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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

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Abstract

This study calculated the energy and greenhouse gas life cycle and cost profiles of transitional aerobic membrane bioreactors (AeMBR) and anaerobic membrane bioreactors (AnMBR). Membrane bioreactors (MBR) represent a promising technology for decentralized wastewater treatment and can produce recycled water to displace potable water. Energy recovery is possible with methane generated from AnMBRs. Scenarios for these technologies were investigated for different scale systems serving various population densities under a number of climate conditions with multiple methane recovery options. When incorporating the displacement of drinking water, AeMBRs started to realize net energy benefits at the 1million gallons per day (MGD) scale and mesophilic AnMBRs at the 5MGD scale. For all scales, the psychrophilic AnMBR resulted in net energy benefits. This study provides insights into key performance characteristics needed before an informed decision can be made for a community to transition towards the adoption of MBR technologies.

Introduction

The multi-faceted challenges facing current water treatment systems can no longer be solved by traditional siloed water management approaches. A paradigm shift based on holistic management to maximize the use and recovery of water, energy, nutrients, and materials and to increase financial viability of water systems is needed (Ma et al., 2015). Alternative water

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Contribution

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resource recovery facility (WRRF) concepts such as decentralized water systems, energy recovery, fit-for-purpose and water reuse are being explored (Ma et al., 2015; Xue et al., 2015). Among the resource recovery treatment technologies, membrane bioreactor (MBR) configurations as a decentralized sewer mining system with possible energy recovery and water reuse potential present promising transitional solutions to achieve improved environmental performance. MBRs are classified as aerobic membrane bioreactors (AeMBR) or anaerobic membrane bioreactors (AnMBR). Transitional decentralized systems require use of the existing centralized wastewater treatment (WWT) infrastructure for processes such as solids handling, and sewer mining allows for use of the existing WWT infrastructure. Compared with the traditional aerobic activated sludge process in centralized WWT, AeMBR has great advantages, including a smaller land requirement, lower sludge production, and improved effluent quality, consequently expanding on opportunities for water reuse (Yurtsever et al., 2015). AnMBRs often require reduced energy demand due to the absence of aeration, capability of recovering energy from biogas, and less sludge production than AeMBR treatment (Martin et al., 2011). However, both AeMBRs and AnMBRs are more complex unit processes than activated sludge in the traditional WWT because they generally require high capital cost, high maintenance and operator skills due to scheduled membrane cleaning (Young, 2012). Although the technological, environmental, and economic performance of MBRs are being studied globally, the appropriate decentralized population density land use types and scales for the application of these systems as well as the impacts of climate factors on their performance have not been fully addressed.

The optimal scale of resource recovery-based technologies largely depends on local characteristics such as population density, topography, geographic location, resource recovery options, and existing sewer conditions. A variety of approaches have been used to investigate the optimal decentralization scale of resource recovery based WWT technologies. Economic assessments conducted by Eggimann et al. (2015) suggest that the decentralized technologies appear more attractive when population density is low. Eggimann et al. (2015) developed a two-step techno-economic heuristic modeling approach to determine the optimal degree of centralization in the field of wastewater management and found the benefit of centralization decreases with increasing terrain complexity and dispersion due to increased energy expenditure to transport water over long distance. Guo and Englehardt (2015) estimated the cost of direct potable water reuse systems for various topographies and housing densities through a hierarchical clustering method and suggested that the costs of direct potable water reuse systems in urban/suburban areas from centralized wastewater treatment plants (WWTP) are competitive with current water/wastewater service at scales of 10,000 residences.

