Engineering 59 (2026) 297–311

Contents lists available at ScienceDirect

Engineering

journal homepage: www.elsevier.com/locate/eng

Research
Environmental Engineering—Review

Techno–Economic and Environmental Impact Assessment of Membrane Bioreactors for Wastewater Treatment: A Review

Tingwei Gao [a,b,c], Yana Jin [b,*], Kang Xiao [a,*]

[a] Beijing Yanshan Earth Critical Zone National Research Station, College of Resources and Environment, University of Chinese Academy of Sciences, Beijing 101408, China
[b] College of Environmental Sciences and Engineering, Peking University, Beijing 100871, China
[c] Binzhou Institute of Technology, Weiqiao-UCAS Science and Technology Park, Binzhou 256606, China

ARTICLE INFO

Article history:
Received 26 April 2024
Revised 8 April 2025
Accepted 7 May 2025
Available online 12 May 2025

Keywords:
Membrane bioreactor
Economic analysis
Environmental impact
Carbon emission
Sustainability analysis

ABSTRACT

Despite the advantages of high effluent quality and small footprint, membrane bioreactor (MBR) technology faces challenges in sustainable development due to energy consumption and membrane fouling. Weighing the advantages and disadvantages of MBRs requires a comprehensive assessment from techno–economic–environmental perspectives. In this paper, we reviewed the related research on MBRs from three aspects: economic cost analysis, environmental impact assessment, and comprehensive techno–economic–environmental assessment. The aim of this paper is to understand the sustainable development performance of MBRs, and to review the current status of the application of multiple techno–economic–environmental assessment methods in the field of wastewater treatment. The currently available results of the economic cost analysis of MBRs showed that the operating cost and energy consumption of MBRs are higher than those of other wastewater treatment processes if MBRs' potential benefit of smaller footprint is not taken into account. The results of the environmental impact assessment showed that MBRs have a positive environmental impact due to high quality effluent, although global warming potential limits the sustainability of MBRs to some extent. Combined techno–economic–environmental assessment showed that MBRs are economically feasible and technically efficient, while their sustainability is controversial. Given the rapid development of MBR technology, these results may evolve as new advancements are made. In addition, there is room for improvement in the existing literature regarding the reliability and comparability of results, as well as the applicability of the methods, particularly in defining the accounting scope, clarifying model assumptions, and considering discounting.

1. Introduction

Wastewater treatment and resource recovery is an effective way to cope with the water crisis. The membrane bioreactor (MBR) formed by the combination of membrane separation and biological treatment technology, has been applied worldwide due to the advantages of good effluent quality, small footprint, and so forth [1–4].

However, as the applications of MBRs are gradually expanding, the drawbacks of the MBR process are also noteworthy. On the one hand, membrane fouling leads to a decrease in membrane flux or an increase in transmembrane pressure, which causes a series of problems such as decreased treatment capacity, poor operational stability, increased maintenance costs, and shortened lifespan [5]. On the other hand, high energy consumption and high operating costs limit the widespread application of MBRs. Energy is consumed for membrane scouring, biological aeration, membrane pumping, as well as cleaning of the fouled membrane [6]. Periodic recovery cleaning and membrane module replacement also increase the operating costs of MBRs.

These challenges have attracted extensive attention from researchers into the techno–economics and sustainability of MBR [2,7–10]. Can the advantages of MBR outweigh its disadvantages? How about the sustainability of MBR compared to conventional wastewater treatment processes? It is essential for a comprehensive assessment of MBRs from techno–economic–environmental points of view, which requires not only an understanding of the economic costs and environmental impacts of MBRs but also a comprehensive analysis of the trade-offs between the two.

* Corresponding authors.
E-mail addresses: jin.yana@pku.edu.cn (Y. Jin), kxiao@ucas.ac.cn (K. Xiao).

https://doi.org/10.1016/j.eng.2025.05.001

Existing literature has discussed these aspects, but there are some limitations in research design, methodology, and results. The objectives of this study are to clarify the current status of the application of techno–economic–environmental assessment in MBRs for wastewater treatment, and to better understand the sustainable development performance of MBRs. The MBRs reviewed in this paper include both real operating MBR wastewater treatment plants (WWTPs) and MBR processes simulated through experimentation or modeling.

2. Methods

2.1. Description of the study scope

This paper reviewed the research on techno–economic–environmental assessment in MBRs for wastewater treatment from three aspects: economic cost analysis, environmental impact assessment, and comprehensive techno–economic–environmental assessment. Fig. 1 presents the scope of this study.

Studies on techno–economic–environmental assessment can be divided into three groups. One category focuses on the techno–economics of wastewater treatment processes. Based on actual or experimentally simulated wastewater treatment processes, the energy consumption and cost of treating a unit of wastewater or a unit of pollutant are calculated for different scenarios (e.g., different process flows, different sizes, and different influent patterns). The second category is to assess the environmental impacts of wastewater treatment processes. The environmental impacts of the processes (e.g., global warming potential (GWP)) are mainly assessed using life cycle assessment (LCA). The third category aims to comprehensively assess the sustainability performance of wastewater treatment processes. A common type of study is to conduct economic cost assessment (i.e., the first category) and environmental impact assessment (i.e., the second category) separately, and analyze the sustainability of a wastewater treatment process by presenting the results of these two parts directly. Since this approach cannot quantitatively compare the environmental impacts with the economic situation, it is difficult to conclude which scenario or process is better from environmental and economic perspectives. Therefore, an increasing number of studies applied some quantitative models for integrated sustainability assessment of wastewater treatment processes.

Existing integrated assessment methods include three types. The first method is cost–benefit analysis (CBA), which is an analytical approach based on economic cost assessment and environmental impact assessment. The method requires further monetization of environmental impacts (e.g., greenhouse gas (GHG) emissions calculated through LCA), and then comparing them with costs to obtain net benefits, which evaluates the feasibility of the wastewater treatment process. The second method is data envelopment analysis (DEA), which calculates the technical efficiency based on the inputs (e.g., costs and energy consumption) and outputs (e.g., the volume of wastewater treated) of the wastewater treatment process. This method generally uses inputs and outputs that do not require additional calculations, instead of using the results of the economic cost assessment and environmental impact assessment. For instance, DEA often uses operating costs as one of the inputs, rather than total costs or life cycle costs as inputs. Multi-criteria decision analysis (MCDA), the third approach, calculates a composite index to comprehensively evaluate the wastewater treatment process by assigning weights to indicators from different aspects (e.g., economic, technical, and environmental aspects).

2.2. Bibliometric information

This paper collected and analyzed the literature on the economic cost analysis of MBRs, environmental impact assessment of MBRs, and integrated techno–economic–environmental assessment of MBRs using bibliometric methods based on the Web of Science (WoS) database. To improve the quality of the search results, only articles from the WoS Core Collection database were selected, and the literature types were restricted to articles and reviews. As of September 2, 2024, the total number of search results obtained was 294, of which 38 were reviews and 256 were research articles, with a total of 10 389 citations. This means that the average number of citations per article was 35 (the median number of citations was 16). The search formula is shown in Table S1 in Appendix A.

Literature related to the topic of this paper was first published in *Water Science and Technology*, where Adham et al. [11] investigated the feasibility of the MBR process for wastewater reuse. Through a literature review, a global survey, and a preliminary cost estimation, they demonstrated that the MBR process is cost competitive with other conventional wastewater treatment processes.

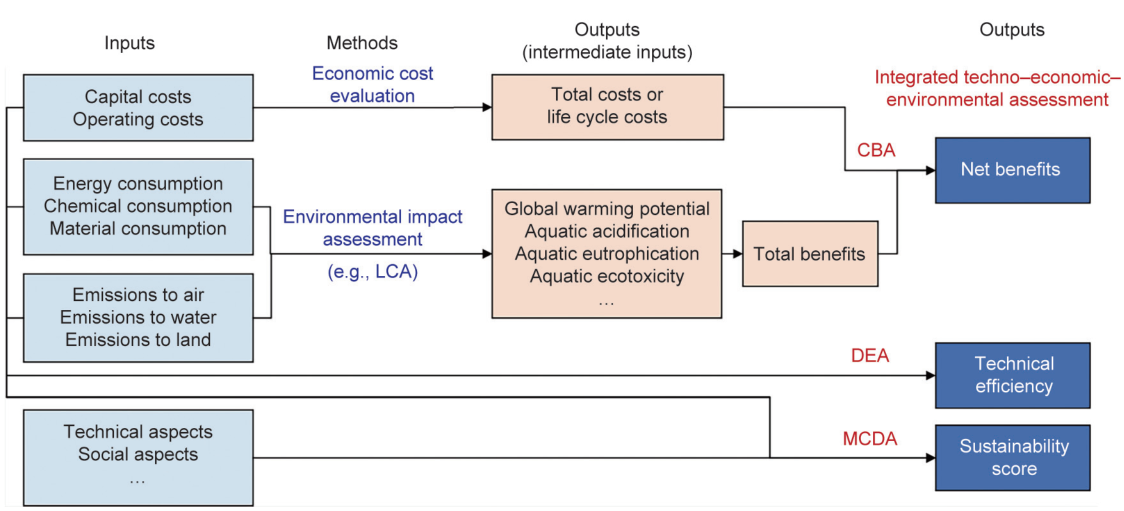


Fig. 1. Scope of the study on techno–economic–environmental assessment. LCA: life cycle assessment; CBA: cost-benefit analysis; DEA: data envelopment analysis; MCDA: multi-criteria decision analysis.

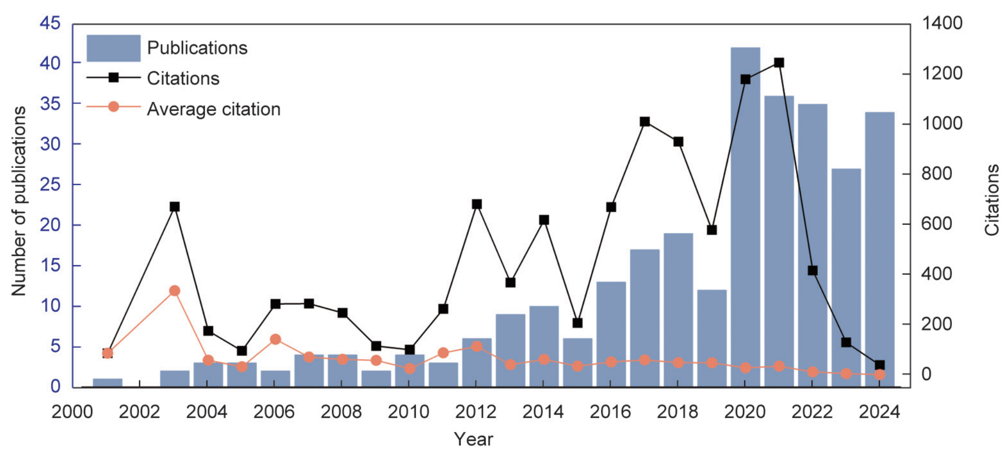
There was a general upward trend in publications in the selected area, with a significant increase since 2013, indicating the attention given by academics to economic analysis, environmental impact assessment, or techno–economic–environmental assessment of MBRs (Fig. 2(a)).

There was no lack of highly cited literature, both in early and recent publications [8,10,12–18]. The most highly cited article (with 668 citations) was a review by Wei et al. [18] in *Water Research*. The argument at the time was that the high cost of MBRs was a barrier to the widespread use in large-scale WWTPs and that the choice of sludge reduction strategies for practical application should be based on cost analyses and environmental impact assessments [18].

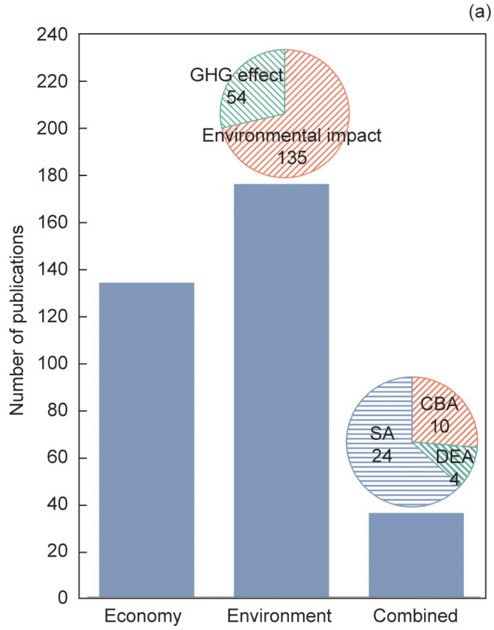
The literature was searched separately for three topics (Fig. 2(b)), and the search formula is detailed in Table S1. There were 134 studies conducting economic analyses of MBRs. The relatively largest number of studies (i.e., 176 studies) dealt with environmental impact assessment (including GHG accounting), with 54 studies focusing on GHG accounting. There were relatively few studies (i.e., 36 studies) on integrated techno–economic–environmental assessment, of which 10 were on CBA, 4 on DEA, and 24 on integrated sustainability assessment (including assessments from different perspectives (e.g., economic, environmental, and technological) and MCDA).

Keywords appearing frequently in the literature (Fig. 2(c)) include "membrane bioreactor," "wastewater treatment," "life cycle assessment," "membrane fouling," and "energy consumption." This indicates that membrane fouling, energy consumption, water reuse, and LCA were popular topics in the study of techno–economic–environmental assessment of MBRs applied in wastewater treatment.

Fig. 3(a) shows the top 20 journals in terms of citations. The most cited journal was *Water Research*, which published 16 relevant articles that were cited 1525 times, with an average of 95 citations per article. Other highly cited journals were *Bioresource Technology*, *Desalination*, *Journal of Cleaner Production*, and *Journal of Membrane Science*, all of which are high level journals in the field of membrane-based water/wastewater treatment. The articles in these highly cited journals were focused primarily on the performance of various processes incorporating MBRs in terms of feasibility and sustainability, measured from one or more perspectives, such as cost, energy consumption (or recovery), and environmental impacts. Fig. 3(b) shows the top 20 countries in terms of citations. China, the United States, and Spain were the



(a)



(b)

Hot keywords	Frequency
Membrane bioreactor	48
Wastewater treatment	45
Life cycle assessment	38
Membrane fouling	29
MBR	16
Sustainability	13
Anaerobic digestion	12
Energy consumption	12
Wastewater	12
Water reuse	12
Anaerobic membrane bioreactor (AnMBR)	11
Anaerobic membrane bioreaction	10
Cost analysis	10
Membrane bioreactors	9
Forward osmosis	8
Life cycle assessment (LCA)	8
Membrane bioreactor (MBR)	7
Municipal wastewater treatment	7
Ultrafiltration	7
Activated sludge	6

(c)

Fig. 2. (a) Time trend of publications, citations, and average citation; (b) the number of publications on different topics; (c) the most popular keywords. SA: sustainability analysis.

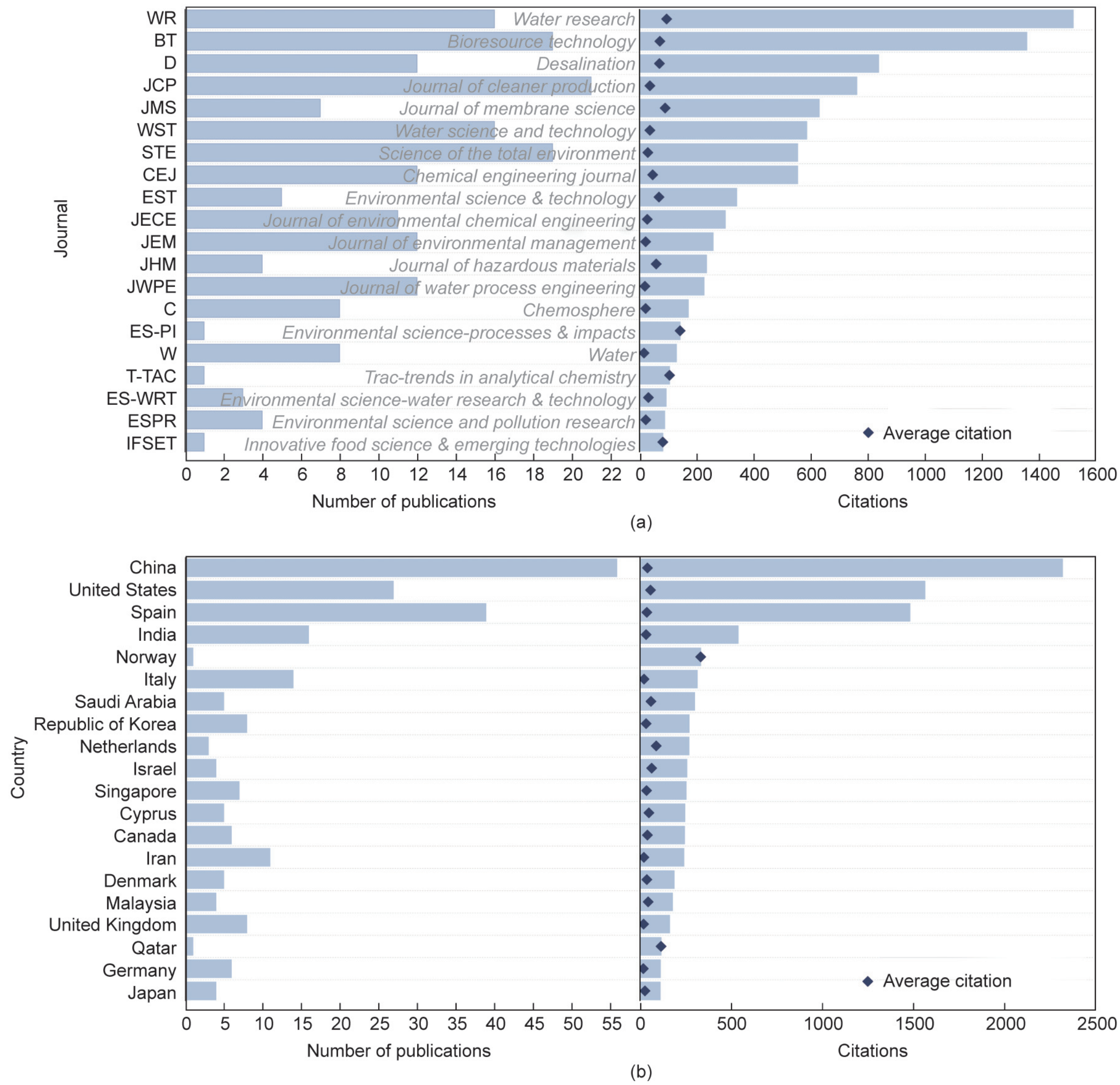


Fig. 3. (a) The most popular journals by publications and citations; (b) the countries with the most publications and citations.

top three in terms of number of publications and total citations. Norway's average citation per article was far ahead of the other countries, thanks to the review article by Krzeminski et al. [8] (with a high number of citations of 334) on MBRs in terms of energy saving, membrane fouling control, and LCA.

3. Economic cost evaluation

Since the 1980s, MBRs have been widely applied worldwide due to their superior effluent quality and space savings [19]. However, MBRs are still considered an expensive wastewater treatment process option due to the membrane cost and energy consumption. Many studies have carried out techno-economic evaluations of MBRs [9,20–22].

3.1. Cost components

The cost of MBRs usually includes capital and operating costs [23,24]. Capital costs include plant construction, pipeline network, and non-engineering costs. Tanks, pipelines, membrane modules, and other equipment typically account for about 40%, 10%, 20%, and 30% of the plant construction costs, respectively [4,10]. More than half of the plant construction cost comes from the membrane treatment unit (55%–85%) [10]. Non-engineering costs include management fees, interest expenses, financing costs, and taxes. Most studies [25,26] have not considered land cost in the capital cost because of the difficulty of evaluation.

The operating costs of MBRs mainly include energy consumption (40%–60%), chemical consumption (10%–30%), sludge disposal costs (5%–19%), and personnel costs (5%–28%) [7,10,22,25,27]. The total energy consumption of MBR WWTPs consists of aeration (membrane scouring and biological aeration), liquid pumping (lifting and recirculation), sludge mixing, and so forth. Aeration accounts for about 60% of the energy consumption of MBRs [4]. The chemical consumption of MBRs is mainly attributed to the addition of external carbon sources, phosphorus removal, disinfection, sludge conditioning, and membrane fouling control (membrane chemical cleaning and mixed liquor conditioning) [10]. Maintenance costs are often discussed together with operating costs, which are called operating and maintenance (O&M) costs.

MBRs also have membrane replacement costs or membrane depreciation costs. As membrane modules need to be replaced, some studies regarded membrane replacement costs as operating costs [28]. Membrane modules also have the characteristic of a fixed asset that wears out over the years. Some studies regarded membrane depreciation costs as capital costs [22,26,29]. For clarity, this review discussed membrane replacement/depreciation costs separately from capital and operating costs.

3.2. Methods of cost evaluation

Whether MBRs or other wastewater treatment processes, costing requires the sum of capital costs, operating costs, and other costs (e.g., membrane depreciation costs). Since capital and operating costs do not occur at the same time, the costs need to be discounted to the same time before further summing. Some studies [30–33] directly added up the capital cost and the annual operating cost without taking into account the time value of money, which is contrary to objective economic laws. Currently, there are two main costing methods: calculating annual costs and calculating life-cycle costs (or the net present value).

For the first method, capital costs are depreciated and amortized each year. The annual depreciated capital cost is added to the annual operating cost to obtain the total annual cost [25,34–36], as shown in Eq. (1):

$$\mathrm{TC} = \alpha\mathrm{CC} + \mathrm{OC} + \beta\mathrm{RC} \tag{1}$$

$$\mathrm{CRF} = [i \times (1+i)^T]/[(1+i)^T - 1] \tag{2}$$

The annual total cost (TC) of an MBR is the linear sum of the annual amortized capital cost (CC), the annual operating cost (OC), and the annual amortized membrane replacement cost (RC) [25,34]. α and β are the capital recovery factor (CRF) of fixed assets and membrane modules of WWTPs, respectively. The CRF is calculated from the discount rate i and the asset lifetime T (Eq. (2)). The lifetime of fixed assets (excluding membrane modules) in capital costs is 10–40 years [17,22,37], while the membrane lifetime t_m is usually 5–10 years [35,38]. The discount rate is assumed to be 3%–10% [17,22,39].

The second method is to sum the costs discounted to the present value in each period to obtain the net present value (NPV) of the lifecycle costs [7,22,40–43], as shown in the following Eq. (3). The term t represents the number of years an MBR has been in operation.

$$\mathrm{NPV} = \mathrm{CC} + \sum_{t=1}^{T} \frac{\mathrm{OC}_t}{(1+i)^t} + \sum_{t=t_m}^{T-t_m} \frac{\mathrm{RC}_t}{(1+i)^t} \tag{3}$$

3.3. Results of cost evaluation

Fig. 4 shows the cost and energy consumption of MBRs, with more detailed cost data provided in Table S2 in Appendix A. To ensure comparability across studies, Fig. 4 includes only those that compare the costs of MBRs with other wastewater treatment processes, excluding studies that focus solely on the economics of MBRs. Among the reviewed literature, 40% are based on data from real WWTPs, while the remaining 60% rely on simulation or experimental data.

The operating costs and energy consumption of MBRs varied with wastewater quality, treatment scale, and other factors. As a result, there were large differences in the operating costs reported in different studies. Compared to conventional wastewater treatment processes, MBRs had higher operating costs and energy consumption [42,44,45]. The operating costs of conventional activated sludge (CAS) processes were 0.02–0.40 USD·m^{-3} (0.15–3.0 CNY·m^{-3}), while those of MBRs were 0.09–0.45 USD·m^{-3} (0.66–3.3 CNY·m^{-3}) [39,41,42,44–46]. The energy consumption of CAS processes was 0.3–0.64 kW·h·m^{-3}, while that of MBRs was 0.4–1.15 kW·h·m^{-3} [8,10,27,47]. Energy consumption is the largest part of operating costs, with aeration accounting for about 60% of total energy consumption, and membrane aeration accounting for more than 60% of total aeration requirements [48]. Therefore, refined design and optimization of aeration are conducive to energy saving, which can improve MBRs' cost competitiveness [21,48–50]. Adopting renewable energy sources and improving the efficiency of energy recovery are also effective ways to improve the economic sustainability of MBRs [51]. In addition, membrane fouling, as a concomitant problem during MBRs' operation, increases the energy consumption for filtration and cleaning, which is a bottleneck for the development of MBRs [52].

The capital cost of MBRs was comparable to or higher than that of CAS processes, but capital costs here usually did not include land costs. The average land occupation area of an MBR for municipal wastewater treatment is 0.8 m_2·(m_3·d^{-1})$^{-1}$, smaller than that of CAS (~1 m_2·(m^{-3}·d^{-1})$^{-1}$ [10]. When land costs are taken into account, MBRs would be competitive in terms of capital costs due to the advantage of the small footprint. Overall, the total cost of an MBR was slightly higher than other wastewater treatment processes (average total cost: MBR vs CAS = 0.25 vs 0.19 USD·m^{-3}). MBRs can be cost competitive in certain applications, such as areas with stringent effluent standards, high capital costs, high concentrations of organic influent, or influent from waste leachate [3,10]. The cost of an MBR is influenced by factors such as temperature, membrane flux, pretreatment method, membrane cost, membrane lifetime, and so forth. In general, improved membrane performance, longer membrane lifetime, lower membrane price, and increased membrane flux all reduce the life cycle cost of MBRs [46,53,54]. Costs are also related to influent flow rate (or treatment capacity). Several studies have developed capacity-based cost functions to predict the cost of MBRs [22,43,55].

4. Environmental impact assessment

Wastewater treatment aims to reduce wastewater pollution caused by human activities and to minimize the negative impacts of wastewater on environmental quality and human health [56]. However, in addition to the positive environmental impacts associated with the removal of pollutants, wastewater treatment processes also have negative environmental impacts due to material and energy consumption, GHG emissions, and residual sludge disposal [57]. MBRs are capable of achieving high effluent quality and saving land area but may come with environmental costs such as exacerbating global warming. Therefore, a comprehensive assessment of the environmental impacts of wastewater treatment processes is needed to promote sustainable development. Several studies [9,31,58–62]. have evaluated the environmental impacts of MBRs using LCA, including primary energy consumption, GWP, carcinogens, aquatic acidification, aquatic ecotoxicity, eutrophication of water bodies, and so forth. In the context of an increasingly critical climate change situation, some studies [17,53] focused on the carbon emissions (or GWP) of MBRs.

4.1. Overall assessment of environmental impact

LCA, which quantifies environmental impacts by collecting inputs and outputs throughout the life cycle, has been widely applied to the environmental impact assessment of WWTPs [63–66]. Table S3 in Appendix A shows some of the studies on LCA of WWTPs. LCA consists of four steps: system boundary determination, life cycle inventory analysis, life cycle impact assessment (LCIA), and result resolution.

As of September 2, 2024, there were 88 articles (including reviews) referring to LCA research on MBRs. Table 1 [9,17,31,53,58–62,67–87] shows the LCA studies of the MBR, excluding those without full details of the LCA analysis. Table 1

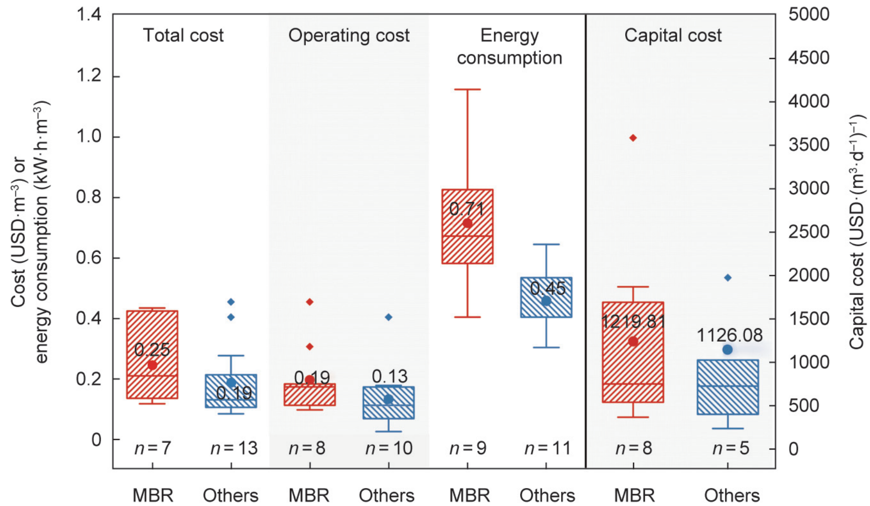


Fig. 4. Cost estimations of MBRs and other wastewater treatment processes (mainly referring to the CAS process). The data are sourced from all the references listed in Table S2.

and Table S3 show that the LCA studies of MBRs and other wastewater treatment processes usually set the wastewater treatment volume as the function unit [34,53,59,88]. Some studies used the removal of pollutants (e.g., chemical oxygen demand (COD) and total phosphorus (TP)) per unit mass as a functional unit [89,90]. Due to the low life cycle impact and the difficulty of obtaining inventory data in the construction and demolition phases of WWTPs, most studies [60,91] only considered the operational phase. Life cycle inventory provides a database for subsequent steps by quantifying the material/energy inputs and outputs of a given process system throughout its life cycle. Inventory data are usually classified and indexed using models (e.g., the tool for the reduction and assessment of chemical and other environmental impacts (TRACI)) to assess environmental impacts such as primary energy consumption and GWP.

Various LCIA models consider different categories of environmental impacts. For WWTPs, GWP, aquatic acidification, aquatic ecotoxicity, aquatic eutrophication, respiratory effects, carcinogens, and non-carcinogens are major categories of the overall environmental impacts [17,31,67,68]. The main sources of these environmental impacts are electricity consumption, sludge disposal, material consumption, and direct emissions [31]. LCA studies of MBRs have shown that MBRs have a relatively high energy consumption (0.47–1.69 $kW·h·m^{-3}$) due to the requirements for membrane fouling mitigation (e.g., aeration flushing and chemical cleaning) [31,67]. Since energy consumption contributes more than 60% to GWP, aquatic acidification, and respiratory effects, MBRs are associated with a high level of these environmental impacts [31,68]. The environmental impacts associated with the production, use, and replacement of membrane materials (e.g., polyvinylidene fluoride (PVDF) and polyvinyl chloride (PVC)), are also noteworthy for MBRs. The environmental impact categories to which membrane materials contribute most over the life cycle are carcinogens and non-carcinogens. The carcinogen category can be reduced from 0.0274 to 0.0195 kg C_2H_3Cl equivalent by extending the membrane life from 4 to 8 years [31]. For both CAS and MBR processes, sludge disposal was reported to contribute more than 50% to aquatic ecotoxicity [9,31]. Environmentally friendly sludge treatment and disposal technologies are thus important for reducing the negative environmental impact of WWTPs. Despite the environmental burden of MBRs in terms of environmental impacts such as GWP, MBRs have a very low impact on the eutrophication of oceans and freshwaters due to high effluent quality. Optimizing energy use, using renewable energy sources or implementing energy recovery strategies, optimizing sludge disposal options, and extending membrane lifespan (e.g., through effective membrane fouling control or the use of new membrane materials) can significantly reduce the life cycle environmental impacts of MBRs [31,92].

Retrofitting the CAS process toward the MBR process for wastewater treatment can increase the life-cycle fossil fuel consumption, photochemical formation, acidification, and respiration impacts [69]. Shifting energy use to a power mix with a higher share of renewable energy (e.g., wind or lignocellulosic biomass power) can reduce the life-cycle environmental impacts of MBRs [69,70]. In addition to the energy mix, climatic conditions, the size of the decentralized system and the amount of water reused can also affect the environmental impacts of MBRs [53,71]. The final use of the effluent and sludge determines the potential environmental impact of MBR applications [72]. For example, the nutrient-rich effluent from anaerobic MBRs (AnMBRs) can be used directly in agricultural production to compensate for the nutrients required by crops and therefore has a low environmental impact [72]. As AnMBRs can recover methane, many studies [17,52,60] have analyzed the environmental impact of AnMBRs using scenario modeling and LCA to maximize energy recovery and dissolved methane recovery.

Overall, there has been an increase in the number of LCA studies of MBRs in recent years, but the limitations of the studies have become increasingly apparent. Different studies defined functional units and impact categories differently, and thus it is difficult to make comparisons between them. Some studies did not detail the basic LCA information such as the scope, functional units, and methods of environmental impact assessment, which would affect the reliability and reproducibility of the results. Most studies [17,31,53,58–62] only considered the environmental impacts during the operational phase, while often neglecting the decommissioning phase. To ensure consistency, reliability, and reproducibility of the results, it is recommended to consider a more standardized LCA implementation [93].

4.2. GWP assessment

GWP, one of the environmental impacts of greatest concern in environmental impact assessment, refers to the direct or indirect emissions of GHGs. The main GHGs emitted from wastewater

Table 1
Main characteristics of LCA related literature for MBRs.

Reference	Country	Function unit	System boundaries	Sludge disposal	LCIA model	Software
[9]	Spain	1 m^3 of treated wastewater	Construction, operation, and demolishment	Agricultural application, landfill, and incineration	CML 2000	DESASS, BNRM2, and SimaPro
[17]	USA	5 MGD of treated wastewater	Operation	Landfill, land application, and incineration	TRACI	TRACI
[31]	China	1 m^3 of treated wastewater	Operation	—	CML 2001	GaBi
[53]	USA	1 m^3 of treated wastewater	Operation	Landfill	TRACI	OpenLCA
[58]	Greece	1 m^3 of treated wastewater	Operation	Landfill	CML 2001	OpenLCA
[59]	USA	5 MGD of treated wastewater	Operation	Landfill, anaerobic digestion, and composting	TRACI	TRACI
[60]	USA	5 MGD of treated wastewater	Operation	Land application	TRACI	SimaPro
[61]	Cyprus	1 m^3 of treated wastewater	Construction and operation	Landfill	IPCC 2013, ReCiPe	SimaPro
[62]	Spain	1 m^3 of treated wastewater	Operation	—	ReCiPe	SimaPro
[67]	India	1 m^3 of treated wastewater	Operation	Landfill	CML 2001	GaBi
[68]	USA	1 m^3 water used for outdoor irrigation and/or toilet flushing	Construction and operation	—	TRACI	SimaPro
[69]	China	1 kg preserved plum	Construction and operation	Landfill	openLCA	TRACI and ReCiPe
[70]	Türkiye	1 m^3 of treated wastewater	Construction and operation	Incineration	Ecoinvent	SimaPro
[71]	Canada	The annual treatment of greywater generated per person	Construction, operation, and demolishment	Landfill	TRACI	OpenLCA
[72]	China	0.22 $m^3 \cdot s^{-1}$ of treated wastewater	Construction and operation	Land application and incineration	TRACI	SimaPro
[73]	Iran	5 MGD of treated wastewater	Operation	Land application	ReCiPe	—
[74]	Iran	1 $m^3 \cdot d^{-1}$ of WWTP's secondary effluent	Construction and operation	—	Eco-Indicator 99	—
[75]	Iran	1 $m^3 \cdot d^{-1}$ of WWTP effluent	Construction and operation	Agricultural application	Impact 2002+, Eco-indicator 99,CML 2001, IPCC	SimaPro
[76]	Greece	1 kg of produced canned peaches	Operation	Landfill and composting	ReCiPe	GaBi
[77]	Italy	1 m^3 of treated wastewater	Operation	Landfill	ReCiPe	Umberto LCA
[78]	USA	1 m^3 of producing water	Operation	Landfill	ReCiPe	SimaPro
[79]	Spain	1 m^3 of treated wastewater	Operation	Agricultural application	CML	SimaPro
[80]	Iran	1 m^3 of treated wastewater	Operation	Sanitary landfill, biowaste, and incineration	ReCiPe	—
[81]	France	1.92 million $m^3 \cdot year^{-1}$ of treated wastewater	Construction and operation	—	IPCC	TEAM and DEAM
[82]	Spain	1 m^3 of treated wastewater	Construction, operation, and demolishment	Landfilling, land application, and incineration	CML 2000	DESASS, BNRM2, and SimaPro
[83]	Republic of Korea	100 000 $m^3 \cdot d^{-1}$ of wastewater	Construction and operation	Landfill	TOTAL	TOTAL
[84]	Thailand	1 kg of sludge produced	Operation	—	SimaPro	CML–IA
[85]	Thailand	2.02 $m^3 \cdot d^{-1}$ of combined wastewater	Operation	—	—	ReCiPe
[86]	Denmark	1000 m^3 of treated wastewater	Operation	Agricultural application	EASETECH	EASETECH
[87]	Australia	1 mL of recycled water	Construction and operation	Landfill	MIET	GaBi

MGD: million gallon·day^{-1}; IPCC: Intergovernmental Panel on Climate Change; CML: the methodology developed by the Centre of Environmental Science of Leiden University; TRACI: the tool for the reduction and assessment of chemical and other environmental impacts; ReCiPe: a method for LCIA, which was first developed in 2008 through cooperation between Rijksinstituut voor Volksgezondheid en Milieu (RIVM), Radboud University Nijmegen, Leiden University, and PRé Sustainability; TOTAL: a software developed by the Korea Environmental Industry and Technology Institute (KEITI); EASETECH: environmental assessment system for environmental technologies; and MIET: the missing inventory estimation tool approach.

treatment processes are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Based on their contribution to global warming, the different GHGs are converted into carbon dioxide equivalents (CO_2e), which are used to measure the GWP.

According to the IPCC guidelines, GHG emissions (i.e., carbon emissions) from wastewater treatment processes can be categorized into Scopes I, II, and III. Scope I carbon emissions are the direct emissions of GHGs into the atmosphere from wastewater treatment processes, including CO_2 from aerobic, anaerobic, and anoxic processes, CH_4 from anaerobic processes and sludge treatment, N_2O from nitrification and denitrification processes and dissolved CH_4 in the final effluent water. Scope II carbon emissions are indirect carbon emissions from electricity or heat consumption in wastewater treatment processes. Aeration energy consumption is the primary source of indirect carbon emissions. Scope III carbon emissions are indirect carbon emissions other than those in Scope II, from the manufacture and transportation of chemicals and replacement materials (e.g., membrane modules). Generally, CO_2 from the oxidation of carbon-containing substances in municipal wastewater is considered biogenic and thus is excluded from the calculation of carbon emissions. However, for CH_4 and N_2O, the deduction of the biogenic component is not considered in the current IPCC guidelines. In addition, resource energy recovery and other carbon sinks (e.g., artificial wetlands, carbon capture and storage) can also be used to offset carbon emissions.

Some of the widely used carbon emission accounting methods in the field of water/wastewater treatment are the direct actual measurement method [94,95], the mass balance method [63], the emission factor method [96], and the LCA method [88]. Among them, the methods of LCA and emission factors are the most widely used. The direct actual measurement method has fewer intermediate links and more accurate results, but this method is disturbed by field sample collection and the difficulty of the data acquisition process. The mass balance method has advantages in data collection but has more intermediate processes. The emission factor method is an international common carbon emission estimation method based on the IPCC, with a wide range of applications, but the results are rough and uncertain. For specific WWTPs or wastewater treatment processes, the LCA method can account for the carbon footprint of the whole life cycle from the bottom up. This method has a detailed calculation process but has difficulty in obtaining the data and constructing the life-cycle inventory [97].

Most of the existing studies [17,59] on carbon accounting for MBRs used the LCA method to calculate the carbon footprint of the wastewater treatment process and compare it with other processes. Some studies [40] used the emission factor method to calculate the carbon emissions of wastewater treatment processes. Some studies [98,99] accounted for carbon emissions based on mass balance equations. Mannina et al. [100–104] combined mathematical modeling and biological models (in-built chemometric and kinetic models) to estimate and predict GHG emissions from MBR processes.

The results of carbon emission accounting for MBRs and other wastewater treatment processes in the existing literature are shown in Fig. 5, and a detailed list of the results is shown in Table S4 in Appendix A. The total carbon emission level of MBRs was 0.19–4.65 kilograms of carbon dioxide equivalents per cubic meter ($kgCO_2e{\cdot}m^{-3}$) with an average (median) value of 1.44 (0.99) $kgCO_2e{\cdot}m^{-3}$, while the total carbon emission level of other wastewater treatment processes (such as CAS) was 0.14–3.87 $kgCO_2e{\cdot}m^{-3}$ with an average (median) value of 0.84 (0.42) $kgCO_2e{\cdot}m^{-3}$. It can be seen that MBRs had a higher level of carbon emissions than the other processes in general [105–109]. A few studies have found that MBRs had lower carbon emissions than conventional wastewater treatment processes [58,62], which might be related to the specific process design, accounting scope boundaries, and accounting methods. It should be noted that the accounting of carbon emissions usually only considers the operational phase. If the whole life cycle carbon emissions are considered, the difference in GWP between MBRs and conventional wastewater treatment processes may be narrowed due to the advantages of MBR's small footprint of land occupation.

The high carbon footprint of MBRs mainly arose from the relatively high energy consumption. The average indirect carbon emissions from energy consumption were 1.01 $kgCO_2e{\cdot}m^{-3}$ (with a median value of 0.69 $kgCO_2e{\cdot}m^{-3}$), which accounted for about 70% of the total carbon emissions. The membrane tank contributed to more than 50% of the total carbon emissions, and controlled aeration could thus significantly reduce the carbon emissions [98,99,110]. Wastewater treatment combined with renewable energy sources (e.g., wind power and photovoltaics) could reduce emissions by around 95% [105]. AnMBRs generally had a lower level of carbon emissions than aerobic MBRs and other wastewater treatment processes, as AnMBRs can recover methane as a form of energy recovery [52,111].

In addition to the limitations of the LCA method mentioned in Section 4.1, the emission factor method for calculating carbon emissions from wastewater treatment processes faces problems such as difficulties in data collection, poor applicability of emission factors, and a high degree of uncertainty in the results. For example, emission factors of N_2O and CH_4 used to calculate direct carbon emissions are obtained from conventional WWTPs and are not necessarily applicable to MBRs. The mass balance method also faces the problems of difficult data collection, complicated calculation processes, and missing or inapplicable key parameters.

5. Techno–economic–environmental assessment

Understanding the sustainability of MBRs requires integrating the results of economic assessments and environmental impact assessments to analyze the trade-offs between the advantages and disadvantages of MBRs. There are currently three main approaches to integrated techno–economic–environmental assessment. The first method is CBA: environmental benefits are compared with economic costs to obtain the net benefits used to assess the economic–environmental viability [112,113]. Some of these studies directly used the price of reclaimed water as an environmental benefit of wastewater treatment [7], while others quantified the environmental benefits of WWTPs based on shadow prices of pollutants [112]. DEA, the second approach, is used to assess the economic–environmental effectiveness by defining the inputs and outputs of the wastewater treatment process and measuring the technical efficiency of WWTPs based on a distance function [114,115]. The third method is MCDA: integrating multiple dimensions such as economic costs, environmental impacts, and social benefits, determining the weight of each indicator through weight assignment methods (e.g., expert scoring), and calculating the sustainability score of the WWTP [7,30].

5.1. Cost–benefit analysis

CBA is a policy assessment method that quantifies in monetary terms the value of all the consequences of a policy for all members of society [116]. More generally, CBA is a useful tool for comparing

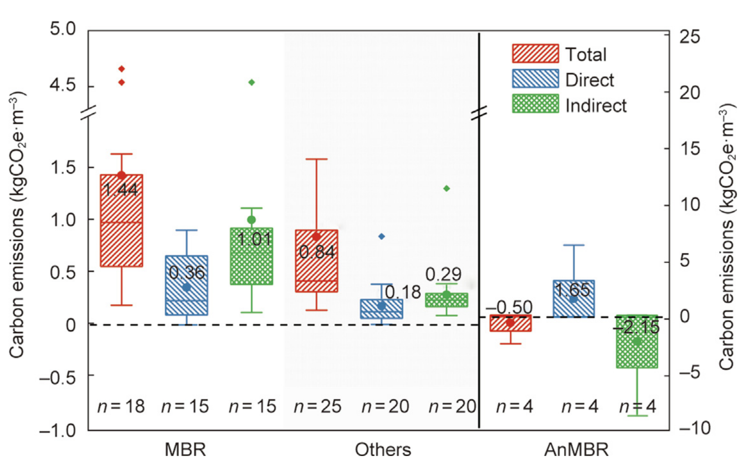


Fig. 5. GHG emissions of MBRs. Total: total carbon emissions (the sum of direct carbon emissions and indirect carbon emissions); Direct: direct carbon emissions; Indirect: indirect carbon emissions (from electricity consumption). The data are sourced from all the references listed in Table S4.

the economic feasibility of technologies, programs, projects, regulations, demonstrations, and policies. In the articles reviewed in this paper, the magnitude of benefits and costs was usually judged by the net benefits (incremental benefits minus incremental costs) [113,117,118]. A project or technology is economically viable when the benefits outweigh the costs. A few studies calculated the benefit–cost ratio, payback time, or internal rate of return to determine the economic feasibility of a project. A project is feasible if the benefit–cost ratio is greater than 1, if the payback time is less than the operating time, or if the internal rate of return is greater than the market interest rate [14,119].

Accounting for the costs of WWTPs is described in Section 3.2, while the calculation of benefits is more complex. Benefits from wastewater treatment are mainly environmental benefits from pollutant removal and water reuse, while negative environmental impacts from GHG and solid waste emissions need to be deducted [112,113]. Although the LCA methodology described in Section 4.1 can assess the environmental impacts of wastewater treatment processes, it is difficult to reasonably sum up and monetize the many environmental impacts. The environmental benefits of wastewater treatment cannot be captured through market pricing due to environmental externalities, and thus some studies calculated environmental benefits by estimating shadow prices using a directional distance function instead of market prices [112,117,120,121].

The shadow price of a pollutant is the marginal cost of removing a unit of pollutant and reflects the marginal benefit of reducing a unit of pollutant discharge. The level of the shadow price depends not only on the nature of the pollutant but also on the socio–economic and environmental conditions of the treatment plant itself [26]. The lower the effluent concentration of a pollutant, the higher the cost per unit of pollutant removed and the higher the shadow price of the pollutant. Generally, within the scope of ordinary pollutants, the removal of nutrients (nitrogen and phosphorus) from the effluent contributes the most to environmental benefits, while the removal of suspended particulate matter contributes relatively little to environmental benefits [26,113,117,122]. Environmental benefits are also related to the destination of pollutant discharges. Discharging to pollutant-sensitive locations (e.g., wetlands) can bring higher environmental benefits than discharging to the sea (with the potential for dilution and dispersion). The environmental benefits are higher if the treated water is reused rather than discharged directly [112,122].

The results of CBA are exemplified in Table S5 in Appendix A. Studies have shown that the majority of WWTPs were economically viable considering the external benefits of wastewater treatment [112,113,117]. CBA studies focusing on MBRs have shown that the MBRs were all economically viable [26,91,118,123]. Based on the data before and after the upgrading of 20 WWTPs (capacity ≥ 10 000 $m^3·d^{-1}$) in China, it was found that the net benefit of retrofitting from the CAS to MBR process increased significantly by 0.68 USD·m^{-3} (5 CNY·m^{-3}) [121].

Existing studies on CBA have mainly focused on conventional WWTPs, and fewer related studies have been conducted for the MBR process. During the process of CBA, most of the literature only considered the benefits of water reuse [118], and a few calculated the environmental benefits of pollutant removal by wastewater treatment [26,117]. However, negative environmental impacts such as material energy consumption and GHG emissions have been neglected [57]. In addition to environmental benefits, other possible benefits of wastewater treatment (e.g., health benefits) have scarcely been considered in the existing literature.

5.2. Data envelopment analysis

DEA is a common method in existing literature on evaluating environmental performance, environmental governance efficiency, and technical efficiency [124–126]. DEA utilizes linear programming to take the optimal input–output as the production frontier and to construct the data envelopment curve, to compute the relative technical efficiency of the decision-making units (DMUs). The closer the sample point of DMUs is to the frontier, the higher its technical efficiency (ranging from 0 to 1). The technical efficiency is equal to 1 when the sample point is on the frontier, which means that the sample has the optimal technical efficiency and its output is the maximum possible output given the inputs.

The CCR model (the three letters in the model name represent the initials of the three authors, respectively, as does the BBC model below) is the earliest DEA model, proposed by Charnes et al. [127], which assumes constant returns to scale and measures the technical efficiency. Banker et al. [128] modified the CCR model into the BCC model by assuming variable returns to scale. Both the CCR and BBC models measure radial efficiency by allowing inputs or outputs to increase or decrease in equal proportions, but this assumption is not applicable in all cases. Tone [129] proposed the slacks-based measure (SBM) model, which takes into account the difference between inputs and outputs (i.e., slack) to measure non-radial efficiency. Considering the existence of undesirable outputs and the fact that the two types of outputs (desirable outputs and undesirable outputs) do not necessarily increase or decrease in the same proportion, Zhou et al. [126] proposed the non-radial directional distance function (NDDF). Due to the dynamic changes in efficiency, the Malmquist index proposed by Malmquist [130] and later extended by Caves et al. [131] can measure the dynamic changes in the technical efficiency of DMUs over time.

Some of the DEA studies on WWTPs are shown in Table 2 [26,114,121,132–141]. Most of the studies used cost or energy consumption as inputs and pollutant reductions as desirable outputs to calculate technical efficiency. The technical efficiency reflects the degree of proximity of WWTPs to the optimal state under the considerations of treatment performance and cost/energy consumption [142]. It was found that the overall technical efficiency of conventional WWTPs was 0.3–0.4, with only 10% of them reaching an efficiency value of 1. The size of the plant (design scale), the amount of organic matter removed, and the type of aeration are important factors explaining the differences in technical efficiency: larger WWTPs are more efficient than smaller ones; plants removing more COD are more efficient; and plants using diffusers are 28% more efficient than those using turbines [132].

However, DEA studies evaluating the efficiency of wastewater treatment technologies have mainly focused on conventional WWTPs and few specific analyses have been conducted for MBRs. Upon searching, as of September 2, 2024, there were only four DEA studies on MBRs [26,121,133,134]. Of these, Bick et al. [133] focused on pilot-scale MBRs, while Gao et al. [121] focused on actual full-scale MBRs. Studies showed that effluent standards, construction positions (aboveground/underground), influent type, geographic location, and year of operation may also affect the technical efficiency of MBRs [26]. After retrofitting the WWTP from the CAS to MBR process, the average energy efficiency was unchanged and the average cost efficiency was improved. This suggests that the MBR and CAS processes are comparable in terms of technical efficiency and that there is still room for MBRs to improve energy efficiency [121].

Overall, there is no unified paradigm for DEA modeling applied to WWTPs, and the relevant literature is limited and lacks interconnections. In addition, there is still room for improvement in DEA modeling when calculating the technical efficiency of WWTPs. On the one hand, undesirable outputs such as sludge generation and GHG emissions have been rarely considered in DEA models in the literature. On the other hand, the model setup itself may suffer from relaxation bias, which leads to increased uncertainty in the model results. When applying a specific DEA model (e.g., the

Table 2
DEA studies of wastewater treatment plants.

Reference	MBR	DEA model	Input	Desirable output	Undesirable output
[26]	Yes	NDDF	Total cost or energy consumption	The volume of treated wastewater and the rate of removing pollutants (i.e., BOD, COD, and NH_3-N)	Sludge production
[114]	No	CCR	Electricity consumption per day	The amount of pollutants per day (i.e., COD, N, and P)	—
[115]	No	Non-concavemeta frontier	Cost per cubic meter of wastewater treated	The annual amount of removing pollutants (i.e., COD, N, and P)	—
[121]	Yes	NDDF	Operating cost or energy consumption	The volume of treated wastewater and the rate of removing pollutants (i.e., COD and NH_3-N)	Sludge production
[132]	No	Non-radial DEA	Cost per cubic meter of wastewater treated (the cost consists of energy consumption, chemical consumption, labor cost, maintenance cost, and sludge treatment cost)	The reduction concentrations of pollutants (i.e., SS and COD)	—
[133]	Yes	—	Feed temperature, transmembrane pressure, and BOD_5 concentrations in bioreactor	Permeate flux and the removal rate of BOD_5	—
[134]	Yes	—	Feed temperature and transmembrane pressure.	Normalized flux and fouling rate prevention	—
[135]	No	WSBM	Cost per cubic meter of wastewater treated (the cost consists of energy consumption, chemical consumption, labor cost, and maintenance cost)	The annual amount of removing pollutants (i.e., COD, SS, N, and P)	—
[136]	No	BCC	Energy consumption and construction cost	The rate of removing pollutants (i.e., COD, SS, TN, and TP)	Carbon emissions
[137]	No	CCR	Cost per cubic meter of wastewater treated (the cost consists of energy consumption, labor cost, maintenance cost, and sludge treatment cost)	Total annual reduction of pollutants (i.e., SS, COD, and BOD)	—
[138]	No	SBM	Total electricity consumption	The amount of removing pollutants (i.e., COD, BOD_5, TN, NH_3-N, and TP)	—
[139]	No	DEA with statistical tolerances	Cost per cubic meter of wastewater treated	The annual amount of removing pollutants (i.e., COD, SS, and N)	—
[140]	No	BCC	Electricity consumption per cubic meter of wastewater treated	The rate of removing pollutants (i.e., COD, SS, NH_3-N, and TP)	—
[141]	No	Malmquist index	Design scale, load rate, accumulated electricity consumption, electricity consumption per cubic meter of wastewater treated, and actual treatment capacity	The amount of pollutant removal per electricity consumption (i.e., COD, BOD, SS, NH_3-N, TN, and TP)	—

WSBM: the weighted slack-based measure; BOD or BOD_5: biochemical oxygen demand or 5 day BOD; SS: suspended solids.

BBC model), the assumptions of the model and the reasons for its application need to be clarified. It is important to note that DEA models calculate the relative technical efficiency of DMUs, and the value of efficiency is not comparable between articles.

5.3. Multi-criteria decision analysis

Many studies have evaluated and compared MBRs and conventional wastewater treatment processes from multiple perspectives, including environmental, technological, and economic aspects. However, these studies only conducted preliminary single-factor evaluations from multiple aspects and did not aggregate multiple dimensions due to the lack of data and methodological limitations [31,51,57]. CBA and DEA methods aggregate the environmental and economic dimensions to some extent, but both methods have requirements for quantitative information.

Some studies compared MBRs and conventional wastewater treatment processes by constructing a comprehensive sustainability evaluation index through MCDA, which takes into account multiple dimensions such as environment, technology, and economy [7,30]. MCDA usually includes five steps, such as, defining the objective, defining the criteria, weighting the criteria, aggregation, and rating options [74].

MCDA related to wastewater treatment processes mainly considered environmental, technological, and economic dimensions, with a few studies also considering other dimensions such as society and management [30,143]. Environmental dimensions include sub-dimensions such as effluent quality, GWP, and acidification potential, which can be directly utilized with the results of LCA [144]. Technical dimensions usually include sub-dimensions such as reliability, flexibility, modularity, and complexity. Economic dimensions consist of sub-dimensions such as construction costs and O&M costs. Social dimensions include sub-dimensions such as public acceptance, simplicity, and level of risk to health and safety [145].

After determining the criteria, the criteria and sub-criteria need to be quantified and weighted, which in turn can lead to a composite sustainability score. The weighting information of MCDA is shown in Table S6 in Appendix A. The methods of weighting include the Delphi method [146], expert survey method [147], principal component analysis (PCA) [148], analytic hierarchy process (AHP) [108,145,149], technique for order preference by similarity to an ideal solution (TOPSIS) [150] and fuzzy comprehensive evaluation method [151]. Most of these methods are subjective weighting methods that involve a range of subjective factors such as personal perception, natural environment, economic conditions, local policies, and specific assumptions [31].

Studies [30,143] have shown that MBRs had superior technical performance and social sustainability performance, with consistent effluent quality and high pollutant removal efficiency. Thus, MBRs were more adaptable to increasingly stringent standard limits and facilitated the long-term operation of wastewater treatment and reuse projects. However, from a life-cycle perspective, it was reported that MBRs were more environmentally burdensome and economically costly [31]. The conclusions of existing studies on MCDA were inconsistent, with some suggesting that the sustainability performance of MBRs was not dominant compared to conventional wastewater treatment processes [7,30,108,152]. It has also been shown that MBRs had a high sustainability index relative to CAS processes [146,149,151]. The inconsistency in conclusions may be related to factors such as research objectives, guideline setting, data, and methodology.

Although MCDA can provide a comprehensive evaluation of wastewater treatment processes in terms of technology, economy, environment, and society, existing studies have used a large number of qualitative indicators and adopted subjective weighting methods. The results are often limited by local conditions and subjective factors, and there is a great deal of uncertainty in the setting of weights and the construction of indices.

6. Discussion

This paper reviewed the research on techno–economic–environmental assessment in MBRs for wastewater treatment. The relevant research can be divided into three categories: economic cost analysis, environmental impact assessment, and integrated techno–economic–environmental assessment of MBRs. This section provides a comparative analysis of these three types of studies (Table 3). The scope of these studies determines the different focuses, methodologies, and key points of the research.

Economic costing aggregates the various cost categories into total annual or life-cycle costs using the well-established cash-flow methods of economics. The key points in such studies are discounting and amortization, and there are difficulties in obtaining detailed cost data. LCA is commonly used for environmental impact assessment. The key point of this method is to determine the study boundary, functional units, and impact categories while being limited by the availability and applicability of inventory data.

The advantage of integrated techno–economic–environmental assessment is that it combines economic cost assessment and environmental impact assessment to evaluate the sustainability performance of wastewater treatment processes. Such studies can be further divided into three categories based on methodology. CBA assesses the feasibility of a process by calculating net benefits based on total costs and total benefits, but the method is limited by the difficulty of assessing and monetizing environmental impacts. DEA calculates the technical efficiency of DMUs based on multiple inputs and outputs of the wastewater treatment process. The selection of inputs, outputs, and DEA models has a significant impact on the results. Inadequate attention to undesirable outputs and the difficulty of comparing results from different studies are the main bottlenecks in such studies. MCDA is a method for constructing a composite index based on multiple objectives. The key to this method is to identify, quantify, and weigh different indicators. The selection of indicators and the assignment of weights are somewhat subjective, making the results highly uncertain.

The integrated techno–economic–environmental assessment (Section 5) can be interconnected with the economic cost evaluation (Section 3) and environmental impact assessment (Section 4). The implementation of CBA requires the costs calculated in the economic cost assessment discussed in Section 3, as well as the monetized environmental impact results from the environmental impact assessment discussed in Section 4, as inputs. As shown in Fig. 1, the inputs required for DEA are the same as those in the economic cost assessment and environmental impact assessment (e.g., operational costs and energy consumption), but it does not require the results from the economic cost assessment and environmental impact assessment, such as total costs or life cycle costs. For MCDA, the inputs can either be the same as those in the economic cost assessment and environmental impact assessment or derive from the results of these assessments.

Overall, different types of techno–economic–environmental assessment studies have their characteristics. Depending on the research questions and objects, different methods have to be chosen. All such studies need to pay attention to the definition of the study scope, the description of possible omissions, and the basis for the selection and assignment of indicators.

7. Summary and prospects

This paper reviewed the literature on economic cost analysis, environmental impact assessment, and techno–economic–environmental assessment of MBRs.

The cost of an MBR consists of construction costs, operating costs (including maintenance costs), and membrane replacement costs, according to the specific studies. The total cost of an MBR is calculated in two ways. One is to sum the annual operating costs and the annual depreciation of construction costs to obtain the total annual cost. The other is to calculate the net present value of life cycle costs by summing the construction costs and the discounted value of the operating and membrane replacement costs, according to specific studies. Overall, the operating costs and energy consumption of MBRs were generally higher than those of

Table 3
Comparison of different techno–economic and environmental impact assessment methods.

Dimensions	Economic cost evaluation	Environmental impact assessment	Techno–economic–environmental assessment		
	Cost calculation	LCA	CBA	DEA	MCDA
Inputs	Capital cost, operating cost, asset lifetime, membrane lifetime, and discount rate	Energy consumption, material consumption, chemical consumption, emissions to air, emissions to water, emissions to land, and so forth	Total cost and total benefit	Capital cost, operating cost, the volume of treated wastewater, sludge production, and so forth	Economic indicators (e.g., construction costs), technical indicators (e.g., reliability), environmental indicators (e.g., effluent quality), and so forth
Outputs (results)	Total cost or life cycle cost	Various environmental impacts	Net benefit	Technical efficiency	Sustainability index
Key points	Discount and amortization	Definition of assessment scope, functional units, and impact categories	Monetization of environmental impacts	Selection of inputs, outputs, and DEA models	Definition, quantification, and weighting of criteria and sub-criteria
Characteristics	Difficulty of acquisition of detailed cost data	Difficulty of data acquisition and applicability of data inventories	Difficulty of estimation and monetization of environmental impacts	Insufficient consideration of undesirable outputs and non-comparability of results across studies	Highly subjective and uncertain

Scope of this review: techno–economic and environmental impact assessment.
Common points of attention: definition of the scope of the study, description of possible omissions, and basis for the selection and assignment of indicators.

conventional wastewater treatment processes (energy consumption: MBR vs CAS = 0.4–1.15 vs 0.3–0.64 kW·h·m^{-3}; operating costs: MBR vs CAS = 0.09–0.45 vs 0.02–0.4 USD·m^{-3}). Comparisons of construction costs (excluding land costs) with conventional processes have not been established in the literature. The total cost of MBRs was slightly higher than that of other wastewater treatment processes, both in terms of annual costs and net present value of life-cycle costs (average total costs: MBR vs CAS = 0.25 vs 0.19 USD·m^{-3}). MBRs would be more cost-competitive if the land-saving advantages of MBRs were considered. However, the existing studies were still unclear or incorrect in terms of accounting methodology and defining the cost range. The following areas need to be clarified in the subsequent costing process: the definition of the scope of the costing, cost items that may be missing (e.g., land and personnel costs), and discounting or depreciation in the process of cost summation, according to the specific studies.

Existing studies usually employed LCA to assess the environmental impacts of MBRs and other wastewater treatment processes. If we focus only on the GWP of MBRs, the emission factor method based on IPCC guidelines and the mass balance method based on mathematical and biological modeling are also important options for accounting for the GHG emissions of MBRs. The environmental impacts of MBRs on the eutrophication of water bodies are minimal, but the GWP is a key constraint to the sustainable development of MBRs. The level of carbon emissions from MBRs (with an average of 1.44 $kgCO_2e·m^{-3}$) was higher than from other processes (with an average of 0.84 $kgCO_2e·m^{-3}$). Energy consumption is regarded as the major contributor to the relatively high carbon emissions of MBRs. Optimization of aeration and sludge management, increasing the proportion of renewable energy use, and improving the efficiency of energy recovery can contribute to energy saving and emission reduction in MBRs. Note that the accounting scope of carbon emissions here usually only includes the operation phase. When considering life-cycle carbon emissions, the difference in carbon emissions between MBRs and conventional processes may be reduced due to MBR's small footprint. However, in the current LCA studies, the implementation processes and the results lack consistency, reliability, and reproducibility. Therefore, the future study of LCA calls for a more standardized LCA implementation. In terms of carbon emission accounting, no matter LCA, emission factor method, or mass balance method, all of them face problems such as lack of data and uncertainty of key parameters. These methods need to calibrate key parameters such as emission factors with measured carbon emission data.

There are three types of techno–economic–environmental assessment methods: CBA, DEA, and MCDA. The results of the CBA studies indicated that MBRs were all economically viable (net benefits greater than 1). Most of the CBA studies used the benefits of water reuse as the environmental benefits of wastewater treatment processes. A few studies calculated the environmental benefits of removing pollutants from wastewater based on the shadow price assessment of pollutants. However, none of the studies considered the negative environmental impacts of wastewater treatment such as material consumption and GHG emissions. The results of the DEA studies showed that the overall efficiency of conventional WWTPs was not high. By comparing the WWTP before and after retrofitting, the MBR and conventional wastewater treatment processes were at a similar level in terms of technological efficiency, but there is appreciable room for the MBR to improve energy efficiency. There have been very few applications of the DEA model for MBRs, no unified paradigm, and few stated model assumptions. Consequently, the results of DEA calculated by different studies were not comparable. Moreover, most of the previous studies did not take into account undesirable outputs such as sludge generation and GHG emissions. The results of the MCDA studies indicated that MBRs had better technical performance and inferior environmental and economic performance. For the comprehensive evaluation of sustainability performance, the comparison between MBRs and other wastewater treatment processes has not been concluded. MCDA is limited to criteria setting, data collection, and weight setting. The existing studies mainly used qualitative indicators and subjective weighting methods, with great uncertainty in the results. Overall, the integrated techno–economic–environmental assessment method has been less applied to MBRs and other wastewater treatment processes. The application of the method suffers from poorly defined scope and definitions, failure to state model assumptions and reasons for their application and low external validity of the results.

Research on the integrated techno–economic–environmental assessment in the field of wastewater treatment is in its infancy, and studies applied to specific MBR processes are even rarer. The existing studies have limitations in terms of methodological applicability, reliability, and comparability of results. Improving the professionalism of economic assessment, the standardization of environmental impact assessment, and the science of integrated techno–economic–environmental assessment are directions that need to be continued in the future.

CRediT authorship contribution statement

Tingwei Gao: Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Yana Jin:** Writing – review & editing, Validation, Supervision, Resources, Methodology, Formal analysis. **Kang Xiao:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Acknowledgment

This work was supported by the Beijing Natural Science Foundation (JQ22027), the Weiqiao–University of Chinese Academy of Sciences Special Projects on Low-Carbon Technology Development (GYY-DTFZ-2022-021), and the Fundamental Research Funds for the Central Universities (E2EG0502X). Xiaomao Wang of Tsinghua University and Kuichang Zuo of Peking University are gratefully acknowledged for their insightful suggestions.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eng.2025.05.001.

References

[1] Judd S, Judd C. The MBR book: principles and applications of membrane bioreactors in water and wastewater treatment. 2nd ed. Oxford: Elsevier; 2011.
[2] Judd SJ. The status of industrial and municipal effluent treatment with membrane bioreactor technology. Chem Eng J 2016;305:37–45.
[3] Li P, Liu L, Wu J, Cheng R, Shi L, Zheng X, et al. Identify driving forces of MBR applications in China. Sci Total Environ 2019;647:627–38.
[4] Xiao K, Xu Y, Liang S, Lei T, Sun J, Wen X, et al. Engineering application of membrane bioreactor for wastewater treatment in China: current state and future prospect. Front Environ Sci Eng 2014;8(6):805–19.
[5] Huang X, Wen X. Principles and applications of membrane bioreactors for water treatment. Beijing: Science Press; 2012. Chinese.
[6] Zheng X, Wei Y, Wang Z, Zhang Z. Report for sustainable development strategy of China water treatment industry. Beijing: China Renmin University Press Report; 2016. Chinese.

[7] Hao XD, Li J, Van Loosdrecht MCM, Li TY. A sustainability-based evaluation of membrane bioreactors over conventional activated sludge processes. J Environ Chem Eng 2018;6(2):2597–605.
[8] Krzeminski P, Leverette L, Malamis S, Katsou E. Membrane bioreactors—a review on recent developments in energy reduction, fouling control, novel configurations, LCA and market prospects. J Membr Sci 2017;527:207–27.
[9] Pretel R, Robles A, Ruano MV, Seco A, Ferrer J. 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. J Environ Manage 2016;166:45–54.
[10] Xiao K, Liang S, Wang X, Chen C, Huang X. Current state and challenges of full-scale membrane bioreactor applications: a critical review. Bioresour Technol 2019;271:473–81.
[11] Adham S, Gagliardo P, Boulos L, Oppenheimer J, Trussell R. Feasibility of the membrane bioreactor process for water reclamation. Water Sci Technol 2001;43(10):203–9.
[12] Abuabdou SM, Ahmad W, Aun NC, Bashir MJ. A review of anaerobic membrane bioreactors (AnMBR) for the treatment of highly contaminated landfill leachate and biogas production: effectiveness, limitations and future perspectives. J Clean Prod 2020;255:120215.
[13] Asif MB, Zhang Z. Ceramic membrane technology for water and wastewater treatment: a critical review of performance, full-scale applications, membrane fouling and prospects. Chem Eng J 2021;418:129481.
[14] Friedler E, Hadari M. Economic feasibility of on-site greywater reuse in multi-storey buildings. Desalination 2006;190(1–3):221–34.
[15] Valladares Linares R, Li Z, Yangali-Quintanilla V, Ghaffour N, Amy G, Leiknes T, et al. Life cycle cost of a hybrid forward osmosis—low pressure reverse osmosis system for seawater desalination and wastewater recovery. Water Res 2016;88:225–34.
[16] Smith AL, Stadler LB, Love NG, Skerlos SJ, Raskin L. Perspectives on anaerobic membrane bioreactor treatment of domestic wastewater: a critical review. Bioresour Technol 2012;122:149–59.
[17] Smith AL, Stadler LB, Cao L, Love NG, Raskin L, Skerlos SJ. Navigating wastewater energy recovery strategies: a life cycle comparison of anaerobic membrane bioreactor and conventional treatment systems with anaerobic digestion. Environ Sci Technol 2014;48(10):5972–81.
[18] Wei Y, Van Houten RT, Borger AR, Eikelboom DH, Fan Y. Minimization of excess sludge production for biological wastewater treatment. Water Res 2003;37(18):4453–67.
[19] Yamamoto K, Hiasa M, Mahmood T, Matsuo T. Direct solid–liquid separation using hollow fiber membrane in an activated sludge aeration tank. Water Sci Technol 1988;21(4–5):43–54.
[20] DeCarolis J, Adham S, Pearce WR, Hirani Z, Lacy S, Stephenson R. Cost trends of MBR systems for municipal wastewater treatment. In: Proceedings of the Water Quality Event; 80th Annual Water Environment Federation Technical Exhibition and Conference; 2007 Oct 13–17; San Diego, CA, USA. Alexandria: Water Environment Federation; 2007. p. 3407–18.
[21] Gabarrón S, Ferrero G, Dalmau M, Comas J, Rodriguez-Roda I. Assessment of energy-saving strategies and operational costs in full-scale membrane bioreactors. J Environ Manage 2014;134:8–14.
[22] Verrecht B, Maere T, Nopens I, Brepols C, Judd S. The cost of a large-scale hollow fibre MBR. Water Res 2010;44(18):5274–83.
[23] Jiang F, Beck MB, Cummings RG, Rowles K, Russell D. Estimation of costs of phosphorus removal in wastewater treatment facilities: construction *de novo*. In: Water Policy Working Paper. Athens: University of Georgia; 2005. p. 28.
[24] Jiang F, Beck MB, Cummings RG, Rowles K, Russell D. Estimation of costs of phosphorus removal in wastewater treatment facilities: adaptation of existing facilities. Report. Atlanta, GA: Georgia State University; 2005 Feb.
[25] Bashar R, Gungor K, Karthikeyan KG, Barak P. Cost effectiveness of phosphorus removal processes in municipal wastewater treatment. Chemosphere 2018;197:280–90.
[26] Gao T, Xiao K, Zhang J, Zhang X, Wang X, Liang S, et al. Cost–benefit analysis and technical efficiency evaluation of full-scale membrane bioreactors for wastewater treatment using economic approaches. J Clean Prod 2021;301:126984.
[27] Iglesias R, Simón P, Moragas L, Arce A, Rodriguez-Roda I. Cost comparison of full-scale water reclamation technologies with an emphasis on membrane bioreactors. Water Sci Technol 2017;75(11):2562–70.
[28] Vinardell S, Blandin G, Ferrari F, Lesage G, Mata-Alvarez J, Dosta J, et al. Techno–economic analysis of forward osmosis pre-concentration before an anaerobic membrane bioreactor: impact of draw solute and membrane material. J Clean Prod 2022;356:131776.
[29] Nicolaidis C, Vyrides I. Closing the water cycle for industrial laundries: an operational performance and techno–economic evaluation of a full-scale membrane bioreactor system. Resour Conserv Recycling 2014;92:128–35.
[30] Bertanza G, Canato M, Laera G, Vaccari M, Svanström M, Heimersson S. A comparison between two full-scale MBR and CAS municipal wastewater treatment plants: techno–economic–environmental assessment. Environ Sci Pollut Res Int 2017;24(21):17383–93.
[31] Chen Z, Wang D, Sun M, Ngo HH, Guo W, Wu G, et al. Sustainability evaluation and implication of a large scale membrane bioreactor plant. Bioresour Technol 2018;269:246–54.
[32] Ioannou-Ttofa L, Michael-Kordatou I, Fattas SC, Eusebio A, Ribeiro B, Rusan M, et al. Treatment efficiency and economic feasibility of biological oxidation, membrane filtration and separation processes, and advanced oxidation for the purification and valorization of olive mill wastewater. Water Res 2017;114:1–13.
[33] Tawfik A, Al-Sayed A, Hassan GK, Nasr M, El-Shafai SA, Alhajeri NS, et al. Electron donor addition for stimulating the microbial degradation of 1, 4 dioxane by sequential batch membrane bioreactor: a techno–economic approach. Chemosphere 2022;306:135580.
[34] Awad H, Gar Alalm M, El-Etriby HK. Environmental and cost life cycle assessment of different alternatives for improvement of wastewater treatment plants in developing countries. Sci Total Environ 2019;660:57–68.
[35] Da Costa Fonseca MJ, da Silva P, Jr BCP, da Fonseca FV. Ethinylestradiol removal of membrane bioreactor effluent by reverse osmosis and UV/H_2O_2: a technical and economic assessment. J Environ Manage 2021;282:111948.
[36] Vinardell S, Astals S, Koch K, Mata-Alvarez J, Dosta J. Co-digestion of sewage sludge and food waste in a wastewater treatment plant based on mainstream anaerobic membrane bioreactor technology: a techno–economic evaluation. Bioresour Technol 2021;330:124978.
[37] Tan X, Shi L, Chen Z, Li T, Ma Z, Zheng X, et al. Cost analysis of the municipal wastewater treatment plant operation based on 227 samples in China. Jishui Paishui 2015;41(5):30–4. Chinese.
[38] Yu D, Wang J, Zheng L, Sui Q, Zhong H, Cheng M, et al. Effects of hydraulic retention time on net present value and performance in a membrane bioreactor treating antibiotic production wastewater. Front Environ Sci Eng 2020;14(6):101.
[39] Cicekalan B, Kosar S, Cingoz S, Eyit N, Ersahin ME, Ozgun H. Techno–economic and environmental assessment of different municipal wastewater treatment systems. J Water Process Eng 2023;53:103822.
[40] Ab Hamid NH, Smart S, Wang DK, Koh KWJ, Ng KJC, Ye L. Economic, energy and carbon footprint assessment of integrated forward osmosis membrane bioreactor (FOMBR) process in urban wastewater treatment. Environ Sci Water Res Technol 2020;6(1):153–65.
[41] Brepols C, Schäfer H, Engelhardt N. Considerations on the design and financial feasibility of full-scale membrane bioreactors for municipal applications. Water Sci Technol 2010;61(10):2461–8.
[42] Echevarría C, Valderrama C, Cortina JL, Martín I, Arnaldos M, Bernat X, et al. Techno–economic evaluation and comparison of PAC–MBR and ozonation-UV revamping for organic micro-pollutants removal from urban reclaimed wastewater. Sci Total Environ 2019;671:288–98.
[43] Lo CH, McAdam E, Judd S. The cost of a small membrane bioreactor. Water Sci Technol 2015;72(10):1739–46.
[44] AbdelMoula S, Sorour MT, Aly SAA. Cost analysis and health risk assessment of wastewater reuse from secondary and tertiary wastewater treatment plants. Sustainability 2021;13(23):13125.
[45] Arif AUA, Sorour MT, Aly SA. Cost analysis of activated sludge and membrane bioreactor WWTPs using CapdetWorks simulation program: case study of Tikrit WWTP (middle Iraq). Alex Eng J 2020;59(6):4659–67.
[46] Young T, Muftugil M, Smoot S, Peeters J. MBR vs. CAS: capital and operating cost evaluation. Water Pract Technol 2012;7(4):wpt2012075.
[47] Zhang J, Xiao K, Liu Z, Gao T, Liang S, Huang X. Large-scale membrane bioreactors for industrial wastewater treatment in China: technical and economic features, driving forces, and perspectives. Engineering 2021;7(6):868–80.
[48] Krzeminski P, Langhorst W, Schyns P, De Vente D, Van den Broeck R, Smets IY, et al. The optimal MBR configuration: hybrid versus stand-alone—comparison between three full-scale MBRs treating municipal wastewater. Desalination 2012;284:341–8.
[49] Sun J, Liang P, Yan X, Zuo K, Xiao K, Xia J, et al. Reducing aeration energy consumption in a large-scale membrane bioreactor: process simulation and engineering application. Water Res 2016;93:205–13.
[50] Wu Q, Yan X, Xiao K, Guan J, Li T, Liang P, et al. Optimization of membrane unit location in a full-scale membrane bioreactor using computational fluid dynamics. Bioresour Technol 2018;249:402–9.
[51] Jiménez-Benítez A, Ruiz-Martínez A, Ferrer J, Ribes J, Rogalla F, Robles A. Life cycle costing of AnMBR technology for urban wastewater treatment: a case study based on a demo-scale AnMBR system. J Environ Chem Eng 2023;11(3):110267.
[52] Das S, O'Connell MG, Xu H, Bernstein R, Kim JH, Sankhala K, et al. Assessing advances in anti-fouling membranes to improve process economics and sustainability of water treatment. ACS ES&T Eng 2022;2(11):2159–73.
[53] Cashman S, Ma X, Mosley J, Garland J, Crone B, Xue X. Energy and greenhouse gas life cycle assessment and cost analysis of aerobic and anaerobic membrane bioreactor systems: influence of scale, population density, climate, and methane recovery. Bioresour Technol 2018;254:56–66.
[54] Lin H, Chen J, Wang F, Ding L, Hong H. Feasibility evaluation of submerged anaerobic membrane bioreactor for municipal secondary wastewater treatment. Desalination 2011;280(1–3):120–6.
[55] Hernández-Sancho F, Molinos-Senante M, Sala-Garrido R. Cost modelling for wastewater treatment processes. Desalination 2011;268(1–3):1–5.
[56] Wang X, Liu J, Ren NQ, Duan Z. Environmental profile of typical anaerobic/anoxic/oxic wastewater treatment systems meeting increasingly stringent treatment standards from a life cycle perspective. Bioresour Technol 2012;126:31–40.
[57] Høibye L, Clauson-Kaas J, Wenzel H, Larsen HF, Jacobsen BN, Dalgaard O. Sustainability assessment of advanced wastewater treatment technologies. Water Sci Technol 2008;58(5):963–8.

[58] Banti DC, Tsangas M, Samaras P, Zorpas A. LCA of a membrane bioreactor compared to activated sludge system for municipal wastewater treatment. Membranes 2020;10(12):421.

[59] Becker Jr AM, Yu K, Stadler LB, Smith AL. Co-management of domestic wastewater and food waste: a life cycle comparison of alternative food waste diversion strategies. Bioresour Technol 2017;223:131–40.

[60] Harclerode M, Doody A, Brower A, Vila P, Ho J, Evans PJ. Life cycle assessment and economic analysis of anaerobic membrane bioreactor whole-plant configurations for resource recovery from domestic wastewater. J Environ Manage 2020;269:110720.

[61] Ioannou-Ttofa L, Foteinis S, Chatzisymeon E, Fatta-Kassinos D. The environmental footprint of a membrane bioreactor treatment process through life cycle analysis. Sci Total Environ 2016;568:306–18.

[62] Yang X, López-Grimau V, Vilaseca M, Crespi M. Treatment of textile wastewater by CAS, MBR, and MBBR: a comparative study from technical, economic, and environmental perspectives. Water 2020;12(5):1306.

[63] Foley J, De Haas D, Yuan Z, Lant P. Nitrous oxide generation in full-scale biological nutrient removal wastewater treatment plants. Water Res 2010;44 (3):831–44.

[64] Gallego A, Hospido A, Moreira MT, Feijoo G. Environmental performance of wastewater treatment plants for small populations. Resour Conserv Recycl 2008;52(6):931–40.

[65] Lorenzo-Toja Y, Alfonsín C, Amores MJ, Aldea X, Marin D, Moreira MT, et al. Beyond the conventional life cycle inventory in wastewater treatment plants. Sci Total Environ 2016;553:71–82.

[66] Lu B, Du X, Huang S. The economic and environmental implications of wastewater management policy in China: from the LCA perspective. J Clean Prod 2017;142:3544–57.

[67] Kamble S, Singh A, Kazmi A, Starkl M. Environmental and economic performance evaluation of municipal wastewater treatment plants in India: a life cycle approach. Water Sci Technol 2019;79(6):1102–12.

[68] Jeong H, Broesicke OA, Drew B, Crittenden JC. Life cycle assessment of small-scale greywater reclamation systems combined with conventional centralized water systems for the city of Atlanta, Georgia. J Clean Prod 2018;174:333–42.

[69] Romeiko XX, Lin S, Huang G. Life cycle assessment of preserved plum production in Southern China. Clean Technol Environ Policy 2020;22 (1):197–209.

[70] Muñoz I, Aktürk AS, Ayyıldız Ö, Çağlar Ö, Meabe E, Contreras S, et al. Life cycle assessment of wastewater reclamation in a petroleum refinery in Turkey. J Clean Prod 2020;268:121967.

[71] Kobayashi Y, Ashbolt NJ, Davies EG, Liu Y. Life cycle assessment of decentralized greywater treatment systems with reuse at different scales in cold regions. Environ Int 2020;134:105215.

[72] Xu X, Dao H, Bair R, Uman AE, Yeh D, Zhang Q. Discharge or reuse? Comparative sustainability assessment of anaerobic and aerobic membrane bioreactors. J Environ Qual 2020;49(3):545–56.

[73] Abyar H, Nowrouzi M. Highly efficient reclamation of meat-processing wastewater by aerobic hybrid membrane bioreactor-reverse osmosis simulated system: a comprehensive economic and environmental study. ACS Sustain Chem Eng 2020;8(37):14207–16.

[74] Akhoundi A, Nazif S. Sustainability assessment of wastewater reuse alternatives using the evidential reasoning approach. J Clean Prod 2018;195:1350–76.

[75] Akhoundi A, Nazif S. Life-cycle assessment of tertiary treatment technologies to treat secondary municipal wastewater for reuse in agricultural irrigation, artificial recharge of groundwater, and industrial usages. J Environ Eng 2020;146(6):04020031.

[76] Drosou F, Kekes T, Boukouvalas C. Life cycle assessment of the canned fruits industry: sustainability through waste valorization and implementation of innovative techniques. AgriEngineering 2023;5(1):395–412.

[77] Foglia A, Andreola C, Cipolletta G, Radini S, Akyol Ç, Eusebi AL, et al. Comparative life cycle environmental and economic assessment of anaerobic membrane bioreactor and disinfection for reclaimed water reuse in agricultural irrigation: a case study in Italy. J Clean Prod 2021;293:126201.

[78] Glover CJ, Phillips JA, Marchand EA, Hiibel SR. Modeling and life cycle assessment of a membrane bioreactor–membrane distillation wastewater treatment system for potable reuse. Separations 2022;9(6):151.

[79] Hospido A, Sanchez I, Rodriguez-Garcia G, Iglesias A, Buntner D, Reif R, et al. Are all membrane reactors equal from an environmental point of view? Desalination 2012;285:263–70.

[80] Khaki E, Boleydei H, Abyar H, Nowrouzi M. Integrating eco-environmental assessment with energy recovery for petrochemical wastewater treatment technologies: a transition towards green and sustainable management. J Water Process Eng 2023;55:104103.

[81] Lazarova V, Martin Ruel S, Barillon B, Dauthuille P. The role of MBR technology for the improvement of environmental footprint of wastewater treatment. Water Sci Technol 2012;66(10):2056–64.

[82] Pretel R, Shoener BD, Ferrer J, Guest JS. Navigating environmental, economic, and technological trade-offs in the design and operation of submerged anaerobic membrane bioreactors (AnMBRs). Water Res 2015;87:531–41.

[83] Rashidi J, Rhee G, Kim M, Nam K, Heo S, Yoo C, et al. Life cycle and economic assessments of key emerging energy efficient wastewater treatment processes for climate change adaptation. Int J Environ Res 2018;12 (6):815–27.

[84] Satayavibul A, Ratanatamskul C. 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 Biomass Valoriz 2021;12(10):5425–36.

[85] Sakcharoen T, Ratanatamskul C, Chandrachai A. Factors affecting technology selection, techno–economic and environmental sustainability assessment of a novel zero-waste system for food waste and wastewater management. J Clean Prod 2021;314:128103.

[86] Skrydstrup J, Larsen SL, Rygaard M. Eco-efficiency of water and wastewater management in food production: a case study from a large dairy in Denmark. J Ind Ecol 2020;24(5):1101–12.

[87] Tangsubkul N, Beavis P, Moore SJ, Lundie S, Waite TD. Life cycle assessment of water recycling technology. Water Resour Manage 2005;19(5):521–37.

[88] Resende JD, Nolasco MA, Pacca SA. Life cycle assessment and costing of wastewater treatment systems coupled to constructed wetlands. Resour Conserv Recycl 2019;148:170–7.

[89] Rodriguez-Garcia G, Frison N, Vázquez-Padín JR, Hospido A, Garrido JM, Fatone F, et al. Life cycle assessment of nutrient removal technologies for the treatment of anaerobic digestion supernatant and its integration in a wastewater treatment plant. Sci Total Environ 2014;490:871–9.

[90] Zhang Y, Zhang C, Qiu Y, Li B, Pang H, Xue Y, et al. Wastewater treatment technology selection under various influent conditions and effluent standards based on life cycle assessment. Resour Conserv Recycl 2020;154:104562.

[91] Renou S, Thomas JS, Aoustin E, Pons MN. Influence of impact assessment methods in wastewater treatment LCA. J Clean Prod 2008;16(10):1098–105.

[92] Zang Y, Li Y, Wang C, Zhang W, Xiong W. Towards more accurate life cycle assessment of biological wastewater treatment plants: a review. J Clean Prod 2015;107:676–92.

[93] Razman KK, Hanafiah MM, Mohammad AW. An overview of LCA applied to various membrane technologies: progress, challenges, and harmonization. Environ Technol Innov 2022:102803.

[94] Baresel C, Yang J, Bornold N, Tjus K, Kanders L, Westling K. Direct GHG emissions from a pilot scale MBR-process treating municipal wastewater. Adv Clim Chang Res 2022;13(1):138–45.

[95] Spinelli M, Eusebi AL, Vasilaki V, Katsou E, Frison N, Cingolani D, et al. Critical analyses of nitrous oxide emission in a full scale activated sludge system treating low carbon-to-nitrogen ratio wastewater. J Clean Prod 2018;190 (20):517–24.

[96] Yang M, Peng M, Wu D, Feng H, Wang Y, Lv Y, et al. Greenhouse gas emissions from wastewater treatment plants in China: historical emissions and future mitigation potentials. Resour Conserv Recycl 2023;190:106794.

[97] Shi CL, Tian YY, Huang H, Wang S, Jiao B, Xie X. Summary of researches on accounting methods of carbon emissions from urban sewage treatment. Construct Sci Technol 2021;11:39–43.

[98] Mannina G, Cosenza A, Rebouças TF. A plant-wide modelling comparison between membrane bioreactors and conventional activated sludge. Bioresour Technol 2020;297:122401.

[99] Mannina G, Cosenza A, Rebouças TF. Aeration control in membrane bioreactor for sustainable environmental footprint. Bioresour Technol 2020;301:122734.

[100] Mannina G, Cosenza A. Quantifying sensitivity and uncertainty analysis of a new mathematical model for the evaluation of greenhouse gas emissions from membrane bioreactors. J Membr Sci 2015;475:80–90.

[101] Mannina G, Cosenza A, Ekama GA. Greenhouse gases from membrane bioreactors: mathematical modelling, sensitivity and uncertainty analysis. Bioresour Technol 2017;239:353–67.

[102] Mannina G, Cosenza A, Ekama GA. A comprehensive integrated membrane bioreactor model for greenhouse gas emissions. Chem Eng J 2018;334:1563–72.

[103] Mannina G, Chandran K, Capodici M, Cosenza A, Di Trapani D, van Loosdrecht MC. Greenhouse gas emissions from membrane bioreactors: analysis of a two-year survey on different MBR configurations. Water Sci Technol 2018;78 (4):896–903.

[104] Mannina G, Ni BJ, Ferreira Rebouças T, Cosenza A, Olsson G. Minimizing membrane bioreactor environmental footprint by multiple objective optimization. Bioresour Technol 2020;302:122824.

[105] Bailey JR, Bukhary S, Batista JR, Ahmad S. Renewable energy generation and GHG emission reduction potential of a satellite water reuse plant by using solar photovoltaics and anaerobic digestion. Water 2021;13(5):635.

[106] Juneidi SJ, Sorour MT, Aly SA. Proposed systematic approach for assessing different wastewater treatment plants alternatives: case study of Aqaba city (South Jordan). Alex Eng J 2022;61(12):12567–80.

[107] Liao X, Tian Y, Gan Y, Ji J. Quantifying urban wastewater treatment sector's greenhouse gas emissions using a hybrid life cycle analysis method—an application on Shenzhen city in China. Sci Total Environ 2020;745:141176.

[108] Sun Y, Garrido-Baserba M, Molinos-Senante M, Donikian NA, Poch M, Rosso D. A composite indicator approach to assess the sustainability and resilience of wastewater management alternatives. Sci Total Environ 2020;725:138286.

[109] Xi J, Gong H, Zhang Y, Dai X, Chen L. The evaluation of GHG emissions from Shanghai municipal wastewater treatment plants based on IPCC and operational data integrated methods (ODIM). Sci Total Environ 2021;797:148967.

[110] Wang X, Chen Y, Zhang J, Li X, Ren Y. Novel insights into the evaluation of submerged membrane bioreactors under different aeration intensities by carbon emission. Desalination 2013;325:25–9.
[111] Kong Z, Hao T, Chen H, Xue Y, Li D, Pan Y, et al. Anaerobic membrane bioreactor for carbon-neutral treatment of industrial wastewater containing *N*,*N*-dimethylformamide: evaluation of electricity, bio-energy production and carbon emission. Environ Res 2023;216:114615.
[112] Hernández-Sancho F, Molinos-Senante M, Sala-Garrido R. Economic valuation of environmental benefits from wastewater treatment processes: an empirical approach for Spain. Sci Total Environ 2010;408(4):953–7.
[113] Molinos-Senante M, Hernández-Sancho F, Sala-Garrido R. Economic feasibility study for wastewater treatment: a cost–benefit analysis. Sci Total Environ 2010;408(20):4396–402.
[114] Longo S, Hospido A, Lema JM, Mauricio-Iglesias M. A systematic methodology for the robust quantification of energy efficiency at wastewater treatment plants featuring data envelopment analysis. Water Res 2018;141:317–28.
[115] Sala-Garrido R, Molinos-Senante M, Hernández-Sancho F. Comparing the efficiency of wastewater treatment technologies through a DEA metafrontier model. Chem Eng J 2011;173(3):766–72.
[116] Boardman AE, Greenberg DH, Vining AR, Weimer DL. Cost–benefit analysis: concepts and practice. Cambridge: Cambridge University Press; 2018.
[117] Molinos-Senante M, Hernández-Sancho F, Sala-Garrido R. Cost–benefit analysis of water-reuse projects for environmental purposes: a case study for Spanish wastewater treatment plants. J Environ Manage 2011;92(12):3091–7.
[118] Wang HC, Cui D, Han JL, Cheng HY, Liu WZ, Peng YZ, et al. A_2O-MBR as an efficient and profitable unconventional water treatment and reuse technology: a practical study in a green building residential community. Resour Conserv Recycling 2019;150:104418.
[119] Andrade LH, Mendes FDS, Espindola JC, Amaral MCS. Reuse of dairy wastewater treated by membrane bioreactor and nanofiltration: technical and economic feasibility. Braz J Chem Eng 2015;32(3):735–47.
[120] Färe R, Grosskopf S, Weber WL. Shadow prices and pollution costs in US agriculture. Ecol Econ 2006;56(1):89–103.
[121] Gao T, Xiao K, Zhang J, Xue W, Wei C, Zhang X, et al. Techno–economic characteristics of wastewater treatment plants retrofitted from the conventional activated sludge process to the membrane bioreactor process. Front Environ Sci Eng 2022;16(4):49.
[122] Djukic M, Jovanoski I, Ivanovic OM, Lazic M, Bodroza D. Cost–benefit analysis of an infrastructure project and a cost-reflective tariff: a case study for investment in wastewater treatment plant in Serbia. Renew Sustain Energy Rev 2016;59:1419–25.
[123] Yang X, López-Grimau V. Reduction of cost and environmental impact in the treatment of textile wastewater using a combined MBBR–MBR system. Membranes 2021;11(11):892.
[124] Chang YT, Zhang N, Danao D, Zhang N. Environmental efficiency analysis of transportation system in China: a non-radial DEA approach. Energy Policy 2013;58:277–83.
[125] Zhang N, Xie H. Toward green IT: modeling sustainable production characteristics for Chinese electronic information industry, 1980–2012. Technol Forecast Soc Change 2015;96:62–70.
[126] Zhou P, Ang BW, Wang H. Energy and CO_2 emission performance in electricity generation: a non-radial directional distance function approach. Eur J Oper Res 2012;221(3):625–35.
[127] Charnes A, Cooper WW, Rhodes E. Measuring the efficiency of decision making units. Eur J Oper Res 1978;2(6):429–44.
[128] Banker RD, Charnes A, Cooper WW. Some models for estimating technical and scale inefficiencies in data envelopment analysis. Manage Sci 1984;30(9):1078–92.
[129] Tone KA. A slacks-based measure of efficiency in data envelopment analysis. Eur J Oper Res 2001;130(3):498–509.
[130] Malmquist S. Index numbers and indifference surfaces. Trabajos de estadística 1953;4(2):209–42. Spanish.
[131] Caves DW, Christensen LR, Diewert WE. The economic theory of index numbers and the measurement of input, output, and productivity. Econometrica J Econ Soc 1982:1393–414.
[132] Hernández-Sancho F, Molinos-Senante M, Sala-Garrido R. Energy efficiency in Spanish wastewater treatment plants: a non-radial DEA approach. Sci Total Environ 2011;409(14):2693–3269.
[133] Bick A, Tuttle JGP, Shandalov S, Oron G. Immersed membrane bioreactor (IMBR) for treatment of combined domestic and dairy wastewater in an isolated farm. Water Sci Technol 2005;51(10):327–34.
[134] Bick A, Yang F, Shandalov S, Oron G. Data envelopment analysis for assessing optimal operation of an immersed membrane bioreactor equipped with a draft tube for domestic wastewater reclamation. Desalination 2007;204(1–3):17–23.
[135] Castellet L, Molinos-Senante M. Efficiency assessment of wastewater treatment plants: a data envelopment analysis approach integrating technical, economic, and environmental issues. J Environ Manage 2016;167:160–6.
[136] Dong X, Zhang X, Zeng S. Measuring and explaining eco-efficiencies of wastewater treatment plants in China: an uncertainty analysis perspective. Water Res 2017;112:195–207.
[137] Hernández-Sancho F, Sala-Garrido R. Technical efficiency and cost analysis in wastewater treatment processes: a DEA approach. Desalination 2009;249(1):230–4.
[138] Huang R, Shen Z, Wang H, Xu J, Ai Z, Zheng H, et al. Evaluating the energy efficiency of wastewater treatment plants in the Yangtze River Delta: perspectives on regional discrepancies. Appl Energy 2021;297:117087.
[139] Sala-Garrido R, Hernández-Sancho F, Molinos-Senante M. Assessing the efficiency of wastewater treatment plants in an uncertain context: a DEA with tolerances approach. Environ Sci Policy 2012;18:34–44.
[140] Tong QY, Sun F, Dong X, Gao J. Statistical assessment and determinant analysis of the pollutant removal efficiencies of wastewater treatment plants. China Popul Resour Environ 2019;29(4):49–57. Chinese.
[141] Yang J, Chen B. Energy efficiency evaluation of wastewater treatment plants (WWTPs) based on data envelopment analysis. Appl Energy 2021;289:116680.
[142] Jiang H, Hua M, Zhang J, Cheng P, Ye Z, Huang M, et al. Sustainability efficiency assessment of wastewater treatment plants in China: a data envelopment analysis based on cluster benchmarking. J Clean Prod 2020;244:118729.
[143] Delanka-Pedige HMK, Munasinghe-Arachchige SP, Abeysiriwardana-Arachchige ISA, Nirmalakhandan N. Evaluating wastewater treatment infrastructure systems based on UN sustainable development goals and targets. J Clean Prod 2021;298:126795.
[144] Tjandraatmadja G, Sharma A, Grant T, Pamminger F. A decision support methodology for integrated urban water management in remote settlements. Water Resour Manage 2013;27(2):433–49.
[145] Ramezanianpour M, Sivakumar M. Multi-objective analysis for the selection of a sustainable greywater treatment system. Environ Eng Manage J 2019;18(1):159–70.
[146] Su X, Chiang P, Pan S, Chen G, Tao Y, Wu G, et al. Systematic approach to evaluating environmental and ecological technologies for wastewater treatment. Chemosphere 2019;218:778–92.
[147] Dewalkar SV, Shastri SS. Integrated life cycle assessment and life cycle cost assessment based fuzzy multi-criteria decision-making approach for selection of appropriate wastewater treatment system. J Water Process Eng 2022;45:102476.
[148] Viet ND, Jang A. Fertilizer draw solution index in osmotic membrane bioreactor for simultaneous wastewater treatment and sustainable agriculture. Chemosphere 2022;296:134002.
[149] Sawaf MBA, Karaca F. Different stakeholders' opinions toward the sustainability of common textile wastewater treatment technologies in Turkey: a case study Istanbul province. Sustain Cities Soc 2018;42:194–205.
[150] Fernández-López C, González García M, Sánchez-Lozano JM. Analysis of WWTPs technologies based on the removal efficiency of pharmaceutical activated compounds for water reuse purposes. A fuzzy multi-criteria decision making approach. J Water Process Eng 2021;42:102098.
[151] Agarwal S, Singh AP. Performance evaluation of textile wastewater treatment techniques using sustainability index: an integrated fuzzy approach of assessment. J Clean Prod 2022;337:130384.
[152] Wang S, Zou L, Li H, Zheng K, Wang Y, Zheng G, et al. Full-scale membrane bioreactor process WWTPs in East Taihu basin: wastewater characteristics, energy consumption and sustainability. Sci Total Environ 2020;723:137983.