Sustainability evaluation and implication of a large scale membrane bioreactor plant

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Abstract

Membrane bioreactor (MBR) technology is receiving increasing attention in wastewater treatment and reuse. This study presents an integral sustainability evaluation of a full scale MBR plant. The plant is capable of achieving prominent technical performance in terms of high compliance rate, low variation in effluent quality and high removal efficiency during long term operation. It is also more responsive to the new local standard with rigorous limits. However, electricity consumption is found to be the dominant process resulting in elevated life cycle environmental impacts and costs, accounting for 51.6% of the costs. As such, it is suggested to optimize energy use in MBR unit and implement sludge treatment and management. The prolonged membrane life span could also contribute largely to reduced life cycle environmental concerns and expenses. This study is of great theoretical significance and applicable value in guaranteeing the performance and sustainability of large scale MBR schemes.

Keywords: Membrane bioreactor (MBR); sustainability evaluation; energy use; sludge treatment; membrane life span

1. Introduction

Given the rapid pace of population growth, urbanization, industrialization, social and economic growth, climate change and living standard enhancement, water scarcity and water contamination have become prominent around the globe (Jiménez-Cisneros, 2014; UNWWAP, 2015). Water related issues are closely linked to human health, food, agriculture, energy, industrial activity and social stability (Oh and Lee, 2018). It is estimated that global water demand will increase by over 50% by the year 2050 which inevitably challenges water security for human society and the environment (UNDESA, 2015). In this regard, turning wastewater into a resource is an essential part to promote efficient use and move towards a more circular economy approach (Makropoulos et al., 2018). Notably, reclaimed water is being widely practiced in many water-scarce regions as an alternative water resource not only for non-potable applications but also for indirect and direct potable water reuses (Herman et al., 2017).

As for existing water treatment and reuse technologies, membrane bioreactors (MBRs) have been widely applied in more than 200 countries over the last 20 years because of their apparent strengths including the reduced chemical use, superior effluent quality, operational flexibility and reliability, lower excess sludge production and small footprint (Huang and Lee, 2015; Barreto et al., 2017). It is estimated that by 2019 more than 5 million m^3/d of wastewater will be treated by MBR plants worldwide and the global market value for MBR technology by that time is projected to reach 3 billion US dollars (Judd, 2016). Noticeably, the compound annual growth rate for MBRs during the period 2014-2019 is expected to be 17.4% in Asia-Pacific region, compared to 15%, 9.6% and 11.9% in globe, Europe and North America respectively (Krzeminski et al., 2017). It is predicted that China and Brazil will attain the fastest growth rates within the given forecast period (Abass et al., 2015). To be more specific, the number of large scale MBRs in China was about 200 by the end of 2017, with a capacity of over 4.5 million m^3/d and an expected market value of 1.3 billion US dollars (Xiao et al., 2014; Hao et al., 2018; Li et al., 2019).

The above-mentioned figures further drive a boom in scientific research and industrial applications of full scale MBRs in a sustainable pathway. Particularly, barriers with respect to energy consumption, high capital and operational costs, membrane fouling, membrane life span and full scale operational experiences are highly concerned and are likely to restrain MBR market expansion (Ma et al., 2017; Bagheri and Mirbagheri, 2018). A large quantity of studies have already devoted to membrane fouling control through the design of new configurations (Yan et al., 2015), the addition of granular media such as activated carbon, zeolite, sludge-based adsorbent, plastic barriers and quorum quenching enzymes (Iorhemen et al., 2017; Nahm et al., 2017) or the modification of membrane material by nanomaterials (Meng et al., 2017). Nowadays, multi-faceted challenges can no longer be solved by traditional limited scale and monotonous factor approaches. However, the sustainability assessment of MBRs in terms of technical, economic and environmental aspects is limited to few studies (Krzeminski et al., 2017).

Memon et al. (2007) and Ortiz et al. (2007) analyzed environmental aspect of water reuse systems at small scales and concluded that MBRs provoked higher environmental impact than conventional activated sludge and natural treatment systems (e.g. the reed beds and green roof water recycling system) due to higher energy demands. Similarly, Hospido et al. (2012) evaluated the environmental profiles of different MBR configurations on a pilot scale and identified an inverse relation between the environmental impact and technological complexity. Likewise, Ioannou-Ttofa et al. (2016) examined the environmental footprint of an MBR pilot unit without consideration of its sludge treatment and disposal. It is found that energy consumption and membrane unit's material are main impact contributors. Besides, Molinos-Senante et al. (2012) integrated the environmental assessment tool with cost-benefit approach of several technologies in small WWTPs and addressed the advantages of MBRs in terms of high effluent quality production. Nevertheless, there is a continuous doubt that whether experiences based on small or pilot scale MBR studies could actually offer a reliable view since the limited scale could negatively influence the energy performance (Fenu et al., 2010; Krzeminski et al., 2017; Salgot and Folch, 2018).

In addition, Høibye et al. (2008) proposed the concept of holistic assessment of advanced treatment technologies considering technical, economic and environmental aspects. Likewise, Plakas et al. (2016) suggested a multi-criteria analysis of advanced treatment technologies from a perspective of economic, environmental and social concerns. Furthermore, Hao et al. (2018) and Akhoundi and Nazif (2018) established corresponding models to evaluate the sustainability of MBRs. However, their effectiveness and applicability to full scale MBR applications need further research. Presently, there is still a significant challenge to identify the benefits of large scale MBRs for sustainable water reuse under risk and uncertainty while addressing the technical, economic and environmental implications. Hence, this study aims to investigate experiences of full scale MBR via a case study and evaluate performances and advances based on integrative analyses that involve multiple dimensions and indexes concurrently. The results can be beneficial to real practices and expansion of MBRs and offer a better understanding of the interlinkages among water, energy, nutrient and material towards maximum use and recovery.

2. Methods

2.1 Description of study area

In China, in addition to water shortage and high quality reclaimed water demand, the growth of MBR technology is also largely driven by ever stringent water discharge regulation, water reuse standard as well as potentials for upgrading existing WWTPs (Abass et al., 2015). Notably, the anaerobic/anoxic/oxic (AAO) MBR and its derivate processes have become prevailing process compared to oxic MBR and anoxic/oxic MBR for large scale municipal applications (Xiao et al., 2014). In AAO-MBR, the elimination of potential membrane foulants especially soluble extracellular polymeric substances in AAO process is regarded as beneficial to downstream MBR process (Krzeminski et al., 2017; Wang et al., 2017). Therefore, a large scale AAO-MBR plant with a capacity of 60,000 m^3/d in Kunming city of China was selected as a case study. For comparison, another adjacent water reclamation plant (WRP) with ACTIFLO (i.e. coagulation and flocculation) processes (210,000 m^3/d) was analyzed as a reference (Fig. 1).

2.2 Analytical methods

In this study, three dimensions, namely technical, environmental and economic aspects are taken into account for integral sustainability evaluation (Plakas et al., 2016; Akhoundi and Nazif, 2018). Social indexes (e.g. public awareness, acceptance and local development) are excluded from consideration since relevant quantitative data is not available in the study while qualitative information is likely to introduce bias (Chen et al., 2014). The functional unit of the study was the production of 1 m^3 of reclaimed water.

2.2.1 Technical aspect

Statistical analyses on wastewater influent and effluent quality parameters including biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), ammonia-nitrogen (NH_3-N), total nitrogen (TN) and total phosphorus (TP) were conducted by Microsoft Excel and the software package OriginPro 2017 version (developed by OriginLab Corporation, Northampton, USA). Afterwards, the technical performance of the plant was further evaluated by compliance rate, stability of effluent quality, removal efficiency and removal loading. The figures were plotted using the OriginPro 2017 version. The effluent compliance rate is shown in

Eq. (1):

$$Compliance\,rate = \frac{N_s}{N_t} \times 100\% \tag{1}$$

where, N_s refers to the number of samples of which the discharged effluent quality meet the corresponding standard values; N_t refers to the total number of samples.

The stability of effluent quality, removal efficiency and removal loading are described in Eqs. (2), (3) and (4) respectively:

$$Stability\ of\ effluent\ quality = \frac{SD_{eff}}{Mean_{eff}} \times 100\% \tag{2}$$

$$Removal\ efficiency = \frac{C_{in} - C_{eff}}{C_{in}} \times 100\% \tag{3}$$

$$Removal\ loading = C_{in} - C_{eff} \tag{4}$$

where, SD_{eff} is the standard deviation of effluent quality; $Mean_{eff}$ is the average value of effluent quality; C_{in} is the concentration of influent parameter; C_{eff} is the concentration of effluent parameter.

The Class 1A water quality of Chinese national discharge standard of pollutants for municipal WWTPs (GB18918- 2002) has a minimum requirement of BOD_5 < 10 mg/L, COD_{Cr} < 50 mg/L, SS < 10 mg/L, TN < 15 mg/L, TP < 0.5 mg/L and NH_3-N < 5 mg/L. It is specified that effluent meeting Class 1A level can be reused in a recreational or scenic environment that has less diluting capacity (Sun et al., 2016). Moreover, to further reduce pollutant loadings and improve local water environment, the government is planning to release a new local discharge standard of WWTPs. The forthcoming standard will stipulate more rigorous limits with BOD_5 < 6 mg/L, COD_{Cr} < 30 mg/L, TN < 10 mg/L, TP < 0.3 mg/L and NH_3-N<1.5 mg/L.

2.2.2 Environmental aspect

Life cycle assessment (LCA) is conducted to assess the environmental aspects of two WRPs thoroughly. As depicted Fig. 1, the system boundaries include all wastewater treatment process units. All the energy and mass input and output flows, all the chemicals used at different treatment units and sludge treatment are considered. The construction and demolition phase of the life cycle of WRPs were excluded from the system boundary because of limited environmental impacts and lack of data (Qin et al., 2018; Sun et al., 2018). The detailed modeling and calculation processes are performed by GaBi CML2001 LCA software. Fifteen midpoint categories are included in the methodology: aquatic acidification, aquatic ecotoxicity, aquatic eutrophication, carcinogens, global warming potential, ionizing radiation, land occupation, mineral extraction, non-carcinogens, non-renewable energy, ozone layer depletion, photochemical oxidation, respiratory effects, terrestrial acidification/ nitrification, and terrestrial ecotoxicity. For ease of comparison, the midpoint categories per unit of emission were further normalized by the per capita world impact for the year 2000 (Sleeswijk et al., 2008). The life cycle inventory data are shown in Table 1. All the inputs, outputs and emissions are presented based on the functional unit of 1 m^3 of treated water.

2.2.3 Economic aspect

For economic assessment, life cycle cost (LCC) is performed where the capital and operation and maintenance costs of the WRPs are taken into account (Table 1) and calculated as expense per functional unit. According to the actual design planning, the life span of plant A and B are measured as 23 and 25 years respectively. Capital costs include the costs associated with civil works, equipment and land acquisition, while operation and maintenance costs include the costs related to energy consumption, chemicals consumption, MBR membrane material consumption (i.e. polyvinylidene fluoride, PVDF) and pollution discharge (i.e. water emission in terms of COD, SS, NH_3-N, NO_3-N and TP and sludge disposal). All these costs data are sourced from field investigation.

3. Results and discussion

3.1 Technical performance

Fig. 2 presents the concentrations of six crucial water quality parameters in influent and effluent of both WRPs. Notably, the sources of wastewater influent are mainly from municipal sewage streams, as well as a possible mixture of industrial wastewater, stormwater and surface water. Hence, compared with pure industrial wastewaters, the influent quality of two plants is relatively stable and has low contamination of organic matters and nutrients. Since the two plants are located in adjacent areas, their influent quality possesses similar contamination levels. The concentrations of all water quality parameters in effluent of both plants were decreased largely. However, since different treatment processes were applied in these two plants, their technical performances with respond to effluent quality are further evaluated. Additional details of the parameter values are given in Appendix A.

Presently, the discharged effluent of both plants follows the Class 1A level of national standard and is subsequently supplied for scenic environmental uses in Dianchi Lake. As a consequence, overall effluent limit compliance rates should be maintained at acceptable levels. As can be seen from Fig. 3, the MBR plant performs well with all water quality parameters achieve 100% compliance with the national guideline limits. Comparatively, lower compliance rates were observed in plant A (ACTIFLO) where only BOD_5 and COD_{Cr} concentrations reached at 100% compliance rates.

Based on the new local guideline limits, corresponding compliance rates of both plants are calculated. As shown in Fig. 3, the effluent quality of MBR plant will still achieve a satisfactory effect with an average compliance rate of 97% in all parameters except for TN. In comparison, when improving the discharge limit, effluent compliance rates of water quality parameters in plant A (ACTIFLO) will decrease sharply, suggesting additional treatment need to be conducted. Nevertheless, under this scenario, both plants should pay attention to extra nitrogen removal.

Besides, the operation and management of WRPs is likely to be affected by multiple factors, such as shock loadings of influent quality (e.g. toxic and harmful wastewater intrusion), sudden failure of one treatment unit and microbial reactivation after disinfection. A key capability of the WRP is to ensure the stability of effluent quality despite of complex and varying situations. According to Fig. 3, MBR plant produces a more stable effluent quality due to less variations in concentrations of water quality parameters. As for removal efficiency, the MBR plant exhibits better performances than plant A in terms of BOD_5, COD_{Cr}, SS, TN, TP and NH_4-N removal. Particularly, the removal efficiencies of all water quality parameters in MBR plant are above 95% expect for TN. However, the removal efficiency of TN is only 58.3% and 69.9% in plant A and B respectively. Similar phenomenon was detected by Zhang et al. (2016) which shows that nearly 90% of WWTPs demonstrated poor removal efficiency for nitrogen and phosphorus, especially the TN. Furthermore, removal loadings per unit of treated effluent of plant A is slightly higher than that of MBR plant.

3.2 Environmental performance

The LCA midpoint results and dominant contributing processes are summarized in Table 2. In plant A, dominant processes that contributed to the environmental impact are electricity, sludge landfill, direct emissions and PAC production. Comparatively, for the MBR plant, dominant processes in environmental performance are electricity, sludge landfill, PVDF consumption and direct emissions. The results are in accordance with other reported findings by Ioannou-Ttofa et al. (2016). Particularly, sludge landfill is a major factor contributing significantly to most impact categories. This indicates that the current treatment method of sludge via direct landfill can be environmental unfriendly. Alternative treatment approaches of sludge such as anaerobic digestion and advanced oxidation can be considered (Pang et al., 2018). Besides, to maximize energy recovery and move towards energy positive wastewater treatment, novel sludge treatment technologies such as biosolids gasification, free ammonia and free nitrous acid (e.g.

HNO_2) are also being increasingly explored and analyzed (Wang et al., 2016; Gikas, 2017; Wang, 2017).

For both of the WRPs, electricity was also an important contributor to the overall environmental burden. Noticeably, for the MBR plant, since the membranes need to be replaced every four years, environmental impacts related to MBR membrane material consumption (i.e. PVDF) should be addressed. When the membrane life span can be possibly extended, such as through effective membrane fouling control or employment of new material, membrane material associated effects can be reduced to a large extent. For instance, when the membrane life span can be extended to 8 years, the life cycle environmental impact of the main PVDF affected category (i.e. carcinogens) can be reduced to 0.0195 compared with 0.0274 in current circumstances (Table 2). Further, Ioannou-Ttofa et al. (2016) claimed that the adoption of a more environmentally friendly membrane material (i.e. ethylene propylene diene monomer) is likely to reduce life cycle costs and introduce less impact on the environment.

Moreover, the normalized midpoint results are illustrated in Fig. 4. It is observed that categories including aquatic acidification, aquatic ecotoxicity, respiratory effects, non-carcinogens, terrestrial acidification/nitrification, carcinogens and global warming potential are major components of the overall environmental impact. By contrast, other LCIA categories play relatively minor roles in assessing environmental performance of the WRPs and can thusly be neglected. Further investigation indicates that sludge landfill and electricity are the most significant sides that are likely to result in elevated aquatic ecotoxicity. To be more specific, the contributions of different treatment processes and categories in the MBR plant to the ultimate environmental impact are further explored (Fig. 5). It is apparent that the MBR unit alone is likely to introduce environmental burdens in terms of ozone layer depletion, mineral extraction and ionizing radiation. Apart from sludge landfill and electricity consumption, the production, use and demolishment of PVDF in the MBR plant, also contribute largely to the categories of carcinogens and non-carcinogens.

Besides, sensitivity analysis was also conducted to identify the main influences that can affect the LCA results. Table 3 presents the results of a 10% variations in main contributing factors. It can be found that for the MBR plant, a 10% variation in electricity consumption can lead to a change of respiratory effect, aquatic acidification, global warming, and terrestrial acidification/nitrification potentials of 7.89%, 6.16%, 5.60% and 4.17% respectively. Likewise, varying the amount of sludge for landfill disposal can also greatly influence major environmental impact categories. The results indicate that if electricity consumption of the MBR plant can be cut down to a certain degree or electricity can be produced largely by renewable energies (e.g. solar and hydroelectricity) rather than fossil fuel consumptions, overall environmental impact of the MBR plant can be reduced considerably (Ortiz et al., 2007; Ioannou-Ttofa et al., 2016) and can be even lower than plants equipped with conventional treatment technologies. Similarly, more effort should be paid for sludge treatment. It is noteworthy that for aquatic and terrestrial acidification concerns, attentions should be paid to TN and NH_3-N concentrations of the effluents as well. Additionally, as shown in Table 4, uncertainty analysis was performed to determine the degree of confidence in LCA results.

3.3 Economic performance

According to Fig. 6, the LCC results suggest that the expense per functional unit of the MBR plant is slightly higher than that of Plant A. The largest increase of expense is attributed to the energy consumption of MBR, which occupies about 51.6% of the total LCC expenses. Similar results were demonstrated in Abass et al. (2015). Presently, the electricity consumption of the MBR plant (60,000 m^3/d) is estimated to be 0.47-0.56 kWh/m^3. Given the economy of the scale, the energy consumption of a larger scale MBR plant is expected to be lower than the current plant. For instance, the electricity consumption of two MBR plants in China (i.e. 150,000 and 200,000 m^3/d) is reported as 0.33 and 0.31-0.46 kWh/m^3 respectively (Li et al., 2019). Nevertheless, the costs of MBR plant in the aspects of pollution discharge, chemical inputs and capital investment are almost the same as Plant A. Hence, the MBR plant would be more competitive if the energy consumption can be reduced further (Cashman et al., 2018).

3.4 Overall consideration and evaluation

The technical performance of the MBR plant exhibit distinct strengths over traditional treatment technologies with respect to compliance rate, stability of effluent quality and removal efficiency and the facility is more adaptive to ever stringent standard limits. This is vital for the long-term operation of wastewater treatment and reuse projects (Xiao et al., 2014). However, from the life cycle perspective, superior technical performance of the existing large scale MBR plant is also accompanied by considerable environmental burdens and high costs. The relatively high electricity consumption and direct landfill of sludge become significant obstacles to the continuous application and expansion of MBR technologies. Their environmental impacts can be possibly reduced by enhanced aeration efficiency, use of high-flux membrane material, change of energy production models and adoption of sound sludge treatment and energy recovery facilities.

While the strengths and weaknesses of MBR schemes need to be further weighed, it is worth noting that the values of different dimensions in technical, environmental and economic aspects are normally manifested in varied forms (e.g. quantitative calculations and qualitative estimations) with different unit scales (e.g. monetary, volumetric and concentration unit). To attain a gross result of the comprehensive evaluation, normalization, weighting and aggregation processes of multiple dimensions can be further considered and performed (Chen et al., 2014; Wang et al., 2018). The commonly used aggregation methods include the data envelopment method (Lorenzo-Toja et al., 2015; Castellet and Molinos-Senante, 2016), analytic hierarchy process (Molinos-Senante et al., 2014; Kalbar et al., 2013), fuzzy comprehensive evaluation method (Tan et al., 2014), etc. However, although the data envelopment method does not require weight distribution, there are requirements on the number of evaluation objects. Besides, both analytic hierarchy process and fuzzy comprehensive evaluation methods require weight assignment. These two methods are highly subjective since a series of subjective factors are involved (e.g. personal perception, natural environment, economic condition, local policies, local and specific assumptions, etc.) (Ouyang et al., 2015). Consequently, due to limited local data on the relative importance or relevance of multiple factors, this study only conducts an initial single-factor evaluation in multiple aspects. The overall comprehensive evaluation can be performed in the future when more information on priorities of multiple factors and managerial preferences is available.

4. Conclusions

This study conducted a sustainability evaluation of a full scale MBR plant in terms of technical, environmental and economic aspects. Statistical analyses on six crucial water quality parameters indicate that MBR technology could achieve satisfactory technical performance and is adaptive to increasingly stringent standard limits. Aquatic ecotoxicity is identified as the major category contributing to overall environmental impact. Continuous understanding on energy use pattern, sludge treatment and membrane material can facilitate the implementation of sound management strategies. Aggregation of multiple dimensions to achieve a general score can be further performed when additional field information are available.

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6. Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version.

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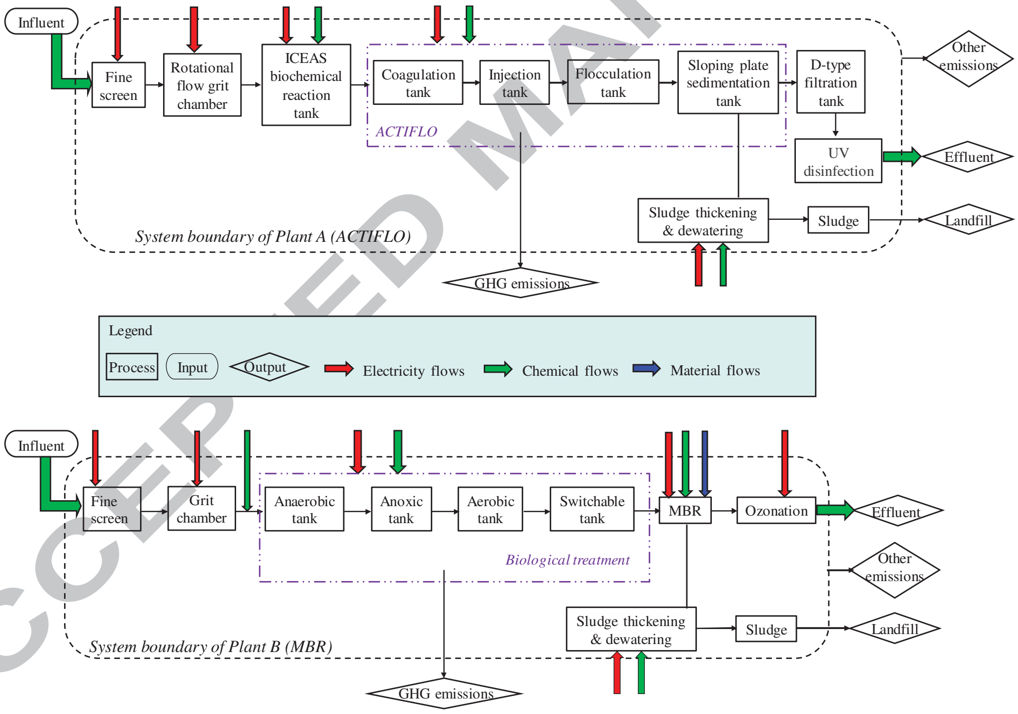

Fig. 1 Generic flow chart and system boundary of two WRPs

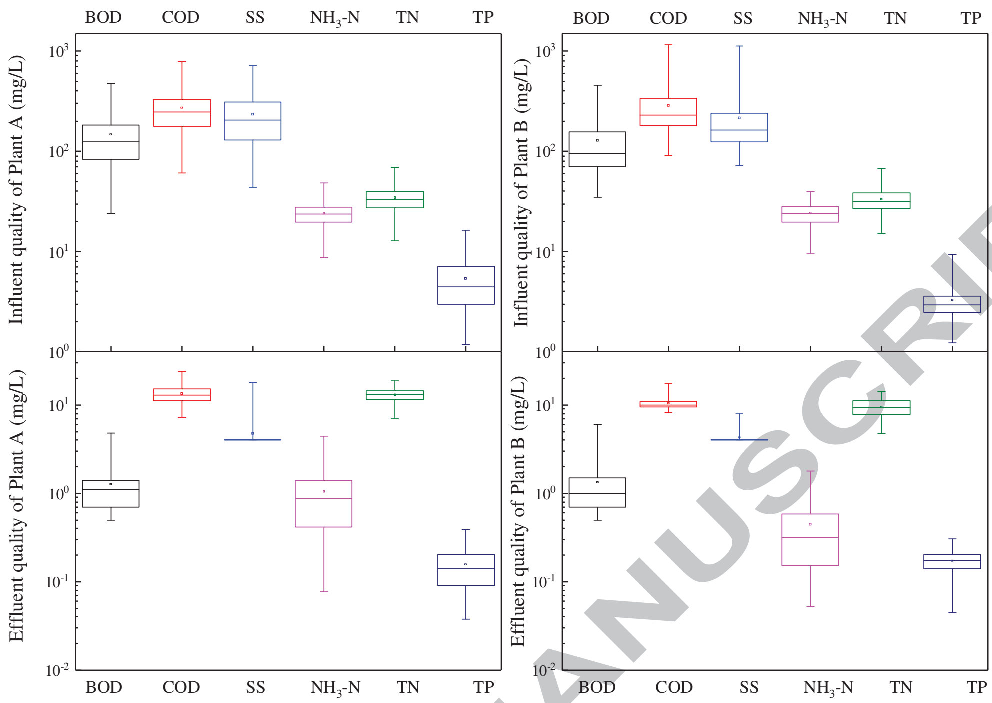

Fig. 2 Influent and effluent water quality of plant A (ACTIFLO) and plant B (MBR) during 2014

Notes: For each water quality parameter, the bottom and top of the box represent the 25th and 75th percentile respectively, the band and the hollow square represent the 50th percentile (median) and the mean value.

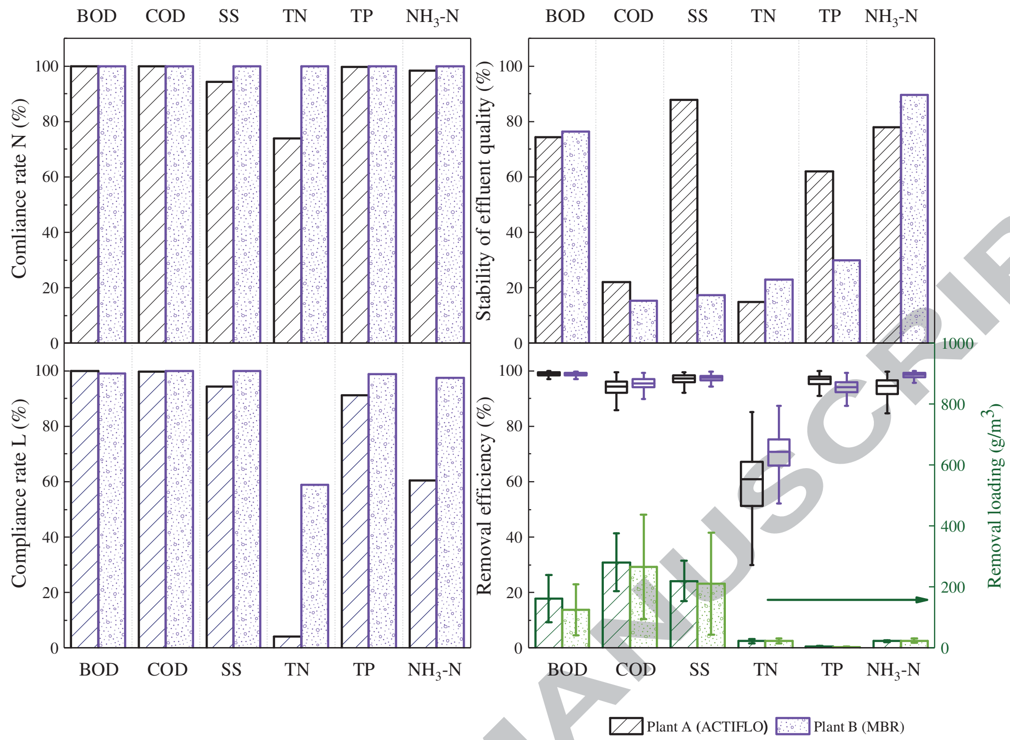

Fig. 3 Technical performances of two WRPs during 2014

Notes: N refers to compliance rates that correspond to Class 1A water quality of Chinese national discharge standard of pollutants for municipal WWTPs (GB18918-2002). L refers to compliance rates that correspond to forthcoming local standard in Kunming.

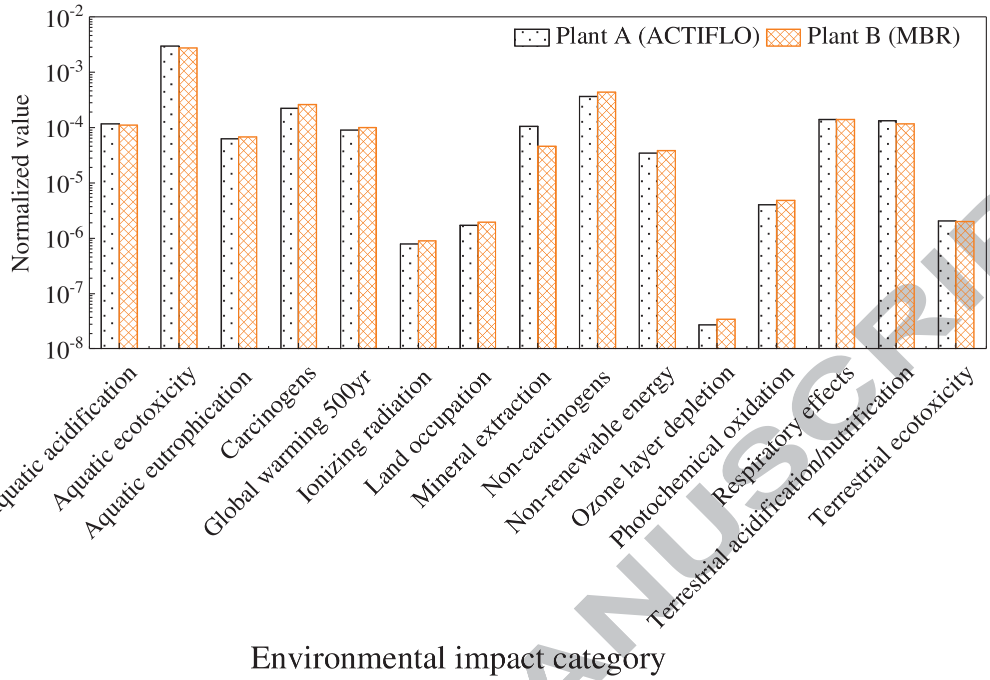

Fig. 4 Normalized midpoint scores for the full LCA of two WRPs

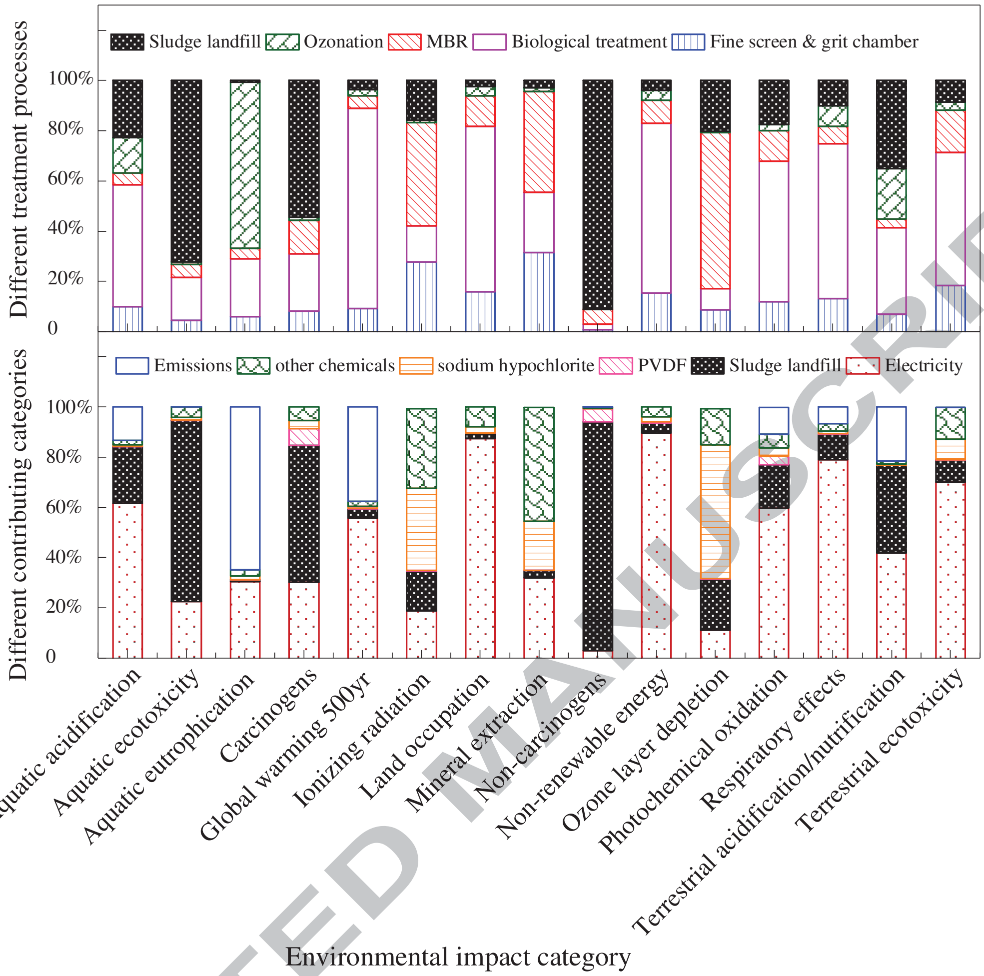

Fig. 5 Contributions of different treatment processes and categories to midpoint scores in LCA of plant B (MBR)

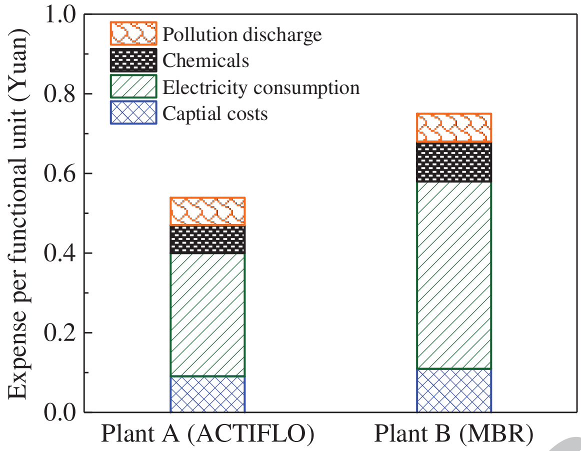

Fig. 6 Life cycle cost of two WRPs

Table 1. The LCIA inventory for life cycle assessment of two WRPs

Material inputs	Unit	Plant A (ACTIFLO)	Plant B (MBR)	Source	Price Yuan/kg
Electricity	kwh	0.305	0.469	Field investigation	1
Tap water	g	196.78	99.41	Field investigation	0.00485
PAC (poly aluminium chloride)	g	68.95	6.47	Field investigation	0.67
Quartz sand (small size)	g	0.564		Field investigation	2.066
PAM (polyacrylamide)	g	0.6		Field investigation	21
PFS (polymeric ferric sulfate)	g	5.58		Field investigation	1.26
$FeSO_4$	g		18.98	Field investigation	0.39
NaOH	g		0.09	Field investigation	4.5
NaClO	g		8.15	Field investigation	1.2
Citric acid	g		0.49	Field investigation	9
PVDF (membrane material)	g		0.53	Field investigation	141.51
Gas emissions					
CO_2	g	346.41	362.97	Calculated from Stoichiometry	
N_2O	g	0.1329	0.134	Calculated from Stoichiometry	
CH_4	g	0.102	1.058	Calculated from Stoichiometry	
Water emissions[*,#]					
COD	g	13.34 [9,19.3]	10.48 [9,13.1]	Field investigation	1.4
SS	g	4.76 [4,8]	4.27 [4,6]	Field investigation	0.35
NH_3-N	g	1.05 [0.14,2.8]	0.45 [0.082,1.28]	Field investigation	1.75
NO_3-N	g	11.88 [8.55,13.9]	8.99 [5.85,11.42]	Field investigation	
Total P	g	0.16 [0.056,0.33]	0.17 [0.085,0.262]	Field investigation	5.6
Solid waste[*]					
Sludge (80% moisture content)	g	465.83 [239.8,754]	532.22 [297.1,794]	Field investigation	0.09

Notes: *The numbers in the bracket are the ends of 95% confidence interval of water and solid wastes generation. #The price of the pollutants in the water emissions is taken from The Law of China Environmental Protection Tax. Gas emissions calculated from stoichiometry refer to equations described by Snip (2010) and Wang et al. (2017).

Table 2. The life cycle environmental impacts of two WRPs and the main contributors of different impact categories

Environmental impact category	Unit	Plant A (ACTIFLO) Value	Plant A (ACTIFLO) Dominating contributing processes	Plant B (MBR) Value	Plant B (MBR) Dominating contributing processes
Aquatic acidification	kg SO_2-Eq. to air	2.06E -02	Direct emissions (71.43%) + Electricity (15.11%)	7.60E -03	Electricity (62.81%) +Sludge landfill (32.71%)
Aquatic ecotoxicity	kg TEG-Eq. to water	4.53E +03	Sludge landfill (61.55%) +PAC production (25.46%)	5.19E +03	Sludge landfill (76.63%) +Electricity (16.75%)
Aquatic eutrophication	kg PO_4-Eq. to water	7.48E -04	Direct emissions (65.44%) + Electricity (22.26%)	3.08E -04	Electricity (83.08%)
Carcinogens	kg C_2H_3Cl-Eq. to air	1.12E -02	Sludge landfill (60.30%) +Electricity (22.03%)	2.74E -02	PVDF (47.32%) +Sludge landfill (35.11%)
Global warming 500 yr	kg CO_2-Eq. to air	9.07E -01	Electricity (41.25%) +Direct emissions (38.25%)	8.27E -01	Electricity (69.51%) +Direct emissions (15.92%)
Ionizing radiation	Bq C-$_{14}$-Eq. to air	4.28E -01	PAC production (56.33%) +Sludge landfill (18.26%)	5.34E -01	Sodium hypochlorite (29.07%) +Iron sulfate (21.56%)+Sludge landfill (20.92%)
Land occupation	m^2*yr-Eq.	5.88E -03	Electricity (67.68%) +PAC production (28.48%)	7.11E -03	Electricity (86.01%)
Mineral extraction	MJ surplus	3.06E -02	PAC production (82.54%)	1.41E -02	Electricity (32.14%) +Iron sulfate (25.06%) +Sodium hypochlorite (18.57%)
Non-carcinogens	kg C_2H_3Cl-Eq. to air	7.45E -02	Sludge landfill (95.86%)	1.70E -01	Sludge landfill (60.13%) +PVDF (38.15%)
Non-renewable energy	MJ	5.29E +00	Electricity (67.00%) +PAC production (27.37%)	6.76E +00	Electricity (80.49%)

Ozone layer depletion	kg CFC-11-Eq. to air	5.75E-09	PAC production (55.88%) + Sludge landfill (25.66%)	8.08E-09	Sodium hypochlorite (45.85%) +Electricity (26.06%)
Photochemical oxidation	kg C_2H_4-Eq. to air	5.19E-05	Electricity (46.28%) +PAC production (29.09%)	9.01E-05	Electricity (40.92%) +PVDF (36.39%)
Respiratory effects	kg PM2.5-Eq. to air	3.54E-03	Direct emissions (71.43%) + Electricity (15.11%)	1.32E-03	Electricity (77.31%)
Terrestrial acidification/nutrification	kg SO_2-Eq. to air	1.43E-01	Direct emissions (80.92%)	3.65E-02	Sludge landfill (52.35%) +Electricity (44.02%)
Terrestrial ecotoxicity	kg TEG-Eq. to soil	2.52E+00	Electricity (45.12%) +PAC production (41.24%)	2.67E+00	Electricity (65.47%) +Sludge landfill (11.39%)

Table 3. The sensitivity of main contributors in LCA of two WRPs

		Environmental impact category													
Contributing category	Variance	Aquatic acidification		Aquatic ecotoxicity		Carcinogens		Global warming 500yr		Non-carcinogens		Respiratory effects		Terrestrial acidification /nutrification	
		Plant A	Plant B	Plant A	Plant B	Plant A	Plant B	Plant A	Plant B	Plant A	Plant B	Plant A	Plant B	Plant A	Plant B
PAC	10%	1.28%	–	2.82%	–	1.77%	–	1.63%	–	0.22%	–	2.33%	–	0.81%	–
Ammonia	10%	2.56%	1.14%	0%	0%	0%	0%	0%	0%	0.01%	0%	1.05%	0.44%	3.74%	1.83%
Electricity	10%	4.03%	6.16%	1.38%	2.24%	2.43%	3.03%	4.17%	5.60%	0.25%	0.30%	5.43%	7.89%	2.49%	4.17%
Sludge landfill	10%	1.90%	2.26%	5.74%	7.23%	5.61%	5.44%	0.35%	0.37%	9.51%	9.10%	0.90%	1.02%	2.68%	3.50%
Sodium hypochlorite	10%	–	0.05%	–	0.10%	–	0.34%	–	0.08%	–	0.03%	–	0.10%	–	0.05%
PVDF	10%	–	0.01%	–	0.03%	–	0.67%	–	0.02%	–	0.52%	–	0.03%	–	0.01%

Notes: Plant A (ACTIFLO); Plant B (MBR).

Table 4. The uncertainty analysis of the LCA midpoint results of two WRPs

Environmental impact category	Unit	Plant A (ACTIFLO)		Plant B (MBR)	
		Mean value	95% interval*	Mean value	95% interval*
Aquatic acidification	kg SO_2-Eq. to air	8.83E-03	[6.30E-03, 1.14E-02]	1.04E-02	[7.75E-03, 1.31E-02]
Aquatic ecotoxicity	kg TEG-Eq. to water	4.33E+03	[3.24E+03, 5.42E+03]	7.21E+03	[3.27E+03, 1.12E+04]
Carcinogens	kg C_2H_3Cl-Eq. to air	1.07E-02	[8.07E-03, 1.33E-02]	2.04E-02	[1.09E-02, 2.99E-02]
Global warming 500yr	kg CO_2-Eq. to air	9.10E-01	[8.95E-01, 9.25E-01]	1.04E+00	[9.87E-01, 1.09E+00]
Non-carcinogens	kg C_2H_3Cl-Eq. to air	6.94E-02	[4.15E-02, 9.73E-02]	1.65E-01	[6.40E-02, 2.66E-01]
Respiratory effects	kg PM2.5-Eq. to air	1.29E-03	[1.12E-03, 1.46E-03]	1.45E-03	[1.25E-03, 1.65E-03]
Terrestrial acidification/nutrification	kg SO_2-Eq. to air	5.09E-02	[3.09E-02, 7.09E-02]	5.99E-02	[3.94E-02, 8.04E-02]

Note: *The numbers in the bracket are the ends of 95% confidence interval of the uncertainty analysis.

Highlights

- Sustainability evaluation is vital for MBR technology application and expansion
- Technical performance of a full scale MBR plant is analysed and discussed
- Major contributors to life cycle environmental impact and cost are identified
- Management strategies are proposed for further improvement of MBR sustainability
- Aggregation of multiple dimensions and additional data collection is recommended