In contrast, a number of life cycle assessment (LCA) studies investigating resource recovery, source separation, and wetland strategies found lower environmental impacts associated with centralization (Hendrickson et al., 2015; Lundin et al., 2000; Shehabi et al., 2012). However, the majority of these studies did not include the energy savings of displaced drinking water supply, and therefore did not capture the broader, systems-based trade-offs. Shehabi et al. (2012) did consider the avoided water demand; however, the water supply compared was from desalinization, not the typical municipal drinking water treatment. A recent LCA study

conducted by Cornejo et al. (2016) investigated the energy, global warming and eutrophication potentials of a septic tank with aerobic treatment and drip irrigation for reuse at the household scale, an advanced reclamation facility at the community scale, and an advanced water reclamation facility at the city scale. Although different treatment technologies were compared, including the benefits of avoided water and fertilizer provisions, Cornejo et al. (2016) suggested that the city scale technology option presented the lowest energy and global warming potentials (GWP) compared to the other scenarios investigated on an equal volume of wastewater treated basis. Based on the life cycle inventory (LCI) of entire water and wastewater systems, Xue et al. (2016) found that urine source recovery and composting toilet systems at household scale, graywater reuse for toilet flushing, and blackwater for energy recovery at community scale outperformed the centralized system at city scale in terms of life cycle energy, global warming and eutrophication potentials. Additionally, in a multi-criteria (cost, renewable energy share and water scarcity) decision analysis of the water and energy recovery options of a centralized system and two hybrid configurations, Lee et al. (2013) found that the centralized resource recovery scheme is the preferred option when the communities are close to the centralized facility, while it is less costly for communities farther from the centralized facility to use satellite resource recovery schemes. Similarly, Kavvada et al. (2016) used a life cycle approach to compare the energy use and greenhouse gas (GHG) emissions associated with centralized and decentralized non potable reuse at a variety of topographies, population densities, and scales in San Francisco in order to develop a planning support framework. The authors found that energy and GHG effects were dominated by distribution system pumping for centralized systems and treatment for decentralized systems but that an optimum design would be dependent upon scale and spatial parameters. In summary, the findings on scales of resource recovery-targeted technologies are limited and not consistent. Despite a few recent LCA studies focused on the life cycle environmental and cost impacts of MBRs under various configurations and operation conditions, the optimal decentralization scale at which to implement MBR systems and how climate factors affect that choice from energy, GHG and cost perspectives remains unknown (Pretel et al., 2016; Smith et al., 2014).

The main goal of this study is to determine the influence of population density, scale and climate factors on the environmental and cost performance of AnMBR and AeMBR systems as a sewer mining option to recover energy (in the case of AnMBR) and reuse water for non-potable purposes (for both AnMBR and AeMBR) by performing an LCA and life cycle cost (LCC) analysis. This study calculated the cumulative energy demand (CED), GWP and cost of MBRs for a variety of population density and land use type scenarios, and assessed the key contributing stages and parameters to develop insights on the most sustainable scale, density, and MBR operation scenarios for decentralized transitional water services. The key system performance characteristics quantified by this study will assist communities in making informed decisions as they consider transition towards adoption of MBR technologies.

Methods

This study calculated the LCA and cost profiles of both AeMBRs and AnMBRs providing decentralized WWT services. LCA models for AeMBRs, operated at ambient temperature,

as well as mesophilic and psychrophilic AnMBRs, operated at 35°C and 20°C respectively, were constructed based on GE ZeeWeed® 500D hollow fiber membranes with LEAPmbr Aeration Technology. The ZeeWeed hollow fiber membrane was chosen as it is the leading MBR system installed in North America (Yang et al., 2006; Cote et al., 2012). The influent wastewater modeled is representative of medium strength municipal wastewater from individual residences (Metcalf and Eddy Inc. et al., 2014). While AeMBRs and AnMBRs were compared on a basis of the wastewater flow treated annually. Results were then standardized to an equivalent functional unit basis of one cubic meter of wastewater treated to assess the relative technology performance across scales. For all systems, recycled water was modeled as treated to a similar standard following the guidelines developed by the U.S. Environmental Protection Agency (EPA) for water reuse (CDM Smith Inc, 2012).

2.1. Membrane bioreactor population density and scale scenarios

A number of MBR scenarios varying both the population density (i.e., land use type), from semi rural single family to high density urban, and also the scale, from 0.05 million gallons per day (MGD) to 10 MGD (i.e., 189m³–37,854m³ per day), were investigated assuming 100 gallons (0.38m³) of wastewater generated per person per day. While this per capita rate of wastewater generation is at the high end of the range for the U.S. (DeOreo et al., 2016; Metcalf and Eddy Inc. et al., 2014), it is consistent with the per capita flow for medium strength wastewater (Metcalf and Eddy Inc. et al., 2014). These scenarios, with the associated land area requirements ranging from 0.013 square kilometers to 25.9 square kilometers, are provided in Table 1. The scenario names throughout this study refer to the scale of the system, not the basis of the results. The majority of results are presented on the functional unit basis of a cubic meter of wastewater treated.

