Abstract
Water reuse is believed to be a sustainable solution to overcome the scarcity of freshwater. Aerobic and anaerobic membrane bioreactors are becoming an effective technology for wastewater treatment and reuse. Aerobic membrane bioreactors show good nutrient removal, whereas those that are anaerobic have nutrient-rich effluent, enabling the direct agricultural use of the effluent. As a result, the end use will dictate the potential environmental impacts of the bioreactor’s application. Therefore, with the consideration of the end use (i.e., discharge or reuse) of the effluent, this study aimed to compare the environmental and economic impacts associated with full-scale aerobic and anaerobic membrane bioreactors for municipal wastewater treatment under different end use scenarios using life cycle assessment and cost analysis. The results of these analyses show that anaerobic bioreactors have greater environmental impacts and life cycle cost than aerobic bioreactors in the discharge scenario due to the incorporation of a biological nutrient removal system. In the reuse scenario, anaerobic membrane bioreactors have lower impacts that are attributable to the offset of the nutrients required for crops, and the potential benefits vary depending on the types of crops receiving the reclaimed water. Integrating anaerobic membrane bioreactors with agricultural fertigation resulted in effluent water nitrate concentrations (after crop uptake and soil treatment) of <2 mg L\(^{-1}\) in most U.S. states. This indicated that the use of the anaerobic membrane bioreactors effluent for fertigation could be a win-win solution to both irrigation water shortage and high environmental impact associated with nutrient removal.

1 | INTRODUCTION

As freshwater becomes scarce, many regions of the world are facing a shortage of clean drinking water and irrigation water for food production. Water reuse or recycling is believed to be a sustainable solution for the water crisis (Miller, 2006). In the past decades, reclaimed water has gradually been accepted for nonpotable purposes (e.g., pond filling, lawn irrigating, or agricultural irrigating; Chen, Lu, Jiao, Wang, & Chang, 2013).
An important factor affecting the use of reclaimed water is its water quality, including nutrient level (Chen et al., 2013). Nutrients in domestic wastewater or treated water, including nitrogen (N) and phosphorus (P) in general, are pollutants of primary concern for aquatic ecosystem protection due to their potential to cause the eutrophication of receiving water bodies (Conley et al., 2009). The membrane bioreactor (MBR) has been demonstrated as an effective technology for wastewater reclamation (Melin et al., 2006). Some MBR configurations can achieve >90% removal of total N (TN) and total P (TP), with effluent TN and TP concentrations as low as 4.2 and 0.17 mg L$^{-1}$ (Fu, Yang, Zhou, & Xue, 2009; Kimura, Nishisako, Miyoshi, Shimada, & Watanabe, 2008), which is acceptable for discharge in some states (e.g., effluent limits of 13 mg TN L$^{-1}$ and 1 mg TP L$^{-1}$ from municipal sewage treatment facilities in Florida; USEPA, 2016).

Membrane bioreactors, as a combination of a suspended growth bioreactor and a perm-selective or semipermeable membrane, can provide the benefits of both biological treatment and physical solid–liquid separation. The application of MBRs for wastewater treatment, however, was hindered by their high expense (e.g., the high cost of membranes and energy consumption for operation; Judd, 2017). Currently, the MBR is commonly adopted for “sewer mining,” a strategy to recycle water from the sewage collection system before it reaches the wastewater treatment plant. A community-scale MBR system can reclaim treated graywater for nondrinking purposes like landscape irrigation and toilet flushing, reducing potable water use by 65% (Cascadia Green Building Council, 2011).

Given the oxygen requirements of microorganisms in the bioreactor, MBRs can generally be categorized into aerobic (AeMBR) and anaerobic MBRs (AnMBR). An AeMBR has a similar biological process to the conventional activated sludge treatment and has been widely used for municipal wastewater treatment since the early 1990s (Gander, Jefferson, & Judd, 2000). An AeMBR has excellent performance on organic matter decomposition and the removal of pathogens and suspended solids (Gander et al., 2000). Similar to an AeMBR, an AnMBR has a high chemical oxygen demand (COD) removal efficiency, but it also reduces the overall energy demand for wastewater treatment, since no aeration energy is required (Ozgun et al., 2013). Moreover, anaerobic processes produce biogas as a potential energy source and mineralize nutrients in the form of ammonia and orthophosphate, enabling the direct agricultural use of the effluent for irrigation (Skouteris, Hermosilla, López, Negro, & Blanco, 2012). An AnMBR is generally operated at higher biomass concentration than an AeMBR, affecting reactor hydraulics and pumping. Thus, the use of AnMBRs is still limited due to problems such as low flux, membrane fouling (Chang, Le Clech, Jefferson, & Judd, 2002), and high capital and operational costs (Gander et al., 2000).

### Core ideas
- The end use of membrane bioreactor (MBR) effluent influences their sustainability.
- Anaerobic MBRs have greater impacts than aerobic MBRs if effluent is for discharge.
- The use of anaerobic MBRs’ effluent for fertigation makes it more sustainable.

To date, most AnMBRs have been implemented at laboratory or pilot scales (Bornare et al., 2014). Limited attempts have been made to implement full-scale AnMBRs in the centralized wastewater treatment plant (Christian et al., 2010; Veolia Water Technologies, 2018). Several studies have evaluated AnMBRs in terms of environmental performance (e.g., global warming potential, ecotoxicity, eutrophication), energy consumption, and costs (Lin, Guo, Shah, & Stuckey, 2016; Linares et al., 2016; Martin, Pidou, Soares, Judd, & Jefferson, 2011; Pretel, Moñino, et al., 2016; Pretel, Robles, Ruano, Seco, & Ferrer, 2013, 2014, 2016; Pretel, Shoener, Ferrer, & Guest, 2015; Smith et al., 2014; Tian, Ji, Wang, & Pierre, 2014; Wei, Harb, Amy, Hong, & Leiknes, 2014). Most of these studies focused on global warming potential and energy use of AnMBRs and generally concluded that AnMBRs were a sustainable solution for domestic wastewater treatment in terms of global warming and ecotoxicity control, energy saving, and operational cost saving.

There are limited studies comparing AnMBRs with other wastewater treatment technologies such as AeMBRs in terms of environmental impacts and costs (Lin et al., 2016; Pretel, Moñino, et al., 2016; Pretel, Robles, et al., 2016; Smith et al., 2014; Wei et al., 2014), most of which were conducted at pilot scale. Only two studies were found that compared AnMBRs with other wastewater treatment technologies at full scale using simulated data (Pretel, Moñino, et al., 2016; Smith et al., 2014). Pretel, Moñino, et al. (2016) evaluated the environmental impacts and costs of the various combinations of an AnMBR, an AeMBR, and a conventional activated sludge system at full-scale implementation. The treatment train including an AnMBR and anoxic/aerobic processes was thought the best option due to its high nutrient removal and low energy consumption. The contribution of AnMBR to the overall treatment performance, however, was not delineated, especially in terms of nutrient removal. Smith et al. (2014) compared the AnMBR with the AeMBR and the conventional activated sludge system, both of which were integrated with anaerobic digestion, and found that the AnMBR had lower environmental impacts, lower net energy use, and lower cost as a result of energy recovery. However, the nutrient-rich
FIGURE 1 The diagrams of treatment trains involving an aerobic membrane bioreactor (AeMBR, upper) and an anaerobic membrane bioreactor (AnMBR, below) (adapted from Smith et al., 2014). All the processes marked with dotted squares were excluded from the analysis. A²O, anaerobic–anoxic/oxic

effluent from membrane treatment was excluded from the system boundary, which might result in a bias toward the final evaluation of the environmental impacts and total costs.

Some studies have investigated the integration of AnMBRs with other post treatment processes (e.g., reverse osmosis) to lower the nutrient concentration in the effluent (Grundestam & Hellström, 2007; Pretel, Robles, et al., 2016). However, these studies focused on the effects of operating conditions (e.g., temperature, raw water characteristics) on the environmental impacts and costs. None of the previous studies evaluated the influence of the end uses of AnMBR effluent on environmental or economic impacts of the AnMBR-based treatment train.

This study aimed to compare the environmental and economic impacts associated with full-scale AeMBR and AnMBR treatment systems for municipal wastewater treatment under different end use scenarios using life cycle assessment (LCA) and life cycle cost analysis.

2 MATERIALS AND METHODS

2.1 Scenario development

As the AeMBR and AnMBR are the primary systems in this study, two basic scenarios of different end uses for each system were considered to compare the associated environmental impacts and cost: the treated wastewater will be either discharged to a surface water body or reused for fertigation (Figure 1). For the reuse scenarios, all the nutrients in the effluent were assumed to be used by the crops during irrigation. The influent wastewater for each treatment train was assumed to be medium-strength domestic wastewater (i.e., COD of 430 mg L⁻¹, 5-d biochemical oxygen demand [BOD₅] of 190 mg L⁻¹, TP of 7 mg L⁻¹, and TN of 40 mg L⁻¹ with 25 mg NH₄⁺–N L⁻¹ and no NO₃–N; more details in Supplemental Table S1), defined by Tchobanoglous, Stensel, Tsuchihashi, and Burton (2013) and Smith et al. (2014).
Some assumptions of the configurations in Figure 1 follow the work of Smith et al. (2014), including the grit removal efficiency, the membrane characteristics and maintenance, energy recovery using the combined heat and power system, COD removal efficiency, biogas production, solids residence time, sludge stabilization (anaerobic digester for the AeMBR and lime for the AnMBR), and dewatering (more details in the supplemental material). For sludge disposal, most of the sludge was transported for land application, and a small portion was incinerated. The same equipment and infrastructure included in all the scenarios were excluded from the study, such as the installation of the combined heat and power system and the incinerator and the construction of the landfill.

In the United States, the discharge criteria for N and P are mainly developed by each state. Most states have not yet established clear TN and TP quality criteria (USEPA, 2018), although some states have stringent numeric nutrient criteria (e.g., 1.87 mg TN L\(^{-1}\) for rivers and streams in north-central Florida; USEPA, 2010). Thus, the effluent from the AeMBR was assumed to be further treated to the acceptable level for discharge in Scenario AeD (where “Ae” indicates an AeMBR and “D” indicates discharge), through an additional nutrient removal process (shown in Figure 1). In the AeMBR reuse scenario (AeR), an amount of fertilizers will be added to the AeMBR effluent to achieve the same nutrient concentrations as that in the AnMBR effluent.

For the AnMBR reuse scenario (AnR), the effluent from the AnMBR was used directly for crop irrigation (i.e., fertigation). For the AnMBR discharge scenario (AnD), however, a biological nutrient removal (BNR) process was added to reduce the nutrient concentrations in the AnMBR effluent to the same level as the concentrations in the AeD effluent. The anaerobic–anoxic/oxic (A\(^2\)O) process was selected in this scenario due to its excellent performance, wide application, and readily available simulation models (Pai, 2007). The A\(^2\)O process includes three stages (i.e., anoxic, anaerobic, and a final aerobic stage that requires energy for aeration). The final effluent for discharge was assumed to meet the same discharge standard as Scenario AeD through an additional nutrient removal process (shown in Figure 1). Because the TN concentration in the AeMBR effluent (Scenario AeD) and the BNR effluent (Scenario AnD) are the same, the additional nutrient removal process would be the same for both scenarios. Thus, this study excluded the additional nutrient removal process in the evaluation for comparative purposes.

To reveal the reuse strategy of nutrient-rich AnMBR effluent, six subscenarios were developed to use the AnMBR effluent to fertigate different agricultural crop species. In addition to a golf course (Scenario AnR\(_G\)), the top five profitable crops in Florida (FDACS, 2013; i.e., oranges [Citrus sinensis (L.) Osbeck], strawberries [Fragaria \times\) ananassa (Weston) Duchesne ex Rozier], tomatoes [Solanum lycopersicum L.], peanuts [Arachis hypogaea L.], and sugarcane (Saccharum officinarum L.) were selected as the alternative agricultural end users of the AnMBR effluent (Scenarios AnR\(_O\), AnR\(_Sb\), AnR\(_T\), AnR\(_P\), and AnR\(_Sc\), respectively). The fertilization requirements (FDACS, 2015; IFAS Extension, 2015, 2017a, 2017b) were acquired (Supplemental Table S3) to calculate the mass difference in TN and TP that the crop needs and that the AnMBR effluent contains.

### 2.2 Scenario evaluation

All the scenarios were modeled to calculate the treatment efficiency throughout the designed treatment system (Supplemental Figure S1). The modeled results from Smith et al. (2014) were used to evaluate the effluent concentration directly from the unit process of AeMBR or AnMBR. They were used as the effluent quality for Scenarios AeD, AeR, and AnR. For Scenario AnD, the Activated Sludge Model no. 3 (ASM3) was used to estimate the effluent quality of the final discharge from the A\(^2\)O process. For the AnMBR reuse scenarios with different crops (AnR\(_X\)), a mass balance calculation was conducted to assess the nutrient remained after plant uptake. Then the Soil Treatment Unit Model (STU-MOD) (Geza, Lowe, & McCray, 2014) was used to simulate the process of soil treatment before the effluent reaches the groundwater.

### 2.3 Life cycle assessment

The LCA in this study followed the International Organization for Standardization (ISO) (2006) 14044 framework containing four primary steps: goal and scope definition, inventory analysis, impact assessment, and interpretation. The LCA examined the stages of material production, system construction, and operation and maintenance (O&M) (Supplemental Figure S2). The transportation and dismantling stages were excluded from the assessment since the impacts from those stages were considered negligible compared with the construction and O&M stages for typical wastewater treatment plants. Both the AeMBR and AnMBR systems were assumed to have a functional lifetime of 40 yr, and the membranes were assumed to be replaced every 10 yr (Smith et al., 2014).

The LCA was conducted with the SimaPro PhD software (version 8.0, PRé Consultants). Environmental impacts were calculated using the Tools for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) version 2.1 developed by the USEPA so that the results represented North American conditions. The impact categories analyzed for each scenario include global warming potential, smog, acidification, eutrophication, carcinogenics, noncarcinogenics, respiratory effects, ecotoxicity, and fossil
fuel depletion. The chosen functional unit was 0.22 m³ s⁻¹ (5 million gallons d⁻¹) of municipal wastewater treated to achieve minimum USEPA secondary treatment effluent standards: BOD₅ and total suspended solids concentration of 30 mg L⁻¹. All the environmental impacts were normalized by functional unit over the lifetime of the system. The data collected for the life cycle inventory were acquired from the literature, online retailers, ecoinvent, and the U.S. Life Cycle Inventory Database.

### 2.4 Life cycle cost analysis

The life cycle cost analysis in this study was performed assuming a lifetime of 40 yr. The costs of capital, operation, maintenance, and additional fertilizers were estimated from the literature and online retailers (Smith et al., 2014). The life cycle cost was reported as the net present value determined at the baseline discount rate of 5%. All the costs were discounted to present values according to Supplemental Equations S1–S3 and summarized in Supplemental Table S2.

### 3 RESULTS AND DISCUSSION

#### 3.1 Aerobic or anaerobic membrane bioreactors?

##### 3.1.1 Nutrient loading limits

For decision makers, whether the effluent pollutants exceed the restricted loading limits is probably the primary concern when they choose a treatment process. Since both AeMBRs and AnMBRs have good performance in terms of BOD and total suspended solids removal and pathogen controls, nutrient loadings from the MBR effluent would be a major consideration. Both the AeMBR and AnMBR treatment trains were simulated under the discharge and reuse scenarios following the steps shown in Supplemental Figure S1 (Geza et al., 2014). Table 1 summarizes the simulated effluent quality for each scenario.

Both the AeMBR and AnMBR treatment trains had acceptable COD and BOD₅ removal (>90%) and achieved secondary treatment standards. The AnMBR had a higher effluent COD concentration than the AeMBR due to a lower organic conversion under anaerobic conditions.

For the discharge scenario, the AeMBR decreased TN from 40 to 27.3 mg L⁻¹. About 32% of TN was removed by the assimilation of microorganisms in the activated sludge. The nitrification process occurring in the AeMBR converted NH₄⁺–N to NO₃⁻–N, which led to a low ammonia concentration in the effluent. Some organic N in the influent was broken down to form ammonia and then converted to nitrate or nitrite as well, increasing the effluent NO₃⁻–N to 27.0 mg L⁻¹. The AeMBR also reduced TP from 7 to 0.1 mg L⁻¹ by microorganism assimilation, meeting the discharge criteria. The AnMBR did not remove N and P (see AnR effluent) and kept almost all the nutrients in its effluent. However, with the help of the additional A²O process, the AnD scenario was able to achieve the same level of TN in the effluent as the AeMBR (see AnD effluent). For the discharge scenario, the effluent nutrients from the MBR systems are expected to be as low as possible. The AeMBR is obviously a better solution if the effluent from the treatment train is discharged to a surface water body as a de facto reuse. On the other hand, the AnMBR is not an ideal option, since an additional BNR process has to be added to reduce nutrient concentrations in the AnMBR effluent to the same level as the AeMBR effluent.

For the reuse scenario, the rich nutrients in the AnMBR effluent make it a valuable water source for agricultural irrigation. Some research has demonstrated that the membrane separation in AnMBRs effectively controls the pathogens from the domestic wastewater (Gander et al., 2000). This implies that the AnMBR effluent could be safe for direct irrigation (Skouteris et al., 2012) and save the artificial fertilizers to meet plant requirements for fertilization. For Scenario AeR, however, the fertilizers had to be added to the effluent from the AeMBR to meet the same level of N fertilization as the AnMBR. For the reuse scenario, the final effluent after the plant uptake and soil treatment should be safe for the groundwater environment. In other words, plants and soil can function as a post-treatment before the water reaches the groundwater table. For the AeR and AnR scenarios, the assumption was made that the amount of the nutrients in the effluent matches the plant requirement for fertilization, and all the nutrients in the effluent are absorbed by plants and removed in the soil during irrigation.

#### Table 1: Influent and effluent characteristics of aerobic membrane bioreactor (AeMBR) and anaerobic membrane bioreactor (AnMBR) treatment trains for the discharge and reuse scenarios

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>Influent</th>
<th>AeD (AeMBR effluent)</th>
<th>AnD (BNR effluent)</th>
<th>AeR (AeMBR effluent with fertilizer added)</th>
<th>AnR (AnMBR effluent)</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD</td>
<td>430</td>
<td>29.0</td>
<td>29.0</td>
<td>12.7</td>
<td>64.5</td>
</tr>
<tr>
<td>BOD₅</td>
<td>190</td>
<td>0.8</td>
<td>0.8</td>
<td>4.3</td>
<td>21.2</td>
</tr>
<tr>
<td>TN</td>
<td>40.0</td>
<td>27.3</td>
<td>40</td>
<td>27.3</td>
<td>40.0</td>
</tr>
<tr>
<td>NH₄⁺–N</td>
<td>25.0</td>
<td>0.3</td>
<td>13.0</td>
<td>1.1</td>
<td>40.0</td>
</tr>
<tr>
<td>NO₃⁻–N</td>
<td>0.0</td>
<td>27.0</td>
<td>27.0</td>
<td>26.2</td>
<td>0.0</td>
</tr>
<tr>
<td>TP</td>
<td>7.0</td>
<td>0.1</td>
<td>0.1</td>
<td>1.0</td>
<td>7.0</td>
</tr>
</tbody>
</table>

*COD, chemical oxygen demand; BOD₅, 5-d biochemical oxygen demand; TN, total N; TP, total P. BNR, biological nutrient removal.
FIGURE 2  The environmental impacts evaluated for the discharge scenarios (upper) and the reuse scenarios (below) in this study. For each impact category, the larger impact between the aerobic membrane bioreactor (AeMBR) and the anaerobic membrane bioreactor (AnMBR) was used to normalize the impact value, and the normalized impact was then expressed as a percentage accordingly. In scenario names, “Ae” and “An” represent AeMBRs and AnMBRs, and “D” and “R” indicate discharge and reuse scenarios, respectively.

3.1.2 | Environmental impacts

The results of the life cycle impact assessment for both the discharge and reuse scenarios are shown in Figure 2. For the discharge scenario, both AeD and AnD had a similar result in terms of eutrophication potential due to the same effluent quality. Effluent quality (nutrient concentrations, specifically) was the major contributor to the eutrophication potential. For the other categories, AnD had much greater impacts in almost every category than AeD. Energy was the dominant factor for impact categories, aside from carcinogenics and ecotoxicity. This is mainly due to the additional A²O process in the AnMBR treatment train, which contributed to high energy consumption (∼5,100 kW h for A²O, 3,900 kW h for AnMBR, and 7,900 kW h for AeMBR) despite the fact that there was 42% more energy recovered from the AnMBR than from the AeMBR. High electricity consumption associated with the aerobic stage in the A²O process contributed to greater impacts of global warming potential and fossil fuel depletion for AnD than for AeD. Chemicals contributed
approximately 22, 17, and 24% to the impacts of carcinogens, noncarcinogens, and ecotoxicity, respectively. The use of limes in sludge stabilization and dewatering contributed to the greater impacts of AnD in these categories. Membranes and their maintenance and replacement contributed to AnD 61% more than AeD in carcinogens, 60% more in noncarcinogens, and 64% more in ecotoxicity, since the AnMBR required more effort to address the membrane fouling issue.

For the general reuse scheme, AnR had less impact than AeR, due to the credits of the nutrients recovered in the effluent, especially in the eutrophication potential category. Since a BNR treatment was not implemented in the AnR scenario, the impacts associated with the operational phase decreased significantly (e.g., 39% less on global warming and 64% less on acidification, compared with AnD). Meanwhile, the impacts of the operational phase of AeMBR in the reuse scenario remained the same as those in the discharge scenario; however, the impacts associated with the added fertilizers for fertigation increased the impacts of AeR by about 5–90% on eutrophication, carcinogens, noncarcinogens, acidification, and smog formation. Compared with the LCA results of Smith et al. (2014), the largest contribution to carcinogens is from cogeneration, which is different from this study since the combined heat and power system was excluded from the analysis.

### 3.1.3 Costs

The net present values of AeMBR and AnMBR for both the discharge and reuse scenarios are shown in Figure 3. All the costs were discounted to the present value in 2018.

For the discharge scenario, AnD costed more than AeD over its functional lifetime. The major contributors to the high expenses of AnD included the construction of the extra BNR treatment unit, the membrane and its replacement, the addition of chemicals, and the O&M (e.g., membrane cleaning, sludge treatment and disposal). The AnMBR has longer solids residence time and the sludge is thicker than from the AeMBR, requiring more input on the membrane cleaning and replacement for AnD (48% more than AeD) due to the more serious membrane clogging issue. However, the biogas generated from the anaerobic digestion saved AnD some expense on energy consumption. In addition, AeD cost more in the construction and operation of the anaerobic digester for sludge treatment due to its process requirement, where AnD required expenses on lime for sludge stabilization in the operation phase. In terms of sludge treatment, AnD saved 22% than AeD in the lifetime of 40 yr.

For the reuse scenario, the addition of fertilizer made the cost of AeR much higher than that of AnR. Besides the fertilizer factor, the two systems cost almost the same. The saving from biogas energy generation (approximately US$0.8 million for AeR vs. $1.4 million for AnR) and the lower cost of the construction and maintenance of sludge treatment were compensated for in AnR in its extra expenses on the membrane, compared with AeR.

Smith et al. (2014) emphasized the impact of different sludge treatment approaches on the costs of AeMBR and AnMBR. The cost difference between AeMBR and AnMBR under their setting (<13%) is much smaller than the one in this study. If the end use is not considered and the system boundary is only limited to the effluents from the AeMBR and AnMBR directly, which are exactly the effluents from AeD and AnR in this study, the AnMBR has a slightly higher cost than the AeMBR with a small difference (approximately $0.3 million). However, considering end use (i.e., discharge or reuse), the two MBR systems have a larger difference in life cycle costs (approximately $11 million for AeD vs. AnD, and $41 million for AeR vs. AnR). This implies significant influence of the end use on the life cycle cost of MBR treatment trains.

### 3.2 Strategy of anaerobic membrane bioreactor effluent reuse

#### 3.2.1 Agricultural crop species

The quantity of effluent nutrient used by irrigated crops depends not only on the effluent quality, but also on the landscape, the plants, and the soil condition (Porder, Asner, & Vitousek, 2005). Landscape helps determine the location where the nutrients are enriched, and the other two factors influence the capacity of nutrients recycling or reuse. This study fixed the soil condition (sandy soil with a temperature of 25 °C) and focused primarily on the plant factor to investigate the role of crop species in the strategy of AnMBR effluent reuse. Each crop species has its unique requirement on nutrient fertilization, depending on its own nutrient uptake ability and the environmental conditions (e.g., climate, soil). For the AnR scenario evaluated in the previous section, the nutrients in the effluent were assumed to be completely absorbed by plants. In reality, the amount of the nutrients absorbed depends on the crop species. Table 2 summarized the final water quality after the effluent from the AnMBR was reused for fertigation during that the nutrients were further removed through crop uptake and soil treatment before they reached groundwater. The process of soil treatment for P was not taken into consideration in this study due to the low value of effluent TP and the small difference between the scenarios. Among all the six reuse scenarios, only the AnR_P (peanut) has a TN concentration of 15.2 mg L\(^{-1}\). This is because the symbiotic microbes of peanuts can fix the N from the air and the growth of peanuts does not require additional N. All the other scenarios reduced the effluent concentration to below 0.6 mg NO\(_3\)\(^-\) N L\(^{-1}\), which is acceptable for receiving groundwater (Nolan,
Figure 3 The net present value of aerobic membrane bioreactor (AeMBR) and anaerobic membrane bioreactor (AnMBR) treatment trains for the discharge and reuse scenarios. In scenario names, “Ae” and “An” represent AeMBRs and AnMBRs, and “D” and “R” indicate discharge and reuse scenarios, respectively. O&M, operation and maintenance; BNR, biological nutrient removal; MBR, membrane bioreactor.

Table 2 The final effluent quality for each reuse scenario of the anaerobic membrane bioreactor (AnMBR)

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>Influent</th>
<th>Final effluent</th>
<th>AnR_G</th>
<th>AnR_O</th>
<th>AnR_Sb</th>
<th>AnR_T</th>
<th>AnR_P</th>
<th>AnR_Sc</th>
</tr>
</thead>
<tbody>
<tr>
<td>TN</td>
<td>40</td>
<td>10/0.2</td>
<td>0/0</td>
<td>9.1/0.2</td>
<td>0/0</td>
<td>0/0</td>
<td>40/15.2</td>
<td>13.8/0.6</td>
</tr>
<tr>
<td>NH₄⁺–N</td>
<td>25</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>NO₃⁻–N</td>
<td>0</td>
<td>0.2</td>
<td>0</td>
<td>0.2</td>
<td>0</td>
<td>15.2</td>
<td>0.6</td>
<td></td>
</tr>
<tr>
<td>TP</td>
<td>7</td>
<td>3.8</td>
<td>2.2</td>
<td>1.6</td>
<td>1.6</td>
<td>3.4</td>
<td>3.4</td>
<td></td>
</tr>
</tbody>
</table>

*In scenario names, “An” represents AnMBRs, “R” indicates reuse scenarios, and “G,” “O,” “Sb,” “T,” “P,” and “Sc” represent golf courses, oranges, strawberries, tomatoes, peanuts, and sugarcane, respectively.

Ruddy, Hitt, & Helsel, 1998). The crops of oranges and tomatoes are able to take up all the N from the AnMBR effluent.

Figure 4 shows the impacts of eutrophication, ecotoxicity, global warming, and fossil fuel depletion in the AnMBR reuse scenarios. For the last three impact categories, all six reuse scenarios of the specific crop species achieved similar impacts. This is because the only difference among the reuse scenarios is the nutrient concentration of the effluent water before reaching the groundwater, and the impact categories of ecotoxicity, global warming, and fossil fuel depletion do not depend on that. For eutrophication, peanuts (AnR-P) had much greater impacts (>240%) than the other crop species. The eutrophication impact of AnR-P was also higher than that of AnD, since AnD adopted the A³O process for controlling N to a lower level in the effluent. The factor of N-fixing microbes of peanuts resulted in zero removal of N through crop uptake (the same concentration as the influent, 40 mg L⁻¹, before the soil treatment). As a result, the TN concentration after soil treatment (62% removal) was 15.2 mg L⁻¹, much higher than in the other reuse scenarios. The AnR-G (golf course) and AnR-Sc (sugarcane) scenarios had high impacts on eutrophication, since the nutrients from the AnMBR effluent were more than the nutrients required for these crops, leading to a relatively high TN concentration before soil treatment. The results indicated that matching the effluent quality with the nutrient requirement for fertigation was one of the key factors leading to lower environmental impacts for AnMBR systems.

3.2.2 Spatial implementation

The implementation of the AnMBR for fertigation was investigated considering the crop species at the national level. The largest planting crops were selected for each state in the contiguous United States (USDA, 2019), and their nutrient requirements based on the given soil condition were also collected (UGA AESL, 2018) and are available in Supplemental.
Figure 4. Environmental impacts—(a) eutrophication, (b) ecotoxicity, (c) global warming, and (d) fossil fuel depletion—for the anaerobic membrane bioreactor (AnMBR) reuse scenarios. In scenario names, “An” represents AnMBRs, “D” and “R” indicate discharge and reuse scenarios, and “G,” “O,” “Sb,” “T,” “P,” and “Sc” represent golf courses, oranges, strawberries, tomatoes, peanuts, and sugarcane, respectively. FU, functional unit (0.22 m³ s⁻¹, or 5 million gallons d⁻¹); CTUe, comparative toxic unit equivalent.

Table S4. The nitrate concentration in the natural groundwater is usually <2 mg L⁻¹ (Nolan et al., 1998). According to this criterion, each state was identified with the fitness of crop species to the effluent reuse of AnMBR. The final nitrate concentration of >2 mg L⁻¹ after plant uptake and soil treatment was identified as “no fit,” 0–2 mg L⁻¹ was identified as “fit, few nutrients after soil,” and 0 was identified as “fit, no nutrients after soil.” Figure 5 showed the fitness of AnMBR implementation with the reuse strategy in the contiguous United States. It was found that most of the states are a good fit for using the AnMBR effluent for fertigation.

For the “no fit” category, most of the states (e.g., Missouri, Arkansas, Mississippi) have soybeans [Glycine max (L.) Merr.] as the major crop that does not need additional N fertilizers. The “fit, no nutrients after soil” group contains most of the states, with various climates and soil conditions. The TN removal by soil treatment in this group ranges from 0 to 6.0 mg L⁻¹, compared with 34.0 to 40.0 mg L⁻¹ by plant uptake. It shows that the plant uptake plays a more important role in TN removal by the major crops in these states, such as corn (Zea mays L.), potatoes (Solanum tuberosum L.), and fruit. As a state in the group of “fit, few nutrients after soil,” California, which is under arid climate, uses the reclaimed water as one major source for agricultural irrigation (NWRI, 2012). For nonpotable reuse in California, the traditional filtration-based treatment processes are added after the conventional secondary wastewater treatment—for example, the processes of coagulation, flocculation, sedimentation, cloth media filtration, and ultraviolet disinfection in Watsonville, CA, or the processes of coagulation, flocculation, dual-media filtration, and chlorine disinfection in Orange County, CA (Diaz-Elayed, Rezaei, Guo, Mohebbi, & Zhang, 2019). In addition, most of the cities in California are experiencing urban sprawl. An AnMBR has great potential to be applied in a state like California due to the water demand for agricultural irrigation, as well as the flexible
FIGURE 5  The map of fitness of anaerobic membrane bioreactor (AnMBR) implementation with the effluent reuse strategy in the contiguous United States

implementation scale of MBR systems. Membrane bioreactor systems can be implemented at different scales, from medium to large (Diaz-Elsayed et al., 2019), in both centralized and decentralized settings.

The performance of an MBR, however, could vary depending on several factors, such as specific gas demand, membrane flux, and wastewater strength for an AnMBR (Smith et al., 2014). Consequently, the results of the spatial implementation may change depending on the performance of the AnMBR. These factors will also influence the input for environmental impact and cost analysis; for example, higher specific gas demand and higher flux require more energy contributing to higher environmental impacts and costs. On the other hand, higher wastewater strength could lead to more energy recovery and higher fertilizer value of the AnMBR effluent. Therefore, the relative advantage of AnMBR over AeMBR for a reuse scenario has to be evaluated under the specific operating conditions.

4 | CONCLUSION

A comparative study was conducted to evaluate the environmental impacts and costs of the AeMBR and AnMBR under the discharge and reuse scenarios. It was found that the AnMBR has greater environmental impacts and life cycle costs than the AeMBR in the discharge scenario, but the opposite conclusion was made for the reuse scenario. In the discharge scenario, the incorporation of the BNR system was the main reason for the greater impacts of the AnMBR. In the reuse scenario, the lesser impacts of the AnMBR are attributable to the offset of the nutrients required for crops, and the potential benefits vary depending on the types of crops receiving the reclaimed water. The fertilizer value of the nutrient-rich effluent makes AnMBR economically competitive with the AeMBR, though the AnMBR has higher O&M costs than the AeMBR. The comparison results in this study are specific for MBRs; however, the conclusion could hold true that the favorability of a treatment process is dependent on the intended use of the effluent. In general, reuse-oriented applications will favor anaerobic treatment processes, whereas discharge scenarios will favor aerobic treatment.

Specifically, the AnMBR could be more environmentally and economically sustainable if it is integrated with agricultural fertigation. Considering only the factors of plant uptake and soil treatment, the implementation of AnMBR with crop fertigation resulted in an effluent water nitrate concentration (after crop uptake and soil treatment) of <2 mg L\(^{-1}\) in most US states. This indicated that the strategy of using the AnMBR effluent for fertigation could be a win-win solution to both irrigation water shortage and high environmental impact associated with nutrient removal.

CONFLICT OF INTEREST

The authors declare no conflict of interest.
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REFERENCES


**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of the article.