renewable biogenic origin. Studies by Li and Feng (2018) and Li *et al.* (2017a) are other examples on pyrolysis and co-incineration (Table 11.2).

For biological systems, AD was combined with a microalgae treatment. Benefits can be obtained from exhaust gas treatment with microalgae and dewatering of digestate from primary and secondary sludge. Because co-digestion would result in increased biogas and reduced energy for aeration in a secondary treatment reactor (the algae pond acts as a treatment process for ammonia emissions and prevents its return to the water line), implementation of the technology improves the environmental profile in relation to climate change, ozone layer depletion, freshwater eutrophication and water consumption (Tua *et al.*, 2021). However, microalgae production does not necessarily need to occur in the sludge line and advantages can also be taken from the substitution of the CAS process.

It is noteworthy that the energy sector seems to benefit from technological advances in AD. A future optimized perspective of a two-stage system including stages of a dark fermentation process and PHA accumulation can reduce the environmental impact associated with climate change by up to 41%. However, AD as a single solution remains as of today the most advantageous alternative for wastewater valorization, as three gaps remain to be filled in two-stage systems: the microbial growth demands an increase, the need of higher accumulation yields for polyhydroxybutyrate production and larger organic loading rates or amount of feedstock processed inside the reactor (Asunis *et al.*, 2021).

11.2.2 Prospective environmental–technological challenges

One of the recurrent applications of LCA is the identification of critical points and a comparison among products/scenarios performing the same function, especially in large-scale operation, and also in the early-design phases when it is possible to identify those stages with the greatest potential impact and on time implementation of changes that can result in a significant environmental performance improvement. Therefore, this section presents a critical review of the environmental consequences of different variables and technologies related to anaerobic treatment. So far, published studies have discussed the concentration of volatile solids, the impact of upstream and downstream processes, the temperature and co-digestion with other organic waste streams.

Table 11.1 Methods and FUs of anaerobic domestic wastewater treatment processes for domestic wastewater and related.

Type of Resource	LCA Method	FU	References
Domestic wastewater, blackwater, urine and greywater	ReCiPe	26,000 inhabitants and 6510 jobs/year	Lehtoranta *et al.* (2022)
Domestic wastewater	IMPACT 2002+	1 m^3 of treated wastewater and 1 kg of sludge	Patel and Singh (2022)
Domestic wastewater	ReCiPe	1 m^3 of water available for consumption or 1 m^3 of wastewater	Boldrin *et al.* (2022)
Domestic wastewater	TRACI	5 million gallons of wastewater	Harclerode *et al.* (2020)
Blackwater	ReCiPe	1 m^3/day of wastewater treated	Estévez *et al.* (2022a)
Blackwater	ReCiPe	1 m^2 of urban garden	Estévez *et al.* (2022b)
Blackwater and kitchen waste	ReCiPe	Blackwater (in m^3) and kitchen waste (in kg) generated in a four-person household/year	Prado *et al.* (2020)
Domestic wastewater and sewage sludge	CML	1 t of TS	Cañote *et al.* (2021)
Domestic wastewater and food waste	TRACI	5 million gallons/day	Becker *et al.* (2017)
Domestic wastewater and food waste	ReCiPe	2000 inhabitants/day	Lijó *et al.* (2017)
Domestic wastewater and sewage sludge	IPCC	1000 m^3 of influent wastewater	Laitinen *et al.* (2017)
Domestic wastewater	TRACI	1 m^3 wastewater	Shoener *et al.* (2016)
Domestic wastewater and sewage sludge	IMPACT 2002+	2 MGD of domestic wastewater, assuming a plant lifetime of 30 years	Sills *et al.* (2016)
Domestic wastewater	CML	1 m^3 of treated wastewater	Pretel *et al.* (2016)
Domestic wastewater	TRACI	5 million gallons/day of wastewater	Smith *et al.* (2014)
Domestic wastewater	CML	1 m^3 of treated effluent	de Sampaio Lopes *et al.* (2014)
Domestic wastewater	CML	1 m^3 of treated wastewater	Pretel *et al.* (2013)

11.2.2.1 Solid concentration

There is direct relationship among feedstock concentration, the conversion efficiency of the organic matter and environmental impact. However, the conclusions drawn by several authors regarding to the best techno-environmental performance do not seem to go in the same direction. The reason may be related to LCA methodological approaches adopted by each author, but also to the use of feedstocks with different solid concentrations (despite considering similar VS/TS (volatile solids/total solids) ratios in the studies) and technological configurations. Research conducted in the scientific community on this topic has been applied mainly to the most concentrated streams, sludge. This is because domestic wastewater usually contains a solid concentration of <0.1% and, therefore, the benefits of stream concentration are not sufficiently relevant unless the wastewater is mixed with food waste (Becker *et al.*, 2017; Lijó *et al.*, 2017). Recent LCA studies have demonstrated that the environmental feasibility of anaerobic treatment depends on both the biodegradability of organic matter and the concentration of solids (Chen *et al.*, 2022; Li & Feng, 2018; Li *et al.*, 2017a). Although all of them mainly focused on sludge treatment, lessons can be learnt for domestic wastewater treatment. For example, the environmental profile would improve with increasing volatile solid concentration

Table 11.2 Methods and FUs of anaerobic domestic wastewater treatment processes for sewage sludge and related.

Type of Resource	LCA Method	FU	References
Sewage sludge	ReCiPe	1 m^3 of biogas	Singh *et al.* (2022)
Sewage sludge	IPCC	500 t of raw sewage sludge (5% TS)	Chen *et al.* (2022)
Sewage sludge and food waste	CML-IA baseline	1 kg of digested sludge	Satayavibul and Ratanatamskul (2021)
Sewage sludge with algae biomass	ILCD, CED, WC and LO	1000 m^3 of influent wastewater	Tua *et al.* (2021)
Sewage sludge	ReCiPe and CML	1 t of mixed sludge	Arias *et al.* (2021)
Sewage sludge	Environmental footprint	1 m^3 of urban wastewater	Brockmann *et al.* (2021)
Sewage sludge	ReCiPe	1 m^3 of treated wastewater and 1 m^3 of produced methane	Lanko *et al.* (2020)
Sewage sludge	Usetox	1000 kg wet mixed sludge	Medina-Martos *et al.* (2020)
Sewage sludge and food waste	Not specified	1 m^3 wastewater	Morelli *et al.* (2020)
Sewage sludge, cow manure, forage waste and returned dairy products	CML	1 t of wet manure	Adghim *et al.* (2020)
Sewage sludge digestate	ILCD/PEF	1 kg of biopolymer	Vogli *et al.* (2020)
Sewage sludge	eFootprint	1000 kg of dry biomass	Wang *et al.* (2020)
Sewage sludge	CML and ReCiPe	1 m^3 of treated wastewater	Arias *et al.* (2020b)
Sewage sludge	CML and ReCiPe	1 t of mixed sludge	Arias *et al.* (2020a)
Sewage sludge	CML	1 m^3 of treated wastewater	Awad *et al.* (2019)
Sewage sludge	Not specified	1 t of mixed sludge	Cartes *et al.* (2018)
Sewage sludge	CML	1 t TS of thickened sludge	Li and Feng (2018)
Sewage sludge	CML	1000 kg of mixed sludge	Yoshida *et al.* (2018)
Sewage sludge	CML	1 m^3 of raw wastewater	Colzi Lopes *et al.* (2018)
Sewage sludge, septage and high-strength organic waste	ReCiPe and TRACI	1 MGD	Morelli *et al.* (2018)
Sewage sludge	ReCiPe	1 t of TS of sludge entering the sewage sludge treatment line	Gourdet *et al.* (2017)
Sewage sludge	Not specified	1 dry tonne of sludge	Heimersson *et al.* (2017)

(Continued)

Table 11.2 Methods and FUs of anaerobic domestic wastewater treatment processes for sewage sludge and related (*Continued*).

Type of Resource	LCA Method	FU	References
Sewage sludge and food waste	CML	Annual management of municipal waste and sewage	Edwards *et al.* (2017)
Sewage sludge	CML	1 t of TS	Li *et al.* (2017b)
Sewage sludge	CML	1 t of TS	Li *et al.* (2017a)
Sewage sludge	Not specified	1 dry tonne of biosolids	Alvarez-Gaitan *et al.* (2016)
Sewage sludge	ReCiPe, EDIP and CML	1 m^3 of influent wastewater	Piao *et al.* (2016)
Sewage sludge	Not specified	1 day for a plant with capacity of 70,000 inhabitants	Tomei *et al.* (2016a)
Sewage sludge	ReCiPe	1 person equivalent as 60 g of BOD_5/day	Blanco *et al.* (2016)
Sewage sludge	Not specified	Daily inflow to the WWTP	Gianico *et al.* (2015)
Sewage sludge	CM	1 t of TS	Mills *et al.* (2014)
Sewage sludge	ReCiPe and Usetox	10,000 m^3 wastewater per day	Heimersson *et al.* (2014)
Sewage sludge supernatant	CML	1 kg PO_4^{3-} eq. removed	Rodriguez-Garcia *et al.* (2014)
Sewage sludge	IPCC	500 m^3 liquid raw sewage sludge	Cao and Pawłowski (2013)
Sewage sludge	Not specified	Amount of sludge generated/year	Remy *et al.* (2013)
Sewage sludge and food waste	CML	1000 t of OFMSW (Organic Fraction of Municipal Solid Waste) and 2000 t of sewage sludge (dry matter (DM) 17%)	Righi *et al.* (2013)

regardless of reactor configuration. However, the results (methane production) might be negligible and not be able to offset the environmental impacts of other inputs that cross system boundaries.

This is the case of Li *et al.* (2017a), which has obtained a very small (1.6–7.1% among scenarios) modification in the environmental profile when the organic content of the sludge varied from 70 to 40%. Other publications can be highlighted: Chen *et al.* (2022) demonstrated for sludge that incineration is a good option instead of AD for recoverable biodegradable waste with a low-organic content (VS/TS <55%), and Li and Feng (2018) showed that pyrolysis improve as much as its organic matter content does (the results were 4.6 times higher when achieving 70% compared to the baseline value of 50%). Finally, Li *et al.* (2017b) analysed processes with operation at low (3–6%) and high (10–15%) total solid concentrations with the same organic matter fraction (i.e. VS/TS of 70%). In the latter case, environmental benefits are obtained for high-solid concentration technologies at thermophilic temperature and for low-solid concentration, at mesophilic temperature.

11.2.2.2 Upstream and downstream processes

The system boundaries of an LCA should be defined in accordance with the goals of the study. Therefore, the analysis of the environmental impacts of anaerobic treatment may be complemented by other processes or technologies used to facilitate organic matter solubilization, enhanced nutrient recovery or removal or compliance with legislation. The analysis of wider system boundaries has been mainly applied for the sewage line, but two studies have been reported for anaerobic secondary treatment. The first one is the research of Harclerode *et al.* (2020) who studied the differences between the use of primary sedimentation and screening only, included subsequent treatment with dissolved methane removal and nutrient removal and sludge management with AD and lime stabilization. The other one is the study of Laitinen *et al.* (2017) that incorporated a CW as post-treatment for a UASB reactor. Indeed, the substitution of secondary treatment technologies from aerobic to anaerobic can lead to the creation of new technological challenges within the sludge line, which indirectly contribute to the overall environmental sustainability of the system. This technological replacement has been analysed through several studies. For example, Brockmann *et al.* (2021) compared the performance of a CAS system with oxygen photogranulation and Arias *et al.* (2020a) provided information on the benefits of HRASs followed by an integrated film-activated sludge. In anaerobic treatment technologies, the change in environmental impact could be due to both a decrease in excess sludge produced and a change in its quality. This would influence not only the size of the equipment, but also the selection of the most appropriate technologies for its management. Within this topic, the study by Arias *et al.* (2021) is noteworthy as they have proposed a comparison between a chemical and thermal pre-treatment for sewage sludge management after the subsequent AD. Although the absence of these technologies implies a lower energy demand, a worse environmental profile in terms of climate change was found. The best performance in this category corresponded to the chemical approach due to higher biogas production and lower energy demand. However, a higher impact was recorded in other categories such as terrestrial acidification, particulate formation and terrestrial eco-toxicity due to indirect emissions generated for the production of these chemicals. Wang *et al.* (2020) also compared stand-alone AD with the AD process with heat treatment and a new alkali/acid pre-treatment. The latter treatment shows less impact on climate change, although other categories such as acidification and eutrophication potentials favoured the implementation of the thermal technology. Despite this, the authors claim that alkaline treatment is comparable to thermal treatment and that the small differences are due to sodium chloride consumption and higher energy demand.

11.2.2.3 Temperature

Temperature is another parameter that must be considered within the process because there is a strong relationship with the reaction yield and with the energy requirements. When it comes to the valorization of wastewater for energy purposes, the energy consumed must be compensated by that resulting from the manufacture of the co-products. In domestic anaerobic treatment, the only product analysed so far with LCAs is the biogas and, thus, only an energy perspective has been considered in the literature. The temperature was addressed for membrane bioreactors (MBRs) at 15, 20 and 33°C in Smith *et al.* (2014) and Pretel *et al.* (2013). However, for temperatures up to 33°C, energy production from urban wastewater is not able to compensate for the heating needs of the system. For this reason, it is necessary to optimize the biogas yield, but only if the balance between production and consumption is positive. Studies on anaerobic sludge digestion processes (more data available) have shown that higher temperatures do not always lead to better environmental performance, and, in many situations, such improvement is only possible for some impact categories and FUs.

In this regard, Lanko *et al.* (2020) demonstrated the effects of temperature for sewage management through the analysis of three options: mesophilic, thermophilic and temperature-phased AD. The latter alternative showed the best environmental performance of the three in all the categories analysed (i.e. toxicity, land occupation, eutrophication and fossil resource depletion, among others) but climate change on which mesophilic digestion slightly outperforms. However, the authors found inconsistencies among different FUs. By changing the FU from 1 m^3 of the treated wastewater to 1 m^3 of the produced methane, the previously stated environmental results were rearranged and led to a better profile of the thermophilic reactor for all categories except for climate change where mesophilic operation is still more favourable. With a similar approach, Li *et al.* (2017b) presented the environmental differences among five processes (conventional AD, high-solid AD, AD with thermal pre-treatment hydrolysis, thermophilic AD and thermophilic high-solid AD). Regardless of the solid concentration (suitable for both high- and low-solid contents), the best environmental results were obtained with high-solid thermophilic AD (44% improvement for a VSS (volatile suspended solids) concentration of 70%). It is true that this technology requires more energy for heating the feed sludge, agitation and transport, but its efficiency is also higher and therefore more biogas is recovered. It seems that, for the same solid concentration, thermophilic technologies provide the best profile.

On the contrary, domestic anaerobic wastewater treatment still lacks LCA studies referring to the production of non-energy bioproducts. In contrast, the sewage sludge digestion may produce VFAs (volatile fatty acids) as a result. However, the production of non-energy products reduces the energy self-sustainability of facilities because they cannot be converted in electricity or heat to be recycled back to systems. Therefore, the efficiency of heating devices, energy loss prevention (adiabatic systems are preferable) and energy-environmental optimization become essential. For example, Elginoz *et al.* (2020) reported a 10% decrease in environmental load by improving the efficiency of the heating device by 10%. Furthermore, in many impact categories (i.e. terrestrial acidification, eutrophication, eco-toxicity and global warming, among others) temperature reduction showed a better profile (between 13 and 36%) despite the lower performance associated with a temperature decrease.

11.2.2.4 Co-digestion of streams

Temperature and organic matter content are commonly parameters in the evaluation of anaerobic technologies, as indicated in sections 11.2.2.1 and 11.2.2.3. In this sense, co-digestion of other biodegradable waste streams with domestic wastewater would allow process optimization as higher volatile solid concentrations, C/N ratios and removal efficiencies can be achieved. Within this context, Becker *et al.* (2017) have demonstrated decreased impact on GWP with increased treatment of food waste with domestic wastewater. They have evaluated three technologies: a conventional (CAS), an anaerobic (AnMBR) and a newly developed aerobic treatment (HRAS). Their study could not draw conclusive results, as the environmental profiles of each were very similar and the uncertainty analysis showed small confidence intervals (probability within which the results of the environmental profile can be found). However, they highlight the potential of AnMBRs due to the recent development of the technology and the comparability of the net energy balance with HRAS-AD (technology that Arias *et al.* (2020a) stated as environmentally friendly compared to CAS for municipal wastewater).

Although not for domestic wastewater, other publications can be cited for sewage sludge. Morelli *et al.* (2020) have reported that there was a clear reduction (between 46 and 108% depending on the impact category) of the environmental impact with the implementation of co-digestion in wastewater treatment plants. Edwards *et al.* (2017) compared the same co-digestion strategy with a segregated treatment where food waste ended up in landfill. As a result, the co-digestion scenario outperformed (given the additional bioenergy generated) the segregated treatment in climate change potential (represented as 53–71% of the impact of the business-as-usual or landfill scenario for food waste).

11.3 TECHNO-ECONOMIC VIABILITY OF THE AD

Techno-economic assessment (TEA), cost–benefit analysis (CBA), life-cycle costing (LCC), cost–effectiveness analysis, cost–utility analysis and cost method of accounting are some examples of methodologies for the analysis of economic aspects of products, technologies and/or processes. Among them, TEA, CBA and LCC are some of the most frequently mentioned methodologies in the scientific community. Although LCC is the oldest methodology, they have been recently developed in parallel leaving to a vague distinction among them. CBA mainly focuses on projects and policies with the monetization of costs and benefits and aims at financial profitability. Although it is advantageously an autonomous tool unveiling whether a selected system under study has attributable welfare, CBA is considered a 'black-box' methodology. It simply ignores what happens inside the system in order to confirm its economic viability. Because of this problematic, TEA emerged as a tool simultaneously integrating the implications of technological aspects in the economy of a process or project as it systematically examines the interrelationships among them. There is, thus, a combination of engineering decision taking aiming at process optimization with the economic changes associated with them. In this regard, TEA separates from the vision provided for CBA as is not seem only a tool for investment. Therefore, TEA analysis should always go accompanied by a technological description and definition of technology-readiness levels. Despite implying and improvement for industrial processes of its use compared to CBA, TEA still lacks an appropriate regulatory foundation. In this regard, a new ISO standard is being developed (ISO/WD TS 14076) covering not only techno-economic analyses at any size or scale but also incorporating the environmental impacts. However, the inclusion of TEA in the legislation is far behind LCC which has already being recognized in directives (2014/24/EU and 2014/25/EU) and in standards (ISO 15663:2000 and 15686:2017).

Then, considering Tables 11.3 and 11.4, why scientists have decided to use TEA instead of LCC in their studies in the topic of anaerobic treatment? Generally, LCC has been associated with a product approach and a cost analysis though all life-cycle stages whereas TEA has been typically limited to an inherent investor perspective with gate-to-gate system boundaries (around the factory). However, the truth is that both TEA and LCC have their origin altogether in LCA methodology and both of them could be suitably used interchangeably in some contexts. It is then that LCC has a wider applicability whereas the scope of TEA is narrowing more and more to the stand-alone analysis of technologies and facilities. It is also very associated with the use of LCC the monetization of externalities (noise, environmental pollution, social effects and vibration, among others) which could partially be internalized (such as GHG emission taxes or waste-disposal costs) in the analysis. This could be appreciated for the studies dealing with anaerobic wastewater treatment. Out of the 33 studies shown in Tables 11.3 and 11.4 for TEA, only one has included environmental externalities whereas two of nine of the LCC studies were considering them. And yet, the sector seems to prefer the use of TEA instead of LCC (∼72% of the studies were carried out with TEA). On the contrary, only one publication classified its study as CBA. Regarding the type of wastewater, the percentage of economic analyses focused on stand-alone analysis of sewage sludge was higher (57.6%) than for co-digestion and for other flows. The use of anaerobic treatment for domestic wastewater has only been addressed by five publications.

11.3.1 Techno-economic emerging challenges

Although the environmental analysis for anaerobic treatment of the state-of-the art focused primarily on parameter optimization (i.e. temperature and solid concentration) and upstream and downstream

Table 11.3 Methodology and tools used for the economic analysis of anaerobic technologies in domestic treatment for sewage sludge.

Type of Resource	Methodology	Accounting Tools	Reference
Sewage sludge	TEA	OPEX, CAPEX, PBP and TAEC	Gholamian *et al.* (2023)
Sewage sludge mixed with lignocellulosic biomass	TEA	OPEX, CAPEX, NPV and IRR	Ebrahimi *et al.* (2023)
Sewage sludge	TEA	PBP, IRR, NPV and SIR	El-Qanni *et al.* (2022)
Sewage sludge	TEA	OPEX	He *et al.* (2022)
Sewage sludge	TEA	OPEX, CAPEX and NPV	Vinardell *et al.* (2022)
Sewage sludge	TEA	OPEX and PBP	Díaz *et al.* (2021)
Sewage sludge, food waste and fish sludge	TEA	OPEX, CAPEX, ROI and NPV	Fernando-Foncillas and Varrone (2021)
Food waste and sewage sludge	TEA	OPEX, CAPEX and NPV	Vinardell *et al.* (2021)
Sewage sludge and organic municipal solid waste	TEA	OPEX, CAPEX, NPV, IRR and PBP	El Ibrahimi *et al.* (2021)
Sewage sludge and food waste	LCC	OPEX and CAPEX	Andreasi Bassi *et al.* (2021)
Sewage sludge	TEA	OPEX, CAPEX and PBP	Bahreini *et al.* (2020)
Sewage sludge	TEA	OPEX and CAPEX	Medina-Martos *et al.* (2020)
Sewage sludge	LCC	OPEX, CAPEX, TAEC and ROI	Roldán *et al.* (2020)
Sewage sludge	LCC	OPEX, CAPEX, PBP and MARR	Cuéllar-Franca *et al.* (2019)
Sewage sludge	TEA	OPEX and CAPEX	Ranganathan and Savithri (2019)
Sewage sludge and organic municipal solid waste	LCC	NPV and IRR	Francini *et al.* (2019)
Sewage sludge and organic waste	CBA	OPEX, CAPEX and ROI	Thomsen *et al.* (2018)
Sewage sludge	TEA	OPEX and CAPEX	Dussan and Monaghan (2018)
Sewage sludge	LCC	OPEX and CAPEX	Tarpani and Azapagic (2018)
Sewage sludge	TEA	OPEX, CAPEX, NPV, IRR and PBP	Rus *et al.* (2017)
Sewage sludge	TEA	OPEX and CAPEX	García-Gutiérrez *et al.* (2016)
Sewage sludge	TEA	OPEX	Tomei *et al.* (2016a)
Sewage sludge	TEA	OPEX	Tomei *et al.* (2016b)
Sewage sludge	TEA	OPEX and CAPEX	Gianico *et al.* (2015)
Sewage sludge	TEA	OPEX, CAPEX and TAEC	Garrido-Baserba *et al.* (2015)
Sewage sludge	LCC	OPEX	Xu *et al.* (2014)

Note: CBA, cost–benefit analysis; CAPEX, capital expenditure; IRR, internal rate of return; LCC, life-cycle costing; MARR, minimum acceptable rate of return; NPV, net present value; OPEX, operational expenditure; PBP, payback period; ROI, return of investment; SIR, saving to investment ratio; TAEC, total annual equivalent cost; TEA, techno-economic assessment.

Table 11.4 Methodology and tools used for the economic analysis of anaerobic technologies in domestic treatment for wastewater resources.

Type of Resource	Methodology	Accounting Tools	Reference
Domestic wastewater	TEA	OPEX, CAPEX and NPV	Sanchez *et al.* (2023)
Domestic wastewater, blackwater, greywater and urine	LCC	OPEX and CAPEX	Lehtoranta *et al.* (2022)
Domestic wastewater	TEA	OPEX and CAPEX	Vinardell *et al.* (2020)
Blackwater	LCC	OPEX, CAPEX and NPV	Estévez *et al.* (2022b)
Blackwater and urine	LCC	OPEX and CAPEX	Postacchini *et al.* (2022)
Domestic wastewater	TEA	OPEX and CAPEX	Shoener *et al.* (2016)
Domestic wastewater	TEA	OPEX, CAPEX and NPV	Sills *et al.* (2016)

Note: CBA, cost–benefit analysis; CAPEX, capital expenditure; IRR, internal rate of return; LCC, life-cycle costing; MARR, minimum acceptable rate of return; NPV, net present value; OPEX, operational expenditure; PBP, payback period; ROI, return of investment; SIR, saving to investment ratio; TAEC, total annual equivalent cost; TEA, techno-economic assessment.

technologies, the economic studies were mainly aimed at creating a roadmap around anaerobic processes and how the process could overcome cost-effectiveness constraints. Therefore, the study of complementary technologies for anaerobic systems plays an important role in the quality (and thus the price) and marketability improvement of co-products.

11.3.1.1 Upstream and downstream

Although the economic studies centred on anaerobic processes implemented within the sector of wastewater treatment have thermal pre-treatment for sewage (i.e. hydrolysis and carbonization), domestic anaerobic treatment as a secondary process has been investigated by the implementation of forward osmosis and reverse osmosis as pre-treatments. Within this context, Vinardell *et al.* (2020) have analysed the feasibility of forward and reverse osmosis to concentrate diluted solutions. Given that the wastewater pre-concentration accounts for more than 74% of the total budget, pre-treatment worsens the costs associated with the treatment. Therefore, the stand-alone AnMBR operation is cheaper than the scenario with such pre-treatment, those being half of the scenario with pre-concentration. Because of the better economic performance of forward osmosis–AnMBR, Vinardell *et al.* (2022) studied the influence of the draw solution used during the operation of the system because it affects the membrane fluxes and the salinity (inhibiting anaerobic bacteria in the subsequent reactor). The most economically favourable draw solute was dependent on the membrane type but in any of the case studies proposed by the authors the economic impact from these substances was moderate. The hotspot still was the capital investment for pre-concentration. Although the forward and reverse osmosis for water reclamation is still not a viable configuration, pre-concentration of sewage sludge with membrane remains a promising approach.

The techno-economic analysis for domestic anaerobic wastewater treatment has been poorly addressed and the literature has been focused on sewage sludge. Two approaches can be found in the literature: AD is one of the multiple options for sludge treatment and AD is the main process discussed, but it is followed by other treatment. Garrido-Baserba *et al.* (2015), Tarpani and Azapagic (2018) and Xu *et al.* (2014) are examples of economic analysis for sewage treatment.

11.3.1.2 Feasibility of reactor configuration

Despite the large variability of reactors capable of operating at total solid concentration lower than 3% (CSTRs or continuous stirred tank reactors, UASBs, AFBs or anaerobic fluidized bed reactors, ECGS or expanded granular sludge bed reactors, MBRs, among others), the techno-economic analysis of domestic anaerobic treatment has only paid attention to the operation of MBRs and ABRs.

For example, Sills *et al.* (2016) compared the performance of ABRs with other systems such as the combination of an aerobic tricking filter with AD. This reactor provided a lower cycle cost (∼40% in agreement with the results of its net current value) because of both better capital and operating costs. It should be highlighted also the lower solid production of this system, which reduced the costs associated with disposal from 25 to 7%. Regarding the implementation of MBRs, membrane purchase seems to be the largest hotspot (over 49%) of the analysis followed by chemical addition for cleaning (between 0 and 26% depending on the flowrate) (Sanchez *et al.*, 2023). Vinardell *et al.* (2020) also agreed on a higher contribution of capital expenditures (CAPEX) in membrane reactors (between 63 and 77%).

11.3.1.3 Influence of co-digestion on economic profile

The organic matter concentration of domestic wastewater is not expected to be as high as, for example, that of molasses, cheese whey or sewage, among others. What can be expected from co-digestion of organic waste and municipal wastewater? To clarify this issue, Vinardell *et al.* (2021) proposed a comparison of the following scenarios: (1) anaerobic secondary treatment with an AnMBR and with another AD reactors as side-stream for sewage sludge and food co-digestion, (2) the CAS system, (3) anaerobic secondary treatment but without co-digestion in the side-stream with AD and (4) another including nutrient recovery from the centrate of the side-stream AD. Although the analysis centred in the sludge line, the highest costs were also achieved by domestic WWTPs with an aerated secondary treatment because a larger amount of sludge with poor biodegradability is produced. The implementation of an AD AnMBR and co-digestion significantly increased the revenues (triple) despite the increase of the costs up to a 44%. Among the case studies with anaerobic co-digestion, the greatest benefits were obtained with the configuration in which the nutrients are treated in the mainstream. Co-digestion of food waste in a conventional domestic WWTP is not only preferable in terms of the facility itself, but also in comparison to the separate treatment of both types of waste.

11.3.1.4 Nutrient recovery efficiencies

The nutrient recovery can be performed both in the water and sludge lines and brings multiple benefits beyond the transformation of pollutants into marketable products. This strategy also supports the increase of quality of effluents and prevents the uncontrolled precipitation, which increases the performance efficiencies of unitary operations. However, nutrient recovery poses a challenge because it may compromise the economic feasibility of the processes. When implemented in the sludge line, the question to be solved is whether the precipitation of phosphorus should be performed before or after AD. Considering the results achieved by Roldán *et al.* (2020), there is 1% difference between both options, which ensures the viability of the processes when the LCC analysis is expressed by an amount of phosphorus recovered. However, a change in the FU per m^3 unit led to a drastic change in the direction of the results because only one of the newly proposed alternatives was viable. This is related, thus, to the lower maintenance costs associated with the uncontrolled phosphorus precipitation and higher biogas production of the alternative considering the phosphorus precipitation before AD.

When anaerobic treatment is applied in the water line, costs are saved compared to aerobic systems, as aeration requirements are avoided. However, anaerobic processes are not capable of removing nutrients such as phosphorus and nitrogen by themselves and thus the effluent do not comply with European standards. Therefore, resource recovery technologies should be installed in the facility to maintain the emission thresholds. Because of this, the economic feasibility of the entire system is compromised because in many cases the methane production of the water line does not offset the higher CAPEX and operational expenditure (OPEX) associated with it (even for concentrations higher than 1100 mg/L). It should be considered that the biogas from the side-stream AD treating the sewage sludge from the main anaerobic treatment located in the water line may not also be sufficient to provide a profitable system (Vinardell *et al.*, 2021).

11.3.1.5 Valorization of the raw biogas

The valorization pathway selected for the raw biogas produced during anaerobic treatment has a strong influence on the economic viability of the facilities, as each of the co-products is co-related to a market price and to a specific post-treatment cost. The selection of the best technology has not been an easy task within this topic as each of the researchers compared different technologies with each other, and all drawn different conclusions. The knowledge in this section comes specifically from publications on AD of domestic sewage sludge treatment. However, and as the product obtained from the anaerobic secondary treatment of the domestic wastewater is also biogas, the recommendations and guidelines from the environmental studies of sewage sludge anaerobic treatment can also be applied. Volpi *et al.* (2023) recommended the use of biogas for the production of PHA, Purwanta *et al.* (2022) for co-firing in boilers, Alfonso-Cardero *et al.* (2021) for electricity production and Fuess and Zaiat (2018) for biomethane production.

Besides upgrading and transformation of the raw biogas, the CO_2 in the biogas can also be transformed into a vast number of co-products. In this regard, Cuéllar-Franca *et al.* (2019) compared four technological configurations for the production of fuels: the capture of the CO_2 by adsorption with mono-ethanolamine (MEA) from the raw biogas, the capture of CO_2 with MEA from the flue gases of combusted raw biogas, combustion of raw biogas and direct use of the CO_2 in the flue gases and capture of CO_2 from raw biogas, combustion and final use of flue gases. The highest capital and operating costs arise from the CO_2 capture by absorption from the flue gas, with a difference of ~13% compared to the best scenario (CO_2 capture from unsweetened biogas). Despite being the most expensive process, it has the most cost-effective design. This is because the liquid fuel has a higher price.

11.4 ANALYSIS OF WASTEWATER TREATMENT TECHNOLOGIES AND PROCESSES WITH LCA AND LCC

11.4.1 LCA approach for wastewater scenarios: a case study

Despite the growing interest in the application of LCA over the last decade, the complexity involved in collecting data throughout the entire life cycle, especially for novel practitioners, remains a threshold to overcome. This becomes more evident across sectors, as the focus (i.e. process or product) and logic (existence of cross-functionality) of the analysis vary depending on the objective of the study. To resolve some issues related to the applicability of LCA in the field of wastewater, Corominas *et al.* (2020) proposed a practical guide with a hypothetical approach for the sector. In this sense, this chapter aims to perform an LCA for domestic anaerobic treatment with experimental data as a starting point. Therefore, the main focus of the example provided is more on the early stages of research and development than on decision making (product purchasing or development of environmental legislation) or marketing (eco-design).

In this sense, the environmental profile of a sequential granular UASB bioreactor has been analysed taking into account the effects of changing temperature and hydraulic retention time. For this purpose, laboratory experimental data were obtained from the study of Stazi *et al.* (2022). Four scenarios were constructed based on two aspects: data availability and comparability between scenarios (Table 11.5). Effluent composition, operating temperature, product characterization (purity and amount of biogas), hydraulic retention times and pump run times are some of the raw data collected. Because two variables were analysed, the scenarios were named accordingly. 'T' refers to temperature whereas 'H' refers to hydraulic retention time. The numbers just after each letter correspond to the temperature in degrees Celsius and the hydraulic retention time in hours. Thus, the stage operating at 35°C and 22 h will be referred to as T35H22.

Table 11.5 Data collection from the experimental study of Stazi *et al.* (2022).

Parameters	Units	Scenario T35H22	Scenario T25H22	Scenario T35H14	Scenario T35H09
Working volume of the reactor	L	0.90	0.90	0.90	0.90
Number of pumps	–	3.00	3.00	3.00	3.00
Feeding volume/cycle	L	0.50	0.50	0.50	0.50
Feeding time	min	12.00	12.00	12.00	12.00
Sedimentation time	min	30.00	30.00	30.00	30.00
Effluent discharge time	min	8.00	8.00	8.00	8.00
COD in the influent	mg/L	500.00	500.00	500.00	500.00
Total nitrogen (TN) in the influent	mg/L	53.00	53.00	53.00	53.00
Total phosphorus (TP) in the influent	mg/L	5.00	5.00	5.00	5.00
Temperature of operation	°C	35.00	25.00	35.00	35.00
Reaction time	min	670.00	670.00	420.00	230.00
HRT	h	22.00	22.00	14.00	9.00
COD in the effluent	mg/L	18.50	39.00	27.00	34.00
Nitrogen in the effluent as NH_3	mg/L	60.50	52.50	59.00	59.50
Phosphorus in the effluent as PO_4^-	mg/L	7.75	8.10	7.25	7.10
Total dissolved solids (TSS) in the effluent	mg/L	15.00	13.50	15.00	15.00
Produced biogas	m^3/kg COD removed	0.24	0.21	0.22	0.18
Methane in biogas	%	75.50	53.00	75.50	75.50
Concentration of methane in the effluent	mg/L	7.09	7.34	7.09	7.09

T35H09, 35°C of temperature and 9 h of hydraulic residence time; T35H14, 35°C of temperature and 14 h of hydraulic residence time; T25H22, 25°C of temperature and 22 h of hydraulic residence time; T35H22, 35°C of temperature and 22 h of residence.

Regardless of the sector or technology to be analysed, LCA should always proceeds according to ISO 14040 and 14044 standards, so the analysis of environmental aspects should be carried out following their four stages.

Goal and scope: One of the main assumptions when approaching anaerobic wastewater treatment is whether to consider the digestate/effluent as a discharge, a stream to be further treated, or a co-product. Generally, effluents from AD processes do not meet the discharge thresholds set by European legislation and, therefore, the removal of some compounds such as nitrogen or phosphorus should be considered. This feature could be managed by an extension of system boundaries to incorporate the respective treatment technologies. However, many other questions must be answered before decisions can be made: Is the analysis focused on reactor optimization? Is the process decentralized and space limited? Does the country where the technology is implemented have restrictive emission limits? On the contrary, the circular economy concept is driving the recovery of nutrients from wastewater streams. However, direct reuse of liquid and solid digestate is sometimes limited by crop type, proper fertigation practices, nutrient concentration in effluent, presence of metals and pathogens (which are only removed at higher temperatures), distance from facility to point of application, community acceptance and economic feasibility (Helmecke *et al.*, 2020). The complexity of the treatment paradigm detailed above is something that must be reflected by LCA practitioners through the definition of system boundaries. As one of the key aspects, system boundaries should predefine the set of unit processes, inputs and outputs for total emissions accounting and is therefore fundamental in the creation of the LCI.

The system boundaries should consider three dimensions: geographical, temporal and technical (Li *et al.*, 2014). Among them, the last one has a particular importance among the wastewater scientific community, corresponding to data collection for new technologies. However, the other two are rarely defined due to the limitations of the environmental characterization factor in the databases. For example, the Eco-invent database only covers 20,000 datasets, far fewer than the 204 million organic substances, alloys, mixtures, polymers and salts accounted for in the Chemical Abstract Service (2023) record. Moreover, many of them are not country or region specific and the inventory refers to a global or European approach. When studying a sequential granular UASB reactor, the system boundaries can therefore only be threefold: cradle-to-gate, gate-to-gate and cradle-to-gate with expansion/substitution/boundary partitioning allocation. In the first approach, emissions are accounted for from raw material extraction to UASB effluent.

In the second case, according to International Reference Life Cycle Data (ILCD) guidelines, direct foreground process emissions, such as methane or carbon dioxide, would be included. The latter perspective is the most difficult to define and a full consensus among LCA practitioners has not yet been achieved. Depending on how emissions are accounted for, three main schools of thoughts can be distinguished as: attributional, consequential and socioeconomic (Moretti *et al.*, 2020). The pure attributional approach advocates that environmental impact should only be described by physical causalities entering and leaving through system boundaries. In contrast, the pure consequential approach aims at the description of the change of the environmental profile in response to a given modification in process technologies or the assessment of the future marginal energy supply. However, not many of the studies referring to anaerobic processes in wastewater treatment (11% according to the results in Tables 11.1 and 11.2) present how these emissions were reported. So, how to approach the limits of the cradle-to-gate system with expansion/substitution for sequential UASB reactors? One option could be to compare the scenarios with an attributional perspective considering that all viable alternatives provide the same number of products. This implies 'expansion' of the system boundaries to introduce other productive processes such as fertilizer or energy industries. 'Substitution' is, however, more typical of the consequential approach and is implemented through the avoidance of emissions from such production processes. Table 11.6 summarizes the decisions made at the goal and scope stage to carry out the case study described in this chapter.

Secondary treatment of domestic wastewater becomes, with the implementation of an anaerobic UASB, a multifunctional system capable of reducing direct pollution to water resources and simultaneously providing marketable products. Therefore, the complete definition of the system boundaries also depends on the function selected and the strategy followed to achieve the objective of the study. The functions of anaerobic wastewater treatment can be oriented at three different levels: micro-level, macro-level and accounting decision support (European Commission *et al.*, 2011). The first is closely related to specific products, the second to the comparison of technological scenarios, material strategies and policy options and the third is interested in the documentation of what has happened or will happen based on decisions already taken. An example of a micro-level approach in the case of AD for the example provided in this chapter could be the comparison of the electricity and liquid bio-fertilizer produced versus products already on the world market.

As far as the process or macro-level analysis is concerned, multiple strategies can be cited: technological optimization by analysing the weak points and identifying the flows with the highest environmental impact, forecasting and comparing technologies, affecting the remaining unit operations of a facility and the overall impact through technology implementation, geographical coverage and system response to specific environmental conditions (especially in the case of open-air reactors, as the profile may change between regions for the same technological approach), as well as the use of specific characterization factors. All these strategies belong to one of the following groups: planning, design and operation/optimization. Within the first group one could include decision-making studies regarding the selection of treatment alternatives, the second group refers mainly to the elucidation of environmental hotspots and the anticipation of impacts and the third to the creation of roadmaps to ameliorate environmental impacts. Therefore, it should be considered whether the objective of the LCA is to provide guidance on planning, design or operation. Because the research objective of Stazi

Table 11.6 Summary of the attributes of the LCA for the case study.

Attribute	Description
Initial hypothesis	(1) Two analyses were used for comparison: (a) the effluent of the UASB is directly discharged and (b) the effluent is directly reused as fertilizer source. (2) The biogas has been valorized into heat and electricity in a heat and power unit with an efficiency of 45% for heat and 35% for electricity. (3) Because of the scale of the facility, a conservative value of 0.4 m distance between equipment has been assumed to estimate the electricity consumption in the pumps. Readers should be aware that the distance may change in accordance with the location/geographical aspects of the area of implementation and with the scale-up. (4) Except for the energy demand of the pumps (whose efficiency is expected to be higher at higher design scales), it is assumed that the results of the remaining parameters do not differ from other scales. (5) The reactor was sized considering the up-flow velocity indicated by Stazi *et al.* (2022) and a height/diameter ratio of 3. (6) The pH control could not be included because in laboratory experiments the amount of chemicals (sodium bicarbonate) consumed is not usually recorded and the initial and final pH remain unchanged. (7) Average data estimated from the upper and lower limits of the operating conditions for each of the scenarios analysed.
Technical system boundaries	Cradle-to-gate for the hypothesis (a) and cradle-to-gate with system expansion for hypothesis (b)
Geographical system boundaries	Continental/global level
Temporary system boundaries	Not applicable
Emission accounting approach	Attributional
Strategy group	Operational, optimization
Level/scale of implementation	Macro-level or process-oriented
FU	1 m^3 of wastewater treated

et al. (2022) for the technology was the optimization of temperature and hydraulic retention time in a domestic wastewater treatment reactor, the case study provided can be framed within operation/optimization.

Finally and as mentioned earlier in Section 11.2, the first stage (called objective and scope) of the LCA should also include the identification of the FU. As can be seen in Tables 11.1 and 11.2 in Section 11.1, the most recurrent FU in AD for wastewater treatment is the volume of treated wastewater or '1 m^3 of treated wastewater'. This FU has also been selected by many other authors focused on the analysis and comparison of technologies to be implemented as anaerobic secondary treatment. Examples are the studies of: Boldrin *et al.* (2022), de Sampaio Lopes *et al.* (2014), Harclerode *et al.* (2020), Laitinen *et al.* (2017), Patel and Singh (2022), Pretel *et al.* (2013), Pretel *et al.* (2016), Sills *et al.* (2016) and Smith *et al.* (2014). However, this FU is only representative of one of the system functions: wastewater treatment. The product approach should be considered with other FUs such as: 1 m^3 of biogas, 1 kWh or 1 kg of methane produced, 1 kg of nutrients recovered and 1 m^2 of soil fertilized.

Life-cycle inventory: In LCA, two LCI perspectives can be differentiated depending on the sources of data collection. In process-based inventories, data have been collected with a 'bottom-up' approach and use primary input–output data associated with the foreground system process. On the contrary, input–output inventories follow a 'top-down' approach and have been collected from statistical data.

In the latter case, emissions are estimated from the selling prices of consumables/processes used and are therefore more suitable for regional/national scales than for industrial/facility scales. Although the data from Stazi *et al.* (2022) cannot be considered primary due to its bibliographic origin, it is certain that similar LCAs can be performed from laboratory data that have not already been published. Consequently, the nature of the study forces the foreground LCI to be a process-based inventory compiled from a 'bottom-up' approach. In addition to the application of mass and energy balances, software databases (for this case SimaPro®) were used to fill in gaps related to background processes, such as electricity and heat production. The LCI should be completed throughout the operation, maintenance, construction and demolition phases of a process/technology lifetime. However, the case study was only exemplified for the operational phase and the others were outside the system boundaries. For this reason, the LCI is constituted by direct and indirect emissions but only related to the treatment process.

Therefore, the emissions for anaerobic treatment in a UASB can be divided into direct and indirect. Direct emissions are considered within the boundaries of a gate-to-gate or inside-the-fence system, as they can be measured or modelled within a foreground system (Li *et al.*, 2022). As a secondary biological treatment, emissions come from the microbial metabolic activities of the biomass involved in the unit operations or from the limited removal of the target pollutant. Estimation of such emissions can be performed with direct laboratory measurements, with IPCC (Intergovernmental Panel on Climate Change) guidelines, with characterization factors exposed in the scientific literature with similar objectives and from process modelling. However, the latter options may resort to overestimation or underestimation of such emissions, as inhibitory effects and other operational particularities of a system are not included. Therefore, only methane and carbon dioxide emissions were considered for the LCA example provided in this chapter. A comparison of scenarios has been shown according to commonly measured parameters at a laboratory scale. In addition, many of the remaining emissions were not considered relevant in other publications and were therefore assumed to have an impact of <5% (cut-off criterion) (Laurent *et al.*, 2020). On the contrary, and depending on the composition achieved by the technologies, some of these components, such as methane, may rather be considered as valuable output. This is because biogas first undergoes a combustion process to produce electricity and/or heat.

Indirect emissions are usually defined for broader system boundaries, such as cradle-to-gate and cradle-to-grave, as emissions upstream (inputs from the Technosphere) and downstream (outputs to the Technosphere) of the target process or product may be considered background (system but not under the direct control or decisive influence of the producer of the good) processes and are often predefined in regulatory databases. Indirect and direct emissions were included in the case study. It should be noted that depending on the hypothesis and initial boundaries, the data may be reorganized differently. Table 11.7 specifically defines effluent nitrogen and phosphorus as emissions and not as feasible outputs.

Life-cycle impact assessment: This third stage of LCA aims to transform with impact assessment modelling factors the inventory in Table 11.7 into results that can be understood in terms of environmental impact or damage. It is further subdivided into the following stages: selection and definition of impact categories, classification, characterization, normalization and weighting. Unlike other LCA methodologies, such as carbon or water footprinting, LCA is a multidimensional methodology. Therefore, a multi-criteria analysis must be provided through the investigation of the results of indicators or categories. Many methods are currently available from which categories can be selected: USEtox, ReCiPe, IMPACT 2002+, TRACI, EDIP, CML, MEEUP, EPS, IPCC, Eco-indicator and LIME (European Commission *et al.*, 2011). The first pre-selection of the method should be performed on the level at which the impact categories should act, namely midpoint or endpoint. This implies that the results will be shown for different parts of the environmental value chain. Furthermore, the selection of the categories does not follow a strict rule; however, the usual practice is to select the method based on the categories of interest. The other option could be to choose the categories independently of the method. However, attributes such as completeness of

Table 11.7 LCI per FU (1 m^3 of wastewater treated) for the case study defined from the data of Stazi *et al.* (2022).

LCI Materials	Units	Scenario T35H22	Scenario T25H22	Scenario T35H14	Scenario T35H09
Inputs from Technosphere					
Feeding pump	kWh/m^3	0.41	0.41	0.41	0.41
Recirculation pump	kWh/m^3	22.99	22.99	14.41	7.89
Effluent pump	kWh/m^3	0.27	0.27	0.27	0.27
Heat	kWh/m^3	18.98	5.92	19.01	19.08
Biogas blower	kWh/m^3	2.51×10^{-3}	2.67×10^{-3}	2.21×10^{-3}	1.60×10^{-3}
Outputs to Technosphere					
Electricity	kWh/m^3	2.93×10^{-1}	1.77×10^{-1}	2.69×10^{-1}	2.11×10^{-1}
Heat	kWh/m^3	0.00	0.00	0.00	0.00
Outputs to the Nature					
Methane (CH_4)	kg/m^3	7.09×10^{-3}	7.34×10^{-3}	7.09×10^{-3}	7.09×10^{-3}
Carbon dioxide (CO_2)	kg/m^3	1.97×10^{-1}	1.70×10^{-1}	1.81×10^{-1}	1.42×10^{-1}
COD	kg/m^3	1.85×10^{-2}	3.90×10^{-2}	2.70×10^{-2}	3.40×10^{-2}
Nitrogen as ammonia (NH_3)	kg/m^3	6.05×10^{-2}	5.25×10^{-2}	5.90×10^{-2}	5.95×10^{-2}
Phosphorus as phosphate (PO_4^-)	kg/m^3	7.75×10^{-3}	8.10×10^{-3}	7.25×10^{-3}	7.10×10^{-3}
Dissolved solids (TSS)	kg/m^3	1.50×10^{-2}	1.35×10^{-2}	1.50×10^{-2}	1.50×10^{-2}

scope, robustness and uncertainty, reproducibility, transparency or stakeholder acceptance would vary among the selected categories. For example, the ILCD manual 'Recommendations for Life Cycle Impact Assessment in the European context' provides guidelines for pre-selection of methods based on the most common impact categories (climate change, ozone depletion, human toxicity, respiratory particulates/inorganics, photochemical ozone formation, ionizing radiation impacts, acidification, eutrophication, eco-toxicity, land use and resource depletion).

However, the guidelines were published in 2010 and many methods have since been updated as ReCiPe 2016. The selection of impact categories for the case study aimed at representativeness of the following aspects: background energy consumed, implications of biogas production in reducing resource depletion, relevance of direct emissions control and water resource pollution. For these reasons, the categories analysed were climate change, ozone depletion, eutrophication and resource depletion. During the classification stage, the LCI data are assigned to each of the previously selected impact categories. All of these categories can be found in CML and ReCiPe, two of the most widely used methods in the field of AD (see Tables 11.1 and 11.2). After selection of the LCIA method, the connection between the LCI and the environmental impact/harm is performed during the classification and characterization stages. Environmental loadings are assigned to each LCIA data element and category using factors. Both stages can be supported by the use of software such as SimaPro, EASETECH and OpenLCA, among many others. In particular, the first three LCIA stages for the environmental comparison of the UASB granular sequential reactor scenarios have been performed with SimaPro version 9.3 for the ReCiPe 2016 Midpoint (H) V1.07/World (2010) H method. Normalization and weighting, the optional ICLC stages were not included within the study.

Interpretation of the results: All of the above phases of LCA are interrelated. The objective of this analysis is not only to provide a clear message to the readers about the environmental expectations of the analysed process, product or technology, but it is also relevant for the identification of missing data and errors. Therefore, the LCA is iterative, and the results obtained at this stage lead to the modification or redesign of the assumptions adopted in the previous stages. When interpreting the results, the first step should be a data and sensitivity analysis, because the results depend on the decisions made throughout the design process. This should be especially important when using data from unreliable sources, numerical data that falls within a range, assumptions, and when alternative production routes can be modelled. Referring to the case study proposed for Stazi *et al.* (2022), ranges of data can be found regarding pollutant removal efficiency, assumptions were used for reactor sizing and distances between pumps and other equipment and the European profile was used to power electrical devices (the more renewable the profile, the lower the environmental impact). To check whether some of these parameters represent a significant change in the results, the sensitivity analysis should be linked to a contribution statement. The hotspots or elements of the process system that have the greatest impact on the overall profile should be identified. Depending on the details of the contribution analysis, the results could highlight the species of substances impacting the environment, the processes involved (i.e. energy or chemicals) or sections of a facility (i.e. primary treatment, secondary treatment). Finally, for comparative analysis, the results of different scenarios can be contrasted with each other. Figure 11.1 shows the visualization of the contribution of one of the four scenarios analysed, as well as the comparison among all scenarios considering hypothesis (a) of the analysis (Table 11.6).

Table 11.6 highlights the two initial hypotheses of the LCA study conducted: discharge of the effluent directly into the environment and valorization of the effluent compounds as liquid fertilizer. Under the first hypothesis, five background processes and six polluting substances emitted to nature are responsible for the current distribution of the environmental profile in Figure 11.1a. Among them, the use of electricity in the pump needed for agitation and recirculation of the stream inside the reactor is the main contributor in three (between 54.8 and 78.6% for FRS or fossil resource scarcity and SOD or stratospheric ozone depletion, respectively) of the four impact categories under study. The ME (marine eutrophication), on the

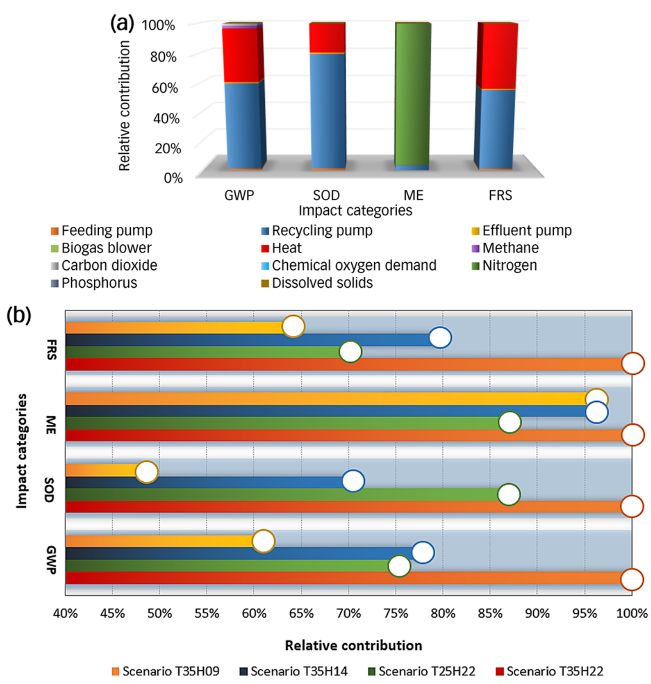

Figure 11.1 Environmental relative contribution profile for the four scenarios of the case study analysed: (a) Process contribution for the scenario T35H22 and (b) scenario comparison. T35H09, 35°C and HRT of 9 h; T35H14, 35°C and HRT of 14 h; T25H22, 25°C and HRT of 22 h; T35H22, 35°C and HRT of 22 h.

contrary, is affected 96.2% by nitrogen emissions to water resources. With the exception of the marine eutrophication category, heat consumption is the second most important process in the profile with a minimum and maximum representativeness of 19.1 and 43.6% for SOD and FRS. This is due to two reasons: the five listed background processes are energy related, with an LCI underlining a much higher influence of electricity demand on the recirculation pump, and the unit process (expressed in impacts per kWh) of electricity is more polluting than the selected heating process. In this context, effluent recirculation is the main hotspot of the process when the distance between units is 0.4 m. A reduction of the length of the pump suction and discharge piping to 0.1 m would lead to completely different results, as the electricity demand would no longer be the main hotspot. Unlike temperature, hydraulic retention time and influent concentrations, the distance between equipment is not an intensive variable. Thus, full-scale anaerobic treatment systems may have pump arrangements and distances differing from those indicated within this chapter. For such reason, the results of the herein described LCA have been proposed as an example for the identification of environmental weak points from an early stage of design. The procedure could be repeated for larger scale technologies or facilities but considering that not all the parameters from the inventory are going to relate proportionally to the scaling-up of the facility. Another option to decrease the environmental impact of electricity use would be a modification of the hydraulic retention time. This is due to the relationship between the running time of the pump used in effluent recycling and the energy demand. To this effect, a comparison of three scenarios is shown in Figure 11.1b. Scenarios T35H09, T35H14 and T35H22 demonstrate how a shorter hydraulic residence time is accompanied by a decrease in environmental impact by an order of magnitude of up to 51.5%, from 22 to 9 h. Lower operating temperature also translates into better results (up to 30%). The answer to which of the two parameters (temperature or HRT (hydraulic retention time)) is more relevant for the profile with the same degree of modification cannot be obtained with this case study, as the hypotheses were built on experimental data. However, it appears that for modifications ~30% the results depend on the distance from the recycle pump to the reactor. For example, three of the categories analysed are favourable to T35H14 for 0.4 m, whereas the same hypothesis is worse than T25H22 for 0.1 m.

When comparing initial hypotheses, A with B, the qualitative LCA results are similar. In both cases, T35H22 is the worst scenario in all impact categories and T35H09 is the best. Accordingly, the results are congruent regardless of the adopted scenario. The decrease in the environmental profile between the most and least polluting scenarios is, however, 11.8% higher for ME under scenario B. This means that T35H09 is still a better alternative in terms of lower effluent nutrient composition and thus under-recovery. The reason for this is the higher affect of a direct emission compared to fertilizer production from other sources (a comparability of the scenarios was performed considering that all of them could provide the same amount of phosphorus and nitrogen fertilizer).

11.4.2 How to approach the techno-economic analysis of anaerobic domestic wastewater treatment

The objective of wastewater treatment plants is to eliminate the pollutants present in the water in order to return the water to the environment, causing the minimum environmental impact, or to use it for other purposes in order to reduce the pressure on conventional water sources. To carry out wastewater treatment, a set of physicochemical treatments are necessary; the degree of treatment required for wastewater depends mainly on the effluent discharge limits. Most facilities have a conventional type of treatment, which consists of pre-treatment, primary and secondary treatment. In turn, various processes mentioned above involve the use of a wide variety of assets and can be grouped into civil construction, piping and electromechanical equipment.

As in any other industrial process, these infrastructures require a series of costs: energy, personnel, reagents, maintenance and so on, for the operation of facilities. In this sense, as with investment costs, these can vary depending on multiple variables that must be considered. Precisely, one of the first phases in any investment project is the feasibility analysis. The study makes possible to assess the profitability and financial sustainability of a project in the long term. This is particularly important when, as is the case in the wastewater treatment sector, there are different treatment technologies available. Another important aspect of feasibility analyses is that they enable decision makers to assess the environmental, social and legal implications of a project. These assessments can help prevent risks and ensure that the project complies with applicable legal and environmental regulations.

A wastewater treatment project is viable when, in addition to technically complying with the legal criteria regarding the quality of the influent, it is sustainable in the long term, which implies identifying a priori all the aspects associated with the process that may jeopardize the operation itself in short- and medium-term future. From the techno-economic point of view, some treatment technologies may require higher operating costs but with a lower investment or, on the contrary, technologies requiring higher economic investment that minimize operating costs. However, any economic analysis must take into account the unique characteristics associated with the site of the facility, local energy costs, specific quality and regulatory requirements, for example. In this sense, the economic analysis of the treatment process can include, in a second stage, the quantification of the benefits generated, or, in other words, the resources produced in the process. In the case of anaerobic treatment, in addition to generating water resources for irrigation, it also generates an alternative source of energy in the form of biogas that is converted into heat or electricity to save energy resources, thereby reducing GHG emissions. With respect to sludge generation, compared to other treatments, it is lower, which reduces the costs associated with sludge disposal. This stabilized sludge can be used as a fertilizer to enrich and improve soil characteristics. In summary, understanding the wastewater treatment infrastructure as a source of resources, which can be quantified, makes it possible to integrate the benefits generated into an overall decision balance (Figure 11.2).

In scenarios where there are different technical alternatives, cycle cost analysis can help to identify which option is economically more viable taking into account the entire investment period. Life-cycle cost, also known by its acronym LCC, is used to evaluate project costs. LCC is a method of economic analysis used to evaluate the total cost of ownership of a product or service over its entire life cycle. This method is based on life-cycle thinking and takes into account all costs associated with a product or service, from acquisition to disposal. The main objective of LCC is to provide a complete and accurate picture of the total costs associated with a product or service, to help companies and individuals make more informed procurement decisions. LCC calculations can be used on any piece of equipment or treatment system to determine the cost of procurement, operation, maintenance or disposal over its lifetime. There are currently a wide range of models used to calculate the total cost over the lifetime of a product. Regardless of the model used, they all share a common objective: to provide an accurate estimate of total pump system costs over time, expressed in today's currency value. In this sense, the result of an LCC can be used to compare different products or services and choose those with a lower total cost.

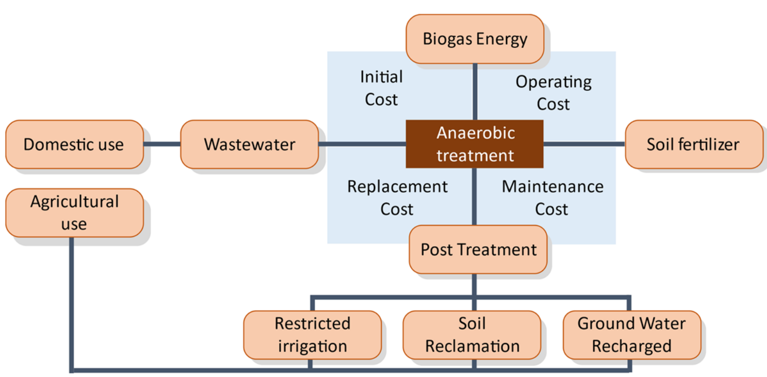

Figure 11.2 Anaerobic wastewater treatment costs and resource generation.

Naturally, the weighting of certain factors in an LCC analysis will depend on local circumstances. For example, in countries with low-energy prices or for stations that run infrequently, energy costs may not be a major factor. Similarly, maintenance costs will not be a major factor in locations where labour is inexpensive. The advantage of an LCC analysis is that it lets the user focus on factors that matter most for a specific treatment system and situation. LCC can help in making informed decisions in the design and planning of the project. By evaluating the total costs associated with different design and planning options, the most cost-effective and sustainable option in the long term can be chosen. In addition to the great usefulness of LCC in the design phase of the project, it is also particularly useful throughout the operation period of the project. By evaluating the total costs associated with the project, areas where resource consumption such as water and energy can be reduced, and waste-disposal costs can be minimized, can be identified. In this way, improvements can be made that increase the sustainability of the project.

LCC is divided into several steps, including identifying all costs associated with a product or service, estimating the future cost and projecting the cost over the life cycle. In general, we can simplify the following four costs:

(1) Acquisition costs (CAPEX). These are all costs associated with the initial investment plus research and development costs, corresponding to the engineering and construction, testing, transfer and integrated logistical support activities incurred to incorporate an asset into an organization.
(2) Operating costs (OPEX). They correspond to the variable costs derived from the operation of the system in accordance with the planned degree of activity (which may be hours/year).
    (2.1) Maintenance costs. These reflect the consumption of resources derived from preventive and corrective maintenance, basic infrastructure, spare parts and associated consumption (not considered in the acquisition), repairs, modifications and/or modernizations, to ensure the availability of the system to fulfil its mission.
    (2.2) Repair costs. These costs relate to technical failures or equipment breakdowns. They may require the intervention of a technician or the replacement of damaged components.

To summarize, the investment and/or acquisition costs are called CAPEX (capital expenditures) and the operating and maintenance costs are called OPEX (operational expenditures). The costs associated with fixed assets (CAPEX) take into account the useful life for which the infrastructures have been designed. Thus, they represent the capacity of the facilities to generate a profit over time. The costs associated with maintenance and operation tasks (OPEX) ensure the correct functioning of the facilities, optimizing their use and avoiding wear and tear and deterioration.

11.4.2.1 Capital expenditures

Investment costs correspond to all those costs related to the acquisition of the necessary assets and the start-up of the project. In the case in question, a wastewater treatment plant mainly involves land, piping, civil works (such as physical unit processes, biological reactors, degritting units and settlers, among others), electromechanical equipment (impulsion pumps, submersible pumps, blowers, centrifuges, etc.) and piping. Moreover, the infrastructure required to carry out water reclamation processes may include different types of technology and can vary depending on the quality of the reclaimed water, which will determine the technology chosen for the process.

It is important to note that the investment costs associated with reclaimed water infrastructure can vary depending on the specific circumstances of each project. However, despite the initial investment costs, these types of infrastructure can provide significant long-term benefits, such as reducing demand for freshwater resources and reducing the discharge of wastewater into surface waters, leading to improved environmental and public health outcomes. The percentage distribution of the investment cost for wastewater treatment and reclaimed water process infrastructure can vary depending on a number of factors, such as the type and size of the facility, the technology used, the geographic location and local regulatory conditions.

In the case of an STP, construction costs can represent ~70–80% of the total investment cost. Within construction costs, the cost of treatment equipment (e.g. biological and chemical treatment technology) and construction costs (e.g. installation of piping, construction of settling tanks and digesters) are the two major components (EPA, 2016). The remaining costs include the cost of land acquisition, engineering, design, licensing and permitting costs and administration and supervision costs. In the case of reclaimed water processing infrastructure, the cost of treatment equipment can represent the majority of the total investment cost. Advanced water treatment technology, such as reverse osmosis and membrane filtration, can be expensive to install and operate. Other costs include the cost of constructing and maintaining pipelines, storage tanks, pumping stations and other equipment associated with the distribution of reclaimed water. There are also costs associated with the management and oversight of the reclaimed water programme, including quality monitoring and regulatory compliance.

Another aspect to be taken into account with regards to investment costs is the useful life of the assets that make up these infrastructures in order to establish their depreciation. It should be borne in mind that the expected life of the different assets (infrastructures, electromechanical equipment and piping) may be defined by operational variables such as operating hours, or fixed variables such as the age of the element. However, there are references that allow these maximum useful life parameters to be established approximately.

A period of 30 years is considered appropriate for civil works and the rest of the first establishment costs, as although it may be necessary to remodel some equipment beforehand, the civil works are perfectly usable for a new installation. Shorter periods are established for the rest of the equipment, and in many cases not because they will continue to operate with good performance at the end of this period, but because of technical obsolescence, given that technical progress may make it advisable to replace them with others that are more efficient or better adapted to the real needs of the installation. With these criteria, the most frequently adopted periods are:

- Mechanical equipment: 12 years
- Membranes: 8 years
- Electrical equipment: 15 years
- Instrumentation and control: 12 years
- Piping: 15 years

The investment required to build any STP depends very much on its size. The specific investment (investment required for each m^3/day of production) decreases as the size of the plant increases, which means that the scale factor plays an important role in the investment. In addition to the scale factor, other aspects such as the quality of the influent and effluent plays an important role, which will determine the type of treatment required. The investment costs of a wastewater treatment system have been studied by numerous authors, many of whom, through the development of cost functions, provide a model to estimate the investment costs for different types of treatment. These authors include the work of Friedler and Pisanty (2006) who establishes a cost function for secondary treatment and advanced secondary treatment with nitrification or Singhirunnusorn and Stenstrom (2010) who develops cost functions for four types of technologies: activated sludge, oxidation tank, aerated lagoons and oxidation ponds. On the contrary, Tsagarakis *et al.* (2003) focus on the analysis of the investment, operation and maintenance costs of different secondary treatments whose main difference is aeration. They conclude that activated sludge treatments can be the most economical above a certain size. On the contrary, Rodríguez-Miranda *et al.* (2015) estimates the investment costs of wastewater treatment by differentiating between investment costs associated with primary treatment and secondary treatment.

With respect to anaerobic wastewater treatment, one of the most used types of treatment is anaerobic sludge blanket reactors. Sludge blanket reactors are a type of anaerobic treatment in which wastewater is passed through a floating 'blanket' of suspended sludge particles. As the anaerobes in the sludge digest the organic components of the wastewater, they multiply and accumulate into larger granules that settle at the bottom of the reactor tank and can be recycled for future cycles. The treated effluent flows up and out of the unit. Biogas resulting from the degradation process is collected through collection hoods throughout the treatment cycle. With respect to investment costs, Tchobanoglus *et al.* (2003), based on a sample of infrastructures with similar characteristics, suggest the following cost function:

$$IC = Q_d \times 35{,}877 \times Z^{-0.43} \tag{11.1}$$

where IC is the investment cost (€), Q_d is the design flow rate and Z is the equivalent inhabitants (design).

In this sense, it is important to point out the influence of economies of scale on investment costs, so that a larger infrastructure would imply a lower unit cost according to the design flow rate (Figure 11.3). These results coincide with studies by other authors relating investment and operating costs to economies of scale in the wastewater treatment sector (Hernández-Chover *et al.*, 2018).

11.4.2.2 Operational expenditures

In the urban water cycle sector, a large number of equipment and infrastructures are necessary to carry out the process, in this sense, operation and maintenance (O&M) costs may exceed the initial investment costs. The initial investment in a project can be significant, but it is often only a fraction of the total costs that must be considered over the entire life cycle of the project. O&M costs can include expenses such as energy required to operate the project, equipment replacement and repair, labour costs and material costs. It is important to take these costs into account when assessing the feasibility and profitability of a project. An effective approach to minimize O&M costs is to carefully plan and design the project from the outset, with the goal of minimizing maintenance and operational requirements and maximizing energy efficiency. In addition, implementing preventive and predictive maintenance programmes can help reduce long-term maintenance costs by detecting and addressing issues before they become serious problems.

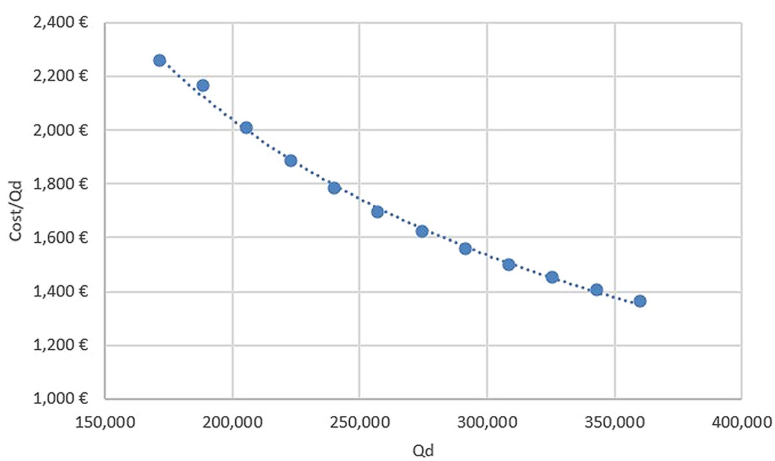

Figure 11.3 Investment costs and design flow rate (€/m³) for an UASB reactor.

Operating and maintenance costs can be divided into two main groups:

(a) Fixed costs, independent of the treatment flow (€/day), such as:
- Operating and maintenance personnel costs
- Electromechanical maintenance costs
- Monitoring and control, health and safety and administrative costs

(b) Variable costs, depending on the flow treated (€/m^3). These are
- Electricity costs
- Costs of chemical reagents
- Replacement costs of membranes or other treatment elements
- Waste management

Any analysis that aims to evaluate the operating costs associated with the wastewater treatment process will include technical and economic aspects that may affect the process. With respect to the technical aspects, the technology used can influence in a higher or lower pollutant collection and consequently in the formation of costs. At the same time, there are other variables that must be evaluated because of the influence they can have on the variation of the economic costs of the process. In the previous section, we have observed the influence of economies of scale on the investment costs of anaerobic treatment, so that the larger the size of the infrastructure, according to the design capacity, the lower the costs. With respect to operating costs, Hernández-Chover *et al.* (2018) confirmed that they have a similar behaviour, that is, larger infrastructure dimensions will generate a lower unit cost in terms of treatment (also shown in Figure 11.4).

There are other aspects that can influence the operational costs of these infrastructures: Sala-Garrido *et al.* (2012) showed that WWTPs located in tourist areas are affected by seasonality and variability of pollutant loads, which implies that these infrastructures operate at full capacity during part of the year, generating higher operational costs and possible problems with effluent quality.

As in the previous section, an economic function is proposed to project the costs associated with the O&M of these infrastructures:

$$OC = 64,286 \times Z^{-0.445} \tag{11.2}$$

where OC is the operational cost (€) and Z is the equivalent inhabitants treated.

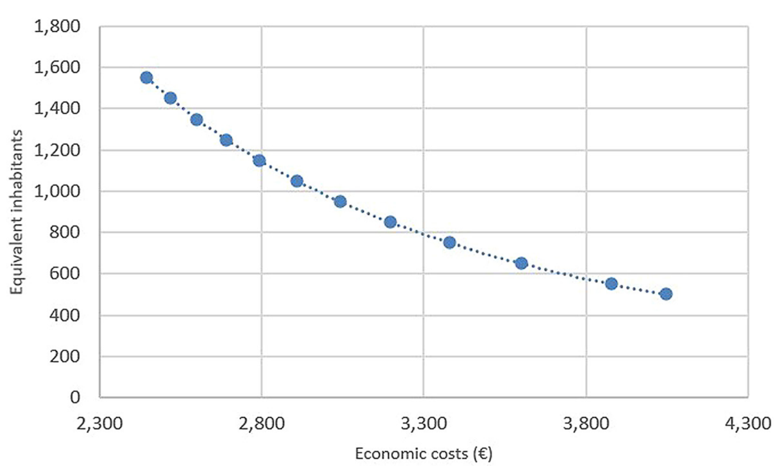

Figure 11.4 Operational costs per equivalent inhabitant treated for an UASB reactor.

The results indicate that the operational costs of this type of technology are influenced by the presence of economies of scale, so that a greater number of treated inhabitants would reduce the relative costs of the treatment process. In addition, it should be taken into account that higher organic loads would generate greater amounts of resources, such as energy in the form of biogas, sludge that can be valorized and water.

In this section, we have synthesized the economic costs associated with this type of facility, suggesting economic functions that model the investment and operating costs of an anaerobic treatment plant. The LCC methodology divides the costs according to their nature, so that the proposed function would be as follows:

$$\mathrm{LCC} = C_{\mathrm{ic}} + C_{\mathrm{in}} + C_{\mathrm{e}} + C_{\mathrm{o}} + C_{\mathrm{m}} + C_{\mathrm{s}} + C_{\mathrm{env}} + C_{\mathrm{d}}$$

where C_{ic} = initial costs. Initial costs relate to the cost of purchasing systems, piping and all mechanical and electrical equipment as well the cost of engineering, testing and inspection, including any spare parts and training; C_{in} = installation and commissioning costs. These costs can include civil work, foundations, connection of piping, electrical wiring and instrumentation. They also cover the cost of setting and grouting of equipment on the foundations, provisions for flushing as well as performance evaluations at start-up. The installation and commissioning of monitoring and control equipment is also included in this item. Installation time can be minimized or eliminated by selecting a pre-programmed variable speed drive that requires a minimum of configuration settings; C_{e} = energy costs. These costs can include civil work, foundations, connection of piping, electrical wiring and instrumentation. They also cover the cost of setting and grouting of equipment on the foundations, provisions for flushing as well as performance evaluations at start-up. The installation and commissioning of monitoring and control equipment is also included in this item. Installation time can be minimized or eliminated by selecting a pre-programmed variable speed drive that requires a minimum of configuration settings; C_{o} = operational costs. Operational costs cover the labour costs for normal operation of the pumping system. This includes, for example, normal wear and tear, system supervision and keeping the station clean. Operational costs do not include costs attributable to energy or maintenance of the treatment water system. An LCC analysis can be a good tool to see how fast the investment in a new supervision system will pay back;

C_{m} = maintenance and repair costs. Such costs relate directly to the total number of hours spent on maintenance and the cost of spare parts, including planned and unplanned maintenance; C_{s} = downtime costs. This category relates mainly to unexpected downtime but may also be due to a loss of production or even loss of trust from a customer. Downtime costs can be minimized by using maintenance contracts that ensure regular service to maximize uptime and shorten response time in the event of emergencies. Monitoring and control solutions can also create early warnings that help to prevent downtime; C_{env} = environmental costs. These include costs for dealing with spills, environmental inspections and contaminant disposal during the lifetime of the water treatment system. Such costs are often set by local regulatory authorities and vary from country to country. The disposal of used parts and materials is also included. For information about what materials are used and their effects on the environment, see the Environmental Product Declaration; C_{d} = decommissioning costs. Decommissioning costs usually include the disposal of the water treatment system and auxiliary services as well as restoration of the local environment. The decommissioning costs seldom vary for similar solutions and are often excluded from an LCC calculation.

An LCC analysis can be used to determine the total cost for the system over its lifetime. When conducting a complete analysis, it is necessary to gather and enter data for all eight categories in the formula. An LCC analysis can also be used to examine how beneficial an investment can be, meaning that only factors that are of relevance for the analysis need to be included. Making two analyses – one with the investment and one without – and comparing the results will show the payback time for the investment. When comparing different systems, the relevant data should be entered for the same categories. The comparison of technological alternatives should include, in addition to the technical aspects, the resources that each technology is capable of generating. The valorization of the resources generated can influence the reduction of the economic costs associated with both investment and operation. Thus, technologies with higher investment or operating requirements may be more viable options due to the resources they can generate over other alternatives.

ACKNOWLEDGEMENTS

The authors express their gratitude to the HP-NANOBIO project (PID2019-111163RB-I00) granted by the Spanish Ministry of Science and Innovation, to the SPOTLIGHT project (PDC2021-121240-I00) funded by MCIN/AEI/10.13039/501100011033 and European Union NextGenerationEU/PRTR and the BIORECER project (No. 101060684) funded by HORIZON-CL6-2021-ZEROPOLLUTION-01. S.E. also acknowledges the Spanish Ministry of Science, Innovation and Universities for financial support (Grant reference PRE2020-092074). S.E., M.T.M. and G.F. belong to Cross-disciplinary Research in Environmental Technologies (CRETUS Research Center, Spain, ED431E 2018/01) and the Galician Competitive Research Group (GRC ED431C 2021/27).

REFERENCES

Adghim M., Abdallah M., Saad S., Shanableh A., Sartaj M. and El Mansouri A. E. (2020). Comparative life cycle assessment of anaerobic co-digestion for dairy waste management in large-scale farms. *Journal of Cleaner Production*, **256**, 120320, https://doi.org/10.1016/J.JCLEPRO.2020.120320

Alfonso-Cardero A., Pagés-Díaz J., Contino F., Rajendran K. and Lorenzo-LLanes J. (2021). Process simulation and techno-economic assessment of vinasse-to-biogas in Cuba: deterministic and uncertainty analysis. *Chemical Engineering Research and Design*, **169**, 33–45, https://doi.org/10.1016/J.CHERD.2021.02.031

Alvarez-Gaitan J. P., Short M. D., Lundie S. and Stuetz R. (2016). Towards a comprehensive greenhouse gas emissions inventory for biosolids. *Water Research*, **96**, 299–307, https://doi.org/10.1016/J.WATRES.2016.03.059

Andreasi Bassi S., Boldrin A., Frenna G. and Astrup T. F. (2021). An environmental and economic assessment of bioplastic from urban biowaste. The example of polyhydroxyalkanoate. *Bioresource Technology*, **327**, 124813, https://doi.org/10.1016/J.BIORTECH.2021.124813

Arias A., Behera C. R., Feijoo G., Sin G. and Moreira M. T. (2020a). Unravelling the environmental and economic impacts of innovative technologies for the enhancement of biogas production and sludge management in wastewater systems. *Journal of Environmental Management*, **270**, 110965, https://doi.org/10.1016/J.JENVMAN.2020.110965

Arias A., Feijoo G. and Moreira M. T. (2020b). What is the best scale for implementing anaerobic digestion according to environmental and economic indicators? *Journal of Water Process Engineering*, **35**, 101235, https://doi.org/10.1016/J.JWPE.2020.101235

Arias A., Feijoo G. and Moreira M. T. (2021). Benchmarking environmental and economic indicators of sludge management alternatives aimed at enhanced energy efficiency and nutrient recovery. *Journal of Environmental Management*, **279**, 111594, https://doi.org/10.1016/J.JENVMAN.2020.111594

Asunis F., De Gioannis G., Francini G., Lombardi L., Muntoni A., Polettini A., Pomi R., Rossi A. and Spiga D. (2021). Environmental life cycle assessment of polyhydroxyalkanoates production from cheese whey. *Waste Management*, **132**, 31–43, https://doi.org/10.1016/J.WASMAN.2021.07.010

Awad H., Gar Alalm M. and El-Etriby H. K. (2019). Environmental and cost life cycle assessment of different alternatives for improvement of wastewater treatment plants in developing countries. *Science of the Total Environment*, **660**, 57–68, https://doi.org/10.1016/J.SCITOTENV.2018.12.386

Bahreini G., Elbeshbishy E., Jimenez J., Santoro D. and Nakhla G. (2020). Integrated fermentation and anaerobic digestion of primary sludges for simultaneous resource and energy recovery: impact of volatile fatty acids recovery. *Waste Management*, **118**, 341–349, https://doi.org/10.1016/J.WASMAN.2020.08.051

Becker A. M., Yu K., Stadler L. B. and Smith A. L. (2017). Co-management of domestic wastewater and food waste: a life cycle comparison of alternative food waste diversion strategies. *Bioresource Technology*, **223**, 131–140, https://doi.org/10.1016/j.biortech.2016.10.031

Blanco D., Collado S., Laca A. and Díaz M. (2016). Life cycle assessment of introducing an anaerobic digester in a municipal wastewater treatment plant in Spain. *Water Science and Technology*, **73**(4), 835–842, https://doi.org/10.2166/wst.2015.545

Boldrin M. T. N., Formiga K. T. M. and Pacca S. A. (2022). Environmental performance of an integrated water supply and wastewater system through life cycle assessment – a Brazilian case study. *Science of the Total Environment*, **835**, 155213, https://doi.org/10.1016/J.SCITOTENV.2022.155213

Brockmann D., Gérand Y., Park C., Milferstedt K., Hélias A. and Hamelin J. (2021). Wastewater treatment using oxygenic photogranule-based process has lower environmental impact than conventional activated sludge process. *Bioresource Technology*, **319**, 124204, https://doi.org/10.1016/J.BIORTECH.2020.124204

Cakir F. Y. and Stenstrom M. K. (2005). Greenhouse gas production: a comparison between aerobic and anaerobic wastewater treatment technology. *Water Research*, **39**(17), 4197–4203, https://doi.org/10.1016/J.WATRES.2005.07.042

Cañote S. J. B., Barros R. M., Lora E. E. S., dos Santos I. F. S., Silva A. P. M., Piñas J. A. V., Cañote A. L. B., Silva d. C. e. and L H. (2021). Life cycle assessment of upflow anaerobic sludge blanket sludge management and activated sludge systems aiming energy use in the municipality of Itajubá, Minas Gerais, Brazil. *Journal of Material Cycles and Waste Management*, **23**(5), 1810–1830, https://doi.org/10.1007/s10163-021-01253-0

Cao Y. and Pawłowski A. (2013). Life cycle assessment of two emerging sewage sludge-to-energy systems: evaluating energy and greenhouse gas emissions implications. *Bioresource Technology*, **127**, 81–91, https://doi.org/10.1016/J.BIORTECH.2012.09.135

Cartes J., Neumann P., Hospido A. and Vidal G. (2018). Life cycle assessment of management alternatives for sludge from sewage treatment plants in Chile: does advanced anaerobic digestion improve environmental performance compared to current practices? *Journal of Material Cycles and Waste Management*, **20**(3), 1530–1540, https://doi.org/10.1007/s10163-018-0714-9

Chemical Abstract Service (2023). CAS Registry. https://www.cas.org/

Chen R., Yuan S., Chen S., Ci H., Dai X., Wang X., Li C., Wang D. and Dong B. (2022). Life-cycle assessment of two sewage sludge-to-energy systems based on different sewage sludge characteristics: energy balance and greenhouse gas-emission footprint analysis. *Journal of Environmental Sciences*, **111**, 380–391, https://doi.org/10.1016/J.JES.2021.04.012

Colzi Lopes A., Valente A., Iribarren D. and González-Fernández C. (2018). Energy balance and life cycle assessment of a microalgae-based wastewater treatment plant: a focus on alternative biogas uses. *Bioresource Technology*, **270**, 138–146, https://doi.org/10.1016/J.BIORTECH.2018.09.005

Corominas L., Byrne D. M., Guest J. S., Hospido A., Roux P., Shaw A. and Short M. D. (2020). The application of life cycle assessment (LCA) to wastewater treatment: a best practice guide and critical review. *Water Research*, **184**, 116058, https://doi.org/10.1016/J.WATRES.2020.116058

Cuéllar-Franca R., García-Gutiérrez P., Dimitriou I., Elder R. H., Allen R. W. K. and Azapagic A. (2019). Utilising carbon dioxide for transport fuels: the economic and environmental sustainability of different Fischer–Tropsch process designs. *Applied Energy*, **253**, 113560, https://doi.org/10.1016/J.APENERGY.2019.113560

de Sampaio Lopes T. A., Matos Queiroz L. and Kiperstok A. (2014). Environmental performance of full-scale a wastewater treatment plant applying life cycle assessment. *Revista Ambiente e Agua*, **9**(3), 445–, https://doi.org/10.4136/1980-993X

Díaz I., Díaz-Curbelo A., Ignacio Matute K., Fdz-Polanco M. and Pérez-Elvira S. I. (2021). Influence of the operating conditions of the intermediate thermal hydrolysis on the energetic efficiency of the sludge treatment process. *Bioresource Technology*, **333**, 125114, https://doi.org/10.1016/J.BIORTECH.2021.125114

Dussan K. and Monaghan R. F. D. (2018). Integrated thermal conversion and anaerobic digestion for sludge management in wastewater treatment plants. *Waste and Biomass Valorization*, **9**(1), 65–85, https://doi.org/10.1007/s12649-016-9812-x

Ebrahimi M., Ramirez J. A., Outram J. G., Dunn K., Jensen P. D., O'Hara I. M. and Zhang Z. (2023). Effects of lignocellulosic biomass type on the economics of hydrothermal treatment of digested sludge for solid fuel and soil amendment applications. *Waste Management*, **156**, 55–65, https://doi.org/10.1016/J.WASMAN.2022.11.020

Edwards J., Othman M., Crossin E. and Burn S. (2017). Anaerobic co-digestion of municipal food waste and sewage sludge: a comparative life cycle assessment in the context of a waste service provision. *Bioresource Technology*, **223**, 237–249, https://doi.org/10.1016/J.BIORTECH.2016.10.044

Elginoz N., Atasoy M., Finnveden G. and Cetecioglu Z. (2020). Ex-ante life cycle assessment of volatile fatty acid production from dairy wastewater. *Journal of Cleaner Production*, **269**, 122267, https://doi.org/10.1016/J.JCLEPRO.2020.122267

El Ibrahimi M., Khay I., El Maakoul A. and Bakhouya M. (2021). Techno-economic and environmental assessment of anaerobic co-digestion plants under different energy scenarios: a case study in Morocco. *Energy Conversion and Management*, **245**, 114553, https://doi.org/10.1016/J.ENCONMAN.2021.114553

El-Qanni A., Alsayed M., Alsurakji I. H., Najjar M., Odeh D., Najjar S., Hmoudah M., Zubair M., Russo V. and Di Serio M. (2022). A technoeconomic assessment of biological sludge dewatering using a thermal rotary dryer: a case study of design applicability, economics, and managerial feasibility. Biomass Conversion and Biorefinery, https://doi.org/10.1007/s13399-022-03480-3

EPA (2016). Clean Watersheds Needs Survey 2012. Report to Congress. https://www.epa.gov/sites/default/files/2015-12/documents/cwns_2012_report_to_congress-508-opt.pdf

Estévez S., Feijoo G. and Moreira M. T. (2022a). Environmental synergies in decentralized wastewater treatment at a hotel resort. *Journal of Environmental Management*, **317**, 115392, https://doi.org/10.1016/J.JENVMAN.2022.115392

Estévez S., González-García S., Feijoo G. and Moreira M. T. (2022b). How decentralized treatment can contribute to the symbiosis between environmental protection and resource recovery. *Science of the Total Environment*, **812**, 151485, https://doi.org/10.1016/J.SCITOTENV.2021.151485

European Commission, Joint Research Centre and Institute for Environment and Sustainability (2011). International Reference Life Cycle Data System (ILCD) Handbook – Recommendations for Life Cycle Impact Assessment in the European context. First edition. EUR 24571 EN. Luxemburg: Publication Office of the European Union. ISBN: 978-92-79-17451-3.

European Environment Agency (2022). Beyond water quality sewage treatment in a circular economy.

Fernando-Foncillas C. and Varrone C. (2021). Potential of the sewage sludge valorization in Scandinavia by co-digestion with other organic wastes: a techno-economic assessment. *Journal of Cleaner Production*, **324**, 129239, https://doi.org/10.1016/J.JCLEPRO.2021.129239

Francini G., Lombardi L., Freire F., Pecorini I. and Marques P. (2019). Environmental and cost life cycle analysis of different recovery processes of organic fraction of municipal solid waste and sewage sludge. *Waste and Biomass Valorization*, **10**(12), 3613–3634, https://doi.org/10.1007/s12649-019-00687-w

Friedler E. and Pisanty E. (2006). Effects of design flow and treatment level on construction and operation costs of municipal wastewater treatment plants and their implications on policy making. *Water Research*, **40**(20), 3751–3758, https://doi.org/10.1016/j.watres.2006.08.015

Fuess L. T. and Zaiat M. (2018). Economics of anaerobic digestion for processing sugarcane vinasse: applying sensitivity analysis to increase process profitability in diversified biogas applications. *Process Safety and Environmental Protection*, **115**, 27–37, https://doi.org/10.1016/J.PSEP.2017.08.007

García-Gutiérrez P., Jacquemin J., McCrellis C., Dimitriou I., Taylor S. F. R., Hardacre C. and Allen R. W. K. (2016). Techno-economic feasibility of selective CO_2 capture processes from biogas streams using ionic liquids as physical absorbents. *Energy and Fuels*, **30**(6), 5052–5064, https://doi.org/10.1021/acs.energyfuels.6b00364

Garrido-Baserba M., Molinos-Senante M., Abelleira-Pereira J. M., Fdez-Güelfo L. A., Poch M. and Hernández-Sancho F. (2015). Selecting sewage sludge treatment alternatives in modern wastewater treatment plants using environmental decision support systems. *Journal of Cleaner Production*, **107**, 410–419, https://doi.org/10.1016/J.JCLEPRO.2014.11.021

Gholamian E., Mehr A. S., Yari M. and Carton J. G. (2023). Dynamic simulation and techno-economic assessment of hydrogen utilization in dual fuel (hydrogen/biogas) micro gas turbine systems for a wastewater treatment plant. *Process Safety and Environmental Protection*, **169**, 220–237, https://doi.org/10.1016/J.PSEP.2022.10.045

Gianico A., Bertanza G., Braguglia C. M., Canato M., Laera G., Heimersson S., Svanström M. and Mininni G. (2015). Upgrading a wastewater treatment plant with thermophilic digestion of thermally pre-treated secondary sludge: techno-economic and environmental assessment. *Journal of Cleaner Production*, **102**, 353–361, https://doi.org/10.1016/J.JCLEPRO.2015.04.051

Gourdet C., Girault R., Berthault S., Richard M., Tosoni J. and Pradel M. (2017). In quest of environmental hotspots of sewage sludge treatment combining anaerobic digestion and mechanical dewatering: a life cycle assessment approach. *Journal of Cleaner Production*, **143**, 1123–1136, https://doi.org/10.1016/J.JCLEPRO.2016.12.007

Harclerode M., Doody A., Brower A., Vila P., Ho J. and Evans P. J. (2020). Life cycle assessment and economic analysis of anaerobic membrane bioreactor whole-plant configurations for resource recovery from domestic wastewater. *Journal of Environmental Management*, **269**, 110720, https://doi.org/10.1016/J.JENVMAN.2020.110720

He C., Fang K., Wang W., Wang Q., Luo J., Ma J., Xue X., Gao F., Sun K., Liu M. and Wang K. (2022). Techno-economic feasibility of 'membrane-based pre-concentration + post-treatment' systems for municipal wastewater treatment and resource recovery. *Journal of Cleaner Production*, **375**, 134113, https://doi.org/10.1016/J.JCLEPRO.2022.134113

Heimersson S., Harder R., Peters G. M. and Svanström M. (2014). Including pathogen risk in life cycle assessment of wastewater management. 2. Quantitative comparison of pathogen risk to other impacts on human health. *Environmental Science and Technology*, **48**(16), 9446–9453, https://doi.org/10.1021/es501481m

Heimersson S., Svanström M., Cederberg C. and Peters G. (2017). Improved life cycle modelling of benefits from sewage sludge anaerobic digestion and land application. *Resources, Conservation and Recycling*, **122**, 126–134, https://doi.org/10.1016/J.RESCONREC.2017.01.016

Helmecke M., Fries E. and Schulte C. (2020). Regulating water reuse for agricultural irrigation: risks related to organic micro-contaminants. *Environmental Sciences Europe*, **32**(1), 4, https://doi.org/10.1186/s12302-019-0283-0

Hernández-Chover V., Bellver-Domingo Á and Hernández-Sancho F. (2018). Efficiency of wastewater treatment facilities: the influence of scale economies. *Journal of Environmental Management*, **228**, 77–84, https://doi.org/10.1016/J.JENVMAN.2018.09.014

IPCC (2021). Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.

ISO (2006a). EN ISO 14040:2006. Environmental management – Life cycle assessment – Principles and framework. http://www.cscses.com/uploads/2016328/20160328110518251825.pdf

ISO (2006b). EN ISO 14044:2006. Environmental management – Life cycle assessment – Requirements and guidelines. https://www.iso.org/standard/38498.html

Kong Z., Li L., Wu J., Wang T., Rong C., Luo Z., Pan Y., Li D., Li Y., Huang Y. and Li Y. Y. (2021). Evaluation of bioenergy recovery from the anaerobic treatment of municipal wastewater by a pilot-scale submerged anaerobic membrane bioreactor (AnMBR) at ambient temperature. *Bioresource Technology*, **339**, 125551, https://doi.org/10.1016/J.BIORTECH.2021.125551

Laitinen J., Moliis K. and Surakka M. (2017). Resource efficient wastewater treatment in a developing area – climate change impacts and economic feasibility. *Ecological Engineering*, **103**, 217–225, https://doi.org/10.1016/J.ECOLENG.2017.04.017

Lanko I., Flores L., Garfí M., Todt V., Posada J. A., Jenicek P. and Ferrer I. (2020). Life cycle assessment of the mesophilic, thermophilic, and temperature-phased anaerobic digestion of sewage sludge. *Water (Switzerland)*, **12**(11), 1–20, https://doi.org/10.3390/w12113140

Laurent A., Weidema B. P., Bare J., Liao X., Maia de Souza D., Pizzol M., Sala S., Schreiber H., Thonemann N. and Verones F. (2020). Methodological review and detailed guidance for the life cycle interpretation phase. *Journal of Industrial Ecology*, **24**(5), 986–1003, https://doi.org/10.1111/jiec.13012

Lehtoranta S., Malila R., Särkilahti M. and Viskari E. L. (2022). To separate or not? A comparison of wastewater management systems for the new city district of Hiedanranta, Finland. *Environmental Research*, **208**, 112764, https://doi.org/10.1016/J.ENVRES.2022.112764

Li H. and Feng K. (2018). Life cycle assessment of the environmental impacts and energy efficiency of an integration of sludge anaerobic digestion and pyrolysis. *Journal of Cleaner Production*, **195**, 476–485, https://doi.org/10.1016/J.JCLEPRO.2018.05.259

Li T., Zhang H., Liu Z., Ke Q. and Alting L. (2014). A system boundary identification method for life cycle assessment. *International Journal of Life Cycle Assessment*, **19**(3), 646–660, https://doi.org/10.1007/s11367-013-0654-5

Li H., Jin C. and Mundree S. (2017a). Hybrid environmental and economic assessment of four approaches recovering energy from sludge with variant organic contents. *Journal of Cleaner Production*, **153**, 131–138, https://doi.org/10.1016/J.JCLEPRO.2017.03.167

Li H., Jin C., Zhang Z., O'Hara I. and Mundree S. (2017b). Environmental and economic life cycle assessment of energy recovery from sewage sludge through different anaerobic digestion pathways. *Energy*, **126**, 649–657, https://doi.org/10.1016/J.ENERGY.2017.03.068

Li L., Wang X., Miao J., Abulimiti A., Jing X. and Ren N. (2022). Carbon neutrality of wastewater treatment – a systematic concept beyond the plant boundary. *Environmental Science and Ecotechnology*, **11**, 100180, https://doi.org/10.1016/J.ESE.2022.100180

Lijó L., Malamis S., González-García S., Moreira M. T., Fatone F. and Katsou E. (2017). Decentralised schemes for integrated management of wastewater and domestic organic waste: the case of a small community. *Journal of Environmental Management*, **203**, 732–740, https://doi.org/10.1016/J.JENVMAN.2016.11.053

Medina-Martos E., Istrate I. R., Villamil J. A., Gálvez-Martos J. L., Dufour J. and Mohedano Á.F. (2020). Techno-economic and life cycle assessment of an integrated hydrothermal carbonization system for sewage sludge. *Journal of Cleaner Production*, **277**, 122930, https://doi.org/10.1016/J.JCLEPRO.2020.122930

Mills N., Pearce P., Farrow J., Thorpe R. B. and Kirkby N. F. (2014). Environmental & economic life cycle assessment of current & future sewage sludge to energy technologies. *Waste Management*, **34**(1), 185–195, https://doi.org/10.1016/J.WASMAN.2013.08.024

Morelli B., Cashman S., Cissy Ma X., Garland J., Turgeon J., Fillmore L., Bless D. and Nye M. (2018). Effect of nutrient removal and resource recovery on life cycle cost and environmental impacts of a small-scale water resource recovery facility. *Sustainability (Switzerland)*, **10**(10), 3546, https://doi.org/10.3390/su10103546

Morelli B., Cashman S., Ma X., Turgeon J., Arden S. and Garland J. (2020). Environmental and cost benefits of co-digesting food waste at wastewater treatment facilities. *Water Science and Technology*, **82**(2), 227–241, https://doi.org/10.2166/wst.2020.104

Moretti C., Corona B., Edwards R., Junginger M., Moro A., Rocco M. and Shen L. (2020). Reviewing ISO compliant multifunctionality practices in environmental life cycle modeling. *Energies*, **13**(14), 3579. https://doi.org/10.3390/en13143579

Patel K. and Singh S. K. (2022). A life cycle approach to environmental assessment of wastewater and sludge treatment processes. *Water and Environmental Journal*, **36**(3), 412–424, https://doi.org/10.1111/wej.12774

Piao W., Kim Y., Kim H., Kim M. and Kim C. (2016). Life cycle assessment and economic efficiency analysis of integrated management of wastewater treatment plants. *Journal of Cleaner Production*, **113**, 325–337, https://doi.org/10.1016/J.JCLEPRO.2015.11.012

Postacchini M., Di Giuseppe E., Eusebi A. L., Pelagalli L., Darvini G., Cipolletta G. and Fatone F. (2022). Energy saving from small-sized urban contexts: integrated application into the domestic water cycle. *Renewable Energy*, **199**, 1300–1317, https://doi.org/10.1016/J.RENENE.2022.09.063

Prado L. O., Souza H. H. S., Chiquito G. M., Paulo P. L. and Boncz M. A. (2020). A comparison of different scenarios for on-site reuse of blackwater and kitchen waste using the life cycle assessment methodology. *Environmental Impact Assessment Review*, **82**, 106362, https://doi.org/10.1016/J.EIAR.2019.106362

Pretel R., Robles A., Ruano M. V., Seco A. and Ferrer J. (2013). Environmental impact of submerged anaerobic MBR (SAnMBR) technology used to treat urban wastewater at different temperatures. *Bioresource Technology*, **149**, 532–540, https://doi.org/10.1016/J.BIORTECH.2013.09.060

Pretel R., Robles A., Ruano M. V., Seco A. and Ferrer J. (2016). Economic and environmental sustainability of submerged anaerobic MBR-based (AnMBR-based) technology as compared to aerobic-based technologies for moderate-/high-loaded urban wastewater treatment. *Journal of Environmental Management*, **166**, 45–54, https://doi.org/10.1016/J.JENVMAN.2015.10.004

Purwanta Bayu A. I., Mellyanawaty M., Budiman A. and Budhijanto W. (2022). Techno-economic analysis of reactor types and biogas utilization schemes in thermophilic anaerobic digestion of sugarcane vinasse. *Renewable Energy*, **201**, 864–875, https://doi.org/10.1016/J.RENENE.2022.10.087

Ranganathan P. and Savithri S. (2019). Techno-economic analysis of microalgae-based liquid fuels production from wastewater via hydrothermal liquefaction and hydroprocessing. *Bioresource Technology*, **284**, 256–265, https://doi.org/10.1016/J.BIORTECH.2019.03.087

Remy C., Lesjean B. and Waschnewski J. (2013). Identifying energy and carbon footprint optimization potentials of a sludge treatment line with life cycle assessment. *Water Science and Technology*, **67**(1), 63–73, https://doi.org/10.2166/wst.2012.529

Righi S., Oliviero L., Pedrini M., Buscaroli A. and Della Casa C. (2013). Life cycle assessment of management systems for sewage sludge and food waste: centralized and decentralized approaches. *Journal of Cleaner Production*, **44**, 8–17, https://doi.org/10.1016/J.JCLEPRO.2012.12.004

Rodriguez-Garcia G., Frison N., Vázquez-Padín J. R., Hospido A., Garrido J. M., Fatone F., Bolzonella D., Moreira M. T. and Feijoo G. (2014). Life cycle assessment of nutrient removal technologies for the treatment of anaerobic digestion supernatant and its integration in a wastewater treatment plant. *Science of the Total Environment*, **490**, 871–879, https://doi.org/10.1016/J.SCITOTENV.2014.05.077

Rodríguez-Miranda J. P., García-Ubaque C. A. and Penagos-Londoño J. C. (2015). Analysis of the investment costs in municipal wastewater treatment plants in Cundinamarca. *Dyna*, **82**(192), 230–238, http://dx.doi.org/10.15446/dyna.v82n192.44699

Roldán M., Bouzas A., Seco A., Mena E., Mayor and Barat R. (2020). An integral approach to sludge handling in a WWTP operated for EBPR aiming phosphorus recovery: simulation of alternatives, LCA and LCC analyses. *Water Research*, **175**, 115647, https://doi.org/10.1016/J.WATRES.2020.115647

Rus E., Mills N., Shana A., Perrault A., Fountain P., Thorpe R. B., Ouki S. and Nilsen P. J. (2017). The intermediate thermal hydrolysis process: results from pilot testing and techno-economic assessment. *Water Practice and Technology*, **12**(2), 406–422, https://doi.org/10.2166/wpt.2017.031

Sala-Garrido R., Molinos-Senante M. and Hernández-Sancho F. (2012). How does seasonality affect water reuse possibilities? An efficiency and cost analysis. *Resources, Conservation and Recycling*, **58**, 125–131, https://doi.org/10.1016/J.RESCONREC.2011.11.002

Sanchez L., Vinardell S., Charreton J., Heran M. and Lesage G. (2023). Assessing the impact of granular anaerobic membrane bioreactor intensification on treatment performance, membrane fouling and economic balance. *Journal of Environmental Chemical Engineering*, **11**(2), 109369, https://doi.org/10.1016/J.JECE.2023.109369

Satayavibul A. and Ratanatamskul C. (2021). Life cycle assessment of a novel zero organic-waste model using the integrated anaerobic digester and oxidation-ditch membrane bioreactor for high-rise building application. *Waste and Biomass Valorization*, **12**(10), 5425–5436, https://doi.org/10.1007/s12649-021-01418-w

Shoener B. D., Zhong C., Greiner A. D., Khunjar W. O., Hong P. Y. and Guest J. S. (2016). Design of anaerobic membrane bioreactors for the valorization of dilute organic carbon waste streams. *Energy and Environmental Science*, **9**(3), 1102–1112, https://doi.org/10.1039/c5ee03715h

Sills D. L., Wade V. L. and DiStefano T. D. (2016). Comparative life cycle and technoeconomic assessment for energy recovery from dilute wastewater. *Environmental Engineering Science*, **33**(11), 861–872, https://doi.org/10.1089/ees.2016.0153

Singhirunnusorn W. and Stenstrom M. K. (2010). A critical analysis of economic factors for diverse wastewater treatment processes. *Journal of Environmental Engineering and Management*, **20**(4), 263–268, https://www.researchgate.net/publication/267382358

Singh E., Mishra R., Kumar A., Shukla S. K., Lo S. L. and Kumar S. (2022). Circular economy-based environmental management using biochar: driving towards sustainability. *Process Safety and Environmental Protection*, **163**, 585–600, https://doi.org/10.1016/J.PSEP.2022.05.056

Smith A. L., Stadler L. B., Cao L., Love N. G., Raskin L. and Skerlos S. J. (2014). Navigating wastewater energy recovery strategies: a life cycle comparison of anaerobic membrane bioreactor and conventional treatment systems with anaerobic digestion. *Environmental Science and Technology*, **48**(10), 5972–5981, https://doi.org/10.1021/es5006169

Stazi V. and Tomei M. C. (2018). Enhancing anaerobic treatment of domestic wastewater: state of the art, innovative technologies and future perspectives. *Science of the Total Environment*, **635**, 78–91. https://doi.org/10.1016/j.scitotenv.2018.04.071

Stazi V., Annesini M. C. and Tomei M. C. (2022). Anaerobic domestic wastewater treatment in a sequencing granular UASB bioreactor: feasibility study of the temperature effect on the process performance. *Journal of Environmental Chemical Engineering*, **10**(5), 108512, https://doi.org/10.1016/J.JECE.2022.108512

Tarpani R. R. Z. and Azapagic A. (2018). Life cycle costs of advanced treatment techniques for wastewater reuse and resource recovery from sewage sludge. *Journal of Cleaner Production*, **204**, 832–847, https://doi.org/10.1016/J.JCLEPRO.2018.08.300

Tchobanoglus G., Burton F. and Stensel H. D. (2003). Wastewater engineering: treatment and reuse. *Water Works Association Journal*, **95**(5), 201.

Thomsen M., Romeo D., Caro D., Seghetta M. and Cong R. G. (2018). Environmental-economic analysis of integrated organic waste and wastewater management systems: a case study from Aarhus City (Denmark). *Sustainability (Switzerland)*, **10**(10), 3742, https://doi.org/10.3390/su10103742

Tomei M. C., Bertanza G., Canato M., Heimersson S., Laera G. and Svanström M. (2016a). Techno-economic and environmental assessment of upgrading alternatives for sludge stabilization in municipal wastewater treatment plants. *Journal of Cleaner Production*, **112**, 3106–3115, https://doi.org/10.1016/J.JCLEPRO.2015.10.017

Tomei M. C., Mosca Angelucci D. and Levantesi C. (2016b). Two-stage anaerobic and post-aerobic mesophilic digestion of sewage sludge: analysis of process performance and hygienization potential. *Science of the Total Environment*, **545–546**, 453–464, https://doi.org/10.1016/J.SCITOTENV.2015.12.053

Tsagarakis K. P., Mara D. D. and Angelakis A. N. (2003). Application of cost criteria for selection of municipal wastewater treatment systems.

Tua C., Ficara E., Mezzanotte V. and Rigamonti L. (2021). Integration of a side-stream microalgae process into a municipal wastewater treatment plant: a life cycle analysis. *Journal of Environmental Management*, **279**, 111605, https://doi.org/10.1016/J.JENVMAN.2020.111605

Vinardell S., Astals S., Mata-Alvarez J. and Dosta J. (2020). Techno-economic analysis of combining forward osmosis-reverse osmosis and anaerobic membrane bioreactor technologies for municipal wastewater treatment and water production. *Bioresource Technology*, **297**, 122395, https://doi.org/10.1016/J.BIORTECH.2019.122395

Vinardell S., Dosta J., Mata-Alvarez J. and Astals S. (2021). Unravelling the economics behind mainstream anaerobic membrane bioreactor application under different plant layouts. *Bioresource Technology*, **319**, 124170, https://doi.org/10.1016/J.BIORTECH.2020.124170

Vinardell S., Blandin G., Ferrari F., Lesage G., Mata-Alvarez J., Dosta J. and Astals S. (2022). Techno-economic analysis of forward osmosis pre-concentration before an anaerobic membrane bioreactor: impact of draw solute and membrane material. *Journal of Cleaner Production*, **356**, 131776, https://doi.org/10.1016/J.JCLEPRO.2022.131776

Vogli L., Macrelli S., Marazza D., Galletti P., Torri C., Samorì C. and Righi S. (2020). Life cycle assessment and energy balance of a novel polyhydroxyalkanoates production process with mixed microbial cultures fed on pyrolytic products of wastewater treatment sludge. *Energies*, **13**(11), 2706, https://doi.org/10.3390/en13112706

Volpi M. P. C., Fuess L. T. and Moraes B. S. (2023). Economic performance of biogas production and use from residues co-digestion in integrated 1G2G sugarcane biorefineries: better electricity or biomethane? *Energy Conversion and Management*, **277**, 116673, https://doi.org/10.1016/J.ENCONMAN.2023.116673

Wang S., Yu S., Lu Q., Liao Y., Li H., Sun L., Wang H. and Zhang Y. (2020). Development of an alkaline/acid pre-treatment and anaerobic digestion (APAD) process for methane generation from waste activated sludge. *Science of the Total Environment*, **708**, 134564, https://doi.org/10.1016/J.SCITOTENV.2019.134564

Xu C., Chen W. and Hong J. (2014). Life-cycle environmental and economic assessment of sewage sludge treatment in China. *Journal of Cleaner Production*, **67**, 79–87, https://doi.org/10.1016/J.JCLEPRO.2013.12.002

Yoshida H., ten Hoeve M., Christensen T. H., Bruun S., Jensen L. S. and Scheutz C. (2018). Life cycle assessment of sewage sludge management options including long-term impacts after land application. *Journal of Cleaner Production*, **174**, 538–547, https://doi.org/10.1016/J.JCLEPRO.2017.10.175

Zieliński M., Kazimierowicz J. and Dębowski M. (2023). Advantages and limitations of anaerobic wastewater treatment – technological basics, development directions, and technological innovations. *Energies*, **16**(1), 83. https://doi.org/10.3390/en16010083