While AeMBRs are largely commercialized at the scales investigated, the data behind the AnMBR model is based on bench-scale and pilot scale systems (Feickert et al., 2012; Martinez-Sosa et al., 2011; Smith et al., 2012). Although the AnMBR data is from bench-scale and pilot scale systems, data for industrial scale membranes were utilized (Giménez et al., 2011). While AeMBR is currently in use at commercial scales, a limited quantity of published literature on AeMBR is based on commercial scale installations (Yang et al., 2006), with most literature reporting results from pilot-scale experiments or models of commercial scale AeMBR (Martin et al., 2011; Pretel et al., 2016; Smith et al., 2014). Therefore, some model input parameters were based on communication with and literature from the manufacturer to better reflect AeMBR performance at the commercial scale. The purpose of investigating larger scale AeMBR and AnMBR systems in this work is to understand the theoretical CED and GWP impacts as these technologies are implemented at commercial scales.

2.2. System boundaries

All scenarios examined were based on theoretical U.S. decentralized WWT systems. For MBR technologies, the system boundary starts at collection of wastewater and ends at downstream recycled water use (Fig. 1). The MBRs were modeled as transitional systems, which replace conventional activated sludge system in centralized WWTP and use existing wastewater collection infrastructure and sludge handling. Prior to treatment via MBR, the

wastewater goes through screening and grit removal (preliminary treatment) followed by fine screening, which is important to prevent membrane fouling (U.S. EPA, 2007). The wastewater then undergoes treatment via MBR, which combines biological treatment with solids removal via filtration. MBR treatment is focused on organic matter removal only, and does not include advanced nutrient removal. Electricity generated using biogas produced in the AnMBR reduces the need to purchase electricity. A sensitivity analysis was conducted to determine the effect of using biogas for combined heat and power (CHP). CHP generates electricity and also utilizes the waste heat from combusting biogas to displace natural gas inputs.

The membranes in both AeMBRs and AnMBRs must be regularly cleaned to dislodge solids from the membrane pores and allow permeate to flow through. Fouling in AeMBR and AnMBRs is controlled via air and biogas scouring respectively and by membrane relaxation and a periodic backflush of sodium hypochlorite. For all scenarios, wasted mixed liquor suspended solids (MLSS) from MBR treatment is returned to the sewer for solids handling at the municipal WWTP. This study made the simplifying assumptions that the decentralized AeMBR and AnMBR WWT facilities are sited to ensure sufficient wastewater flow within the sewer to prevent clogging due to wasted MLSS and that the municipal WWTP is large enough that the operation of the transitional treatment systems and any variations in waste returned to the sewer would have a negligible effect on the operations of the municipal WWTP. Therefore, treatment of sludge from the MBRs is modeled as outside the system boundaries. While the AeMBR and AnMBR treatment achieve different sludge stabilization, this study modeled varying sludge characteristics as having a negligible impact on overall solids handling at the large scale WWTP. It was assumed a similar scale centralized WWTP for the MBR solids handling serves all modeled land use types, which may likely be the case in a greater urban area where a WWTP serves both the metropolitan area as well as outlying suburbs and towns.

The MBR effluent is treated with sodium hypochlorite to disinfect the recycled water to a condition that is suitable for use for a variety of purposes. The delivery of the recycled water to the use point includes infrastructure and energy requirements for a pressurized distribution system. Recycled water can be used for non-potable purposes such as toilet flushing, landscape irrigation, cooling towers, car washing, and other uses depending on the effluent quality and quantity. In this study, recycled water quality is not expected to comply with potable water quality standards. This study assumed all recycled water returns to the same user who produced the wastewater for toilet flushing. This is a conservative estimate as there may be mixed end users of the recycled water, including use for irrigation purposes. If the recycled water were used for irrigation, a larger volume of it may be transported shorter distances for centralized irrigation. Because the recycled water is used for toilet flushing purposes, variation in the AeMBR and AnMBR effluent nutrient content is not expected to impact results. However, potentially higher levels of nitrogen and chemical oxygen demand (COD) in the effluent for the AnMBR compared to the AeMBR (Ozgun et al., 2013) may have implications for the fertilization potential if the recycled water were to be used for irrigation purposes. The recycled water was modeled as displacing potable (i.e., drinking) water.

Displaced potable water was modeled based on drinking water treatment and delivery in the Cincinnati, Ohio region (Cashman et al., 2014a). Cashman et al. (2016) also conducted a sensitivity analysis investigating the relative change in impact assessment results if modeling the drinking water treatment and delivery energy intensity based on other values reported in literature. The drinking water treatment system is representative of a large-scale centralized plant that uses gaseous chlorine as the primary disinfectant, and employs a granular activated carbon system for removal of organics prior to chlorine addition. Utilization of this study LCI over the other energy values reported in literature was preferred, as the Cincinnati, Ohio study included, in addition to electricity requirements, impacts from the full LCI of drinking water treatment such as chemical usage, infrastructure demands, transportation requirements, and other fuels used for heat and equipment purposes.

2.3. Impact assessment scope

The life cycle impact assessment (LCIA) method Tool for the Reduction and Assessment of Chemical and environmental Impacts (TRACI), version 2.1, developed by the U.S. EPA specifically to model environmental and human health impacts in the U.S. was applied in this work (Bare, 2002). Energy was tracked based on point of extraction and higher heating value using the CED method developed by ecoinvent (Ecoinvent Centre – Swiss Centre for Life Cycle Inventories, 2010). CED includes all forms of nonrenewable resources (e.g., coal, crude oil, and natural gas and uranium) and renewable resources (e.g., biomass, water, sunlight, and wind). CED accounts for energy demanded in process operation as well as energy embedded in any materials utilized. GHGs were characterized within TRACI using the Intergovernmental Panel on Climate Change (IPCC) 2007 Fourth Assessment Report (AR4) GWPs (Solomon et al., 2007). GHGs in this analysis cover emissions associated with fuel combustion, electricity generation, chemical and material production as well as fugitive process GHGs from WWT processes. Carbon dioxide, methane, and nitrous oxide GHGs drive GWP for processes associated with fuel combustion and material production, while methane process emissions occur in the anaerobic WWT processes. Although the full suite of TRACI impact assessment results was generated for the MBR systems, the focus of this analysis is GWP and CED as many of the variable parameters in the LCI model most influence these categories. Potential potable water savings is also discussed in the baseline results. Developed LCI were characterized to the assessed LCIA categories in the open-source LCA software openLCA version 1.4.2 (GreenDelta, 2015).

A companion LCC was conducted and calculated the cost of greenfield installations of the MBR systems assuming the collection system is already in place, but the recycled water delivery system needs to be installed. Total costs were amortized over a period of 40years, assuming financing is procured via a long-term municipal bond. Replacement costs were included if the equipment was expected to last less than 40years. Capital equipment and infrastructure with expected lifetimes shorter or longer than 40years are documented in Cashman et al. (2016).

In general, numeric impact assessment results were standardized to three significant figures, which may exceed the number of significant figures presented in the underlying data. LCIA results are a composite of hundreds or thousands of underlying values throughout the life

cycle of the WWT systems and individual tracking of significant figures is often not feasible. Readers should take care when drawing comparisons based on small differences in results. While a detailed uncertainty analysis was not possible due to the overall lack of data points available needed to align the MBR LCI models, notable differences in comparative results can be used for directional guidance. Although there is also uncertainty inherent LCIA methods, the same LCIA methods were applied to all investigated MBR systems, therefore, trends in comparative results can be determined with greater confidence than absolute results.

2.4. Scope of climate and methane recovery scenario analysis

For the psychrophilic AnMBR (operated at 20°C) two specific climate scenarios were run under multiple conditions. Operation was considered in the winter in a cold climate Cincinnati, Ohio (ambient temperature=6°C and influent temperature=17.9°C) and for the annual average in a warm climate Miami, Florida (ambient temperature=26.4°C and influent temperature=26.4°C) (Metcalf and Eddy Inc. et al., 2014). Key differentiating parameters in this scenario analysis include the ambient temperature, influent wastewater temperature, reactor temperature, whether reactor surface insulation is included, whether methane is recovered from the permeate, and whether methane recovered from permeate/headspace is flared, converted to electricity only, or converted to both electricity and heat via CHP. Table 2 displays all scenarios investigated, along with the associated scenario letter that results were calculated for. For this scenario analysis, results were limited to the 1MGD scale.

2.5. Life cycle inventory model

The primary LCI developed under this study is documented in detail in (Cashman et al., 2016). The following subsections describe methods used to develop specific LCI components. All LCI unit processes were built in the LCA software openLCA version 1.4.2 (GreenDelta, 2015). Much of the LCI and cost data were developed using the CAPDETWorks™ (version 3.0) modeling software (Hydromantis, 2014). CAPDETWorks is based on algorithms developed by the U.S. EPA and U.S. Army Corps of Engineers in 1979 and has since been updated by Hydromantis using design standards from Metcalf and Eddy and cost indices from Marshall and Swift, Engineering News Record, and Chemical Engineering Pipe Valves and Fitting Index (Hydromantis, 2014; Metcalf and Eddy Inc. et al., 2014).

2.5.1. Pre and post treatment models—All pre and post treatment steps were modeled as equivalent for the AeMBR and AnMBR LCI. For preliminary treatment, CAPDETWorks default settings were used for input parameters to calculate preliminary treatment cost, infrastructure requirements, energy demand, and quantity of grit landfilled.

To reuse the treated water for non-potable purpose, disinfection with sodium hypochlorite is necessary to meet water quality standards. Although regulations for water reuse are set by each state, this study used a set of guidelines developed by the U.S. EPA (CDM Smith Inc, 2012). Because CAPDETWorks only models chlorination with chlorine gas, a stoichiometric conversion was made to calculate sodium hypochlorite consumption required based on the CAPDETWorks output of average chlorine required per day. CAPDETWorks assumes the

same annual electricity demand for chlorination units in all systems 5MGD or less, which seemed an overestimation for scales as small as 0.05MGD. Therefore, the equation provided in the CAPDETWorks technical manual to calculate annual electrical demand for chlorination for systems larger than 5MGD was used to manually calculate electricity demand for the 0.05, 0.1, 1, and 5MGD systems (Hydromantis, 2014). Water output volume from chlorination was modeled the same as input volume.

2.5.2. Aerobic membrane bioreactor model—AeMBR LCI were derived largely from CAPDETWorks modeling simulations of the combined plug flow activated sludge (PFAS) and MBR processes (Hydromantis, 2014). PFAS, unlike complete mix, can be modeled in conjunction with an MBR in CAPDETWorks. Comparisons between complete mix, which may also be used for small scale systems, and PFAS were conducted, and differences seen in electricity use and air flow were small. PFAS was modeled using default CAPDETWorks user input parameters with a 2.3h hydraulic retention time (HRT) and a 10day solids retention time (SRT) and assuming a process design choice of carbon removal only (i.e., no nutrient removal). Several CAPDETWorks user inputs for the MBR process were adjusted to model the GE ZeeWeed 500D MBR with LEAPmbr aeration. GE documentation indicated a switch from 10/10 sequential aeration (as modeled in CAPDETWorks) to LEAPmbr aeration can reduce specific scour air demand and associated energy use 30–70% by utilizing larger bubble scouring at shorter aeration intervals (Cote et al., 2012; GE Power and Water, 2014; Kicsi, 2014). The midpoint in the range of expected reductions, 50%, was used to calculate the input of $0.15\text{Nm}^3/(\text{m}^2\text{hr})$ for the specific scour air demand parameter. Based on personal communication with a GE Power and Water representative, the physical cleaning interval, physical cleaning duration, chemical cleaning interval, and backflush flow factor inputs were changed to 10min, 45s, 84h, and 0, respectively. The quantity of sodium hypochlorite needed for cleaning the membranes was calculated separately from the CAPDETWorks model using a factor of 900L of a 12.5% by weight solution per year per 370 square feet of membrane surface area (GE Power and Water, 2014).

Infrastructure data for PFAS and MBR processes included earthworks and concrete consumption for construction of tanks as well as the membrane material. The hollow fiber membrane is made of polyvinylidene fluoride (PVDF) with an estimated lifetime of 10years (Cote et al., 2012). The quantity of PVDF used in the membrane was calculated based on CAPDETWorks results for the total surface area of membrane required for each size system and manufacturer specifications for the inner and outer diameter of a hollow fiber (GE Power and Water, 2013).

2.5.3. Anaerobic membrane bioreactor model—While AeMBRs are largely commercialized at the scales investigated, AnMBRs are mostly in the bench or pilot scales and data from such was used here. The energy balance calculations for the AnMBR, including assumptions regarding the biogas recirculation pump operational parameters, were based on work by Feickert et al. (2012). While the Feickert et al. (2012) study modeled a system treating 130m^3 of wastewater per day (34,342gallon/day), the model here was parametrized to theoretically increase the scale to 10MGD. AnMBR energy balance scaling was based on thermodynamic first principles verified through calculations. The overall

AnMBR process flow and operational parameters were found to be consistent with what is reported in literature (Giménez et al., 2011; Metcalf and Eddy Inc. et al., 2014).

This study investigated the operation of AnMBR at 35°C, representative of a mesophilic AnMBR system, and at 20°C, representative of a psychrophilic AnMBR system (Metcalf and Eddy Inc. et al., 2014). While historically biological treatment at low temperatures can suffer from instability in reactor operation, recent literature indicates low temperature tolerance of the psychrophilic AnMBR system, with four pilot scale AnMBRs achieving at least 90% COD removal when operated at temperatures between 18 and 23°C (Shin and Bae, 2017). For both the mesophilic and psychrophilic AnMBR, the influent wastewater to the reactor was modeled as 20°C in the baseline and varied in the climate sensitivity analysis.

Typical biogas from AnMBRs has a methane content of 55% to 70% (this study uses 65%) and most of the remaining content is carbon dioxide (Metcalf and Eddy Inc. et al., 2014; Smith et al., 2012). Methane produced from wastewater streams with a relatively low influent sulfate concentration, as is the case with medium strength municipal wastewater, is largely a function of influent COD strength, and commonly expressed as volume of methane per mass unit of COD (Metcalf and Eddy Inc. et al., 2014; Shin and Bae, 2017). Methane production increases with an increase in the reactor temperature. This study modeled an overall methane production rate of 0.24m³CH₄/kgCOD at 20°C and 0.27m³CH₄/kgCOD at 35°C (Martinez-Sosa et al., 2011). The recovered methane from the headspace is converted to electricity for operating the pumps in the treatment system in the baseline. Negative LCI net electricity values for the AnMBR indicate the system produces more electricity than it consumes, resulting in a net electricity credit.

A portion of methane produced remains dissolved and leaves the system in the permeate (Smith et al., 2012). While supersaturation of dissolved methane occurs in some types of anaerobic reactors, this has not been found in AnMBR systems with biogas sparging and regular membrane cleaning (Crone et al., 2016; Giménez et al., 2011). Thus, the amount of methane per liter of permeate was calculated from Henry's Law (Metcalf and Eddy Inc. et al., 2014; Smith et al., 2014). Based on previous literature, the baseline assumption for this study is 0% recovery of dissolved methane (Smith et al., 2014). Therefore, the amount of methane emitted per liter of permeate is the difference between the total amount of dissolved methane and the dissolved methane that remains in solution when exposed to the atmosphere, assuming normal temperature (20°C) and pressure (1atm.) in all scenarios and 1823ppb methane in the atmosphere (Dlugokencky, 2015).

Several of the sensitivity analysis scenarios include recovery of dissolved methane for conversion to electricity and heat. Membrane separation was selected as the dissolved methane recovery method based on its simplicity of operation and maintenance, no hazardous chemical use, and its beneficial economic performance in low gas flow situations (Makaruk et al., 2010). An 80% recovery of methane dissolved in the permeate was modeled (Makaruk et al., 2010). Equations calculating energy use for adiabatic compression have been found to provide a reasonable estimate for energy use for polytropic compression involved in membrane separation and were used to calculate electricity demand (Perry and

Green, 1997). The capital cost of membrane separation technology was determined from the power requirement for a screw-type compressor in 1985 and adjusted for current cost using the Chemical Engineering Plant Cost Index (CEPCI, 2015; Walas, 1990).

This study used a HRT of 8h, a SRT of 60days, and a MLSS concentration of 12g/L, which were the midpoints of ranges provided by an AnMBR process engineer at GE Power and Water (Fonseca, 2015). The amount of sludge returned to the main sewer system to be treated downstream at the regional WWTP was derived from Metcalf and Eddy Inc. et al. (2014) equation for calculating SRT for aerobic MBR systems and adapted for use with the AnMBR systems.

AnMBR membrane surface area was determined based on an average net flux of 7.5L/m²/hour (LMH) reported in a literature review on AnMBR systems by Chang (2014) and confirmed through personal communication with GE Power and Water (Fonseca, 2015). Shin and Bae (2017) list the range of fluxes for 11 pilot scale reactors reviewed as 4.1–17 LMH with a standard deviation of 3.5 (Shin and Bae, 2017). Under this reported range, the flux assumed for this study is slightly conservative. Calculations and assumptions used to determine tank sizes, train redundancy, and membrane surface area were similar to those used in CAPDETWorks for the AeMBR systems. Flux was modeled the same for both 20°C and 35°C AnMBR operating temperatures, so the potential increase in flux due to decreased viscosity of permeate at the higher temperature was not taken into account. Although this could decrease membrane surface area required by up to 25%, the effect of increased temperature on flux is difficult to predict because mixed liquor viscosity, biological activity, oxygen transfer efficiency, and microbial metabolism are also factors affecting flux (Yoon, 2016). Since results presented in Section 3 show the CED and GWP impacts of AnMBR operation at 35°C and infrastructure are around one tenth of those for heating the influent and total AnMBR operation and infrastructure costs influenced by flux are only 4% to 12% of system costs, the potential increase in flux is not likely to have a large impact on overall life cycle environment or cost impacts.

Costs for the AnMBR system were mainly derived from the CAPDETWorks models for an anaerobic reactor and MBR with the tank dimensions listed in Cashman et al. (2016) replacing default user inputs.

2.5.4. Wastewater collection and recycled water delivery model—The length of sewer pipe calculated per land use type scenario is provided in Cashman et al. (2016) and increases with a decrease in population density and increase in reactor scale. High density urban scenarios require 0.21–42.7km of sewer pipe for 0.05MGD–10MGD scales; whereas, single family scenarios require 1.67–335km of sewer pipe for 0.05MGD–10MGD scales. The modeled pipe length for the recycled water delivery system is similar to that of the collection system, as this study modeled all recycled water as delivered back to the original user for toilet flushing. Additional piping is required to construct the recycled water distribution system. The pipe material, diameters, and lifetimes modeled for both the collection system and recycled water delivery system are provided in Cashman et al. (2016).

The collection system was modeled as a gravity collection system (Cashman et al., 2014b). For distribution of the recycled water from the decentralized MBR WWT facility back to the end user, the pumping energy required for water delivery is affected by changes in elevation throughout the pipe network as well as head losses due to friction inside the pipe. Changes in elevation throughout the distribution network can vary greatly depending on the geographic location being modeled and were not included. Therefore, the recycled water delivery results presented here are most applicable for flat topographies. Frictional head losses are influenced by the inside diameter of the pipe, the flow rate through the pipe, and the smoothness of the interior pipe wall. Smoothness coefficients and the pumping energy calculations to overcome frictional head losses per unit of pipe length are summarized in Cashman et al. (2016).

The use of recycled water was modeled to replace the equivalent quantity of drinking water produced and delivered to the consumer. The drinking water production and delivery model in the baseline results used LCI data previously developed by the U.S. EPA (Cashman et al., 2014a). The recycled water delivery applies the same overall water loss rate of 19% as used in the drinking water delivery LCI model (Cashman et al., 2014a). This is a conservative estimate, as it is possible the newer pipe infrastructure would have less leaks compared to the older installed water distribution network.

Results and Discussion

3.1. Baseline results with population density scenarios

The CED and GWP results for the baseline AeMBR, mesophilic AnMBR, and psychrophilic AnMBR under all investigated population density and scale scenarios are illustrated in Fig. 2. Fig. 3 displays similar results for the multi-family land use type only, but includes results by life cycle stage and results on an annual basis. CED and GWP impacts decreased as the population density increased and the scale of the system increased. As shown in Fig. 2a, under all scale and population density scenarios, the psychrophilic AnMBR resulted in net CED benefits; whereas, net CED benefits were only realized for the AeMBR and for the mesophilic AnMBR at scales above 1MGD and at scales above 5MGD respectively. Similar results are seen for GWP in Fig. 2b, but under no scenario does the mesophilic AnMBR result in net GWP benefits. Net GWP benefits for the psychrophilic AnMBR and for the AeMBR begin at 0.1MGD and 1MGD respectively. While both CED and GWP decreased as the population density increased, the population density is not the main driver of the impacts. For instance, the 1MGD AeMBR CED impacts decreased 14% when shifting from semi-rural single family land use to high density urban. The main reason for this decrease is a reduction in pumping energy to deliver the recycled water to the user (i.e., less friction loss with recycled water traveling less distance in pipe) and a lower infrastructure production and installation for both the collection system and recycled water delivery pipe networks. As evident in Fig. 3, these contributors do not account for the majority of CED and GWP impacts across the life cycle of the MBRs.

Heating of the influent with natural gas was the largest CED and GWP contributor for the mesophilic AnMBR. Operating the reactor at ambient temperature rather than at a higher temperature of 35°C resulted in significant CED and GWP savings, indicating both climate

and reactor operating temperature are key parameters in assessing the AnMBR life cycle GWP and CED impacts. Electricity requirements for aeration dominated the operational GWP and CED burdens of the AeMBR. Chlorination was also a notable contributor to overall CED and GWP impacts, but had similar impacts for all MBR systems investigated. Infrastructure impacts, when standardized by the useful life of the equipment, played a negligible role in life cycle CED and GWP. For example, infrastructure, including collection system and recycled water delivery piping, contributed to 0.2%–1.6% of the overall CED for the AeMBR. This observation that infrastructure has insignificant global warming potential impacts in comparison with operation and maintenance lies at the lower end of the previously reported range of 1%–30% (Morera et al., 2017; Ortiz et al., 2007). Although this study incorporated detailed inventory of construction materials, equipment and civil works, the energy-intensive operational processes including heating of influent, electricity consumption, and chlorination, rather than infrastructure processes, dominated the MBR systems' GWP.

There were multiple potential CED and GWP benefits observed when treating wastewater with decentralized MBR systems and using the MBR system effluent as recycled water to displace potable water. The largest CED and GWP benefit for all MBR systems was from recycled water displacement of potable water. Based on a literature review of studies reporting energy consumption for drinking water treatment and distribution, it was determined that the energy consumption for the displaced drinking water considered in these baseline results is relatively high. A sensitivity analysis applying different ranges of energy consumption for the displaced drinking water treatment is conducted in Cashman et al. (2016), showing that overall study trends were not sensitive to drinking water CED. Therefore, fit-for-purpose water reuse potentially provides the most benefits in decentralization planning and contributes the most towards system efficiency. For the AnMBR, the energy savings resulting from recovery of headspace methane resulted in notable CED and GWP benefits; however, these savings were overshadowed by the requirements for heating of the influent for the mesophilic AnMBR.

While this study focused on CED and GWP impacts of the decentralized MBR systems, there was also a potential significant water savings from using recycled wastewater. This study found that use of recycled water from decentralized MBRs in the baseline scenarios avoided a net 0.94–0.96 cubic meters of fresh water per cubic meter across the life cycle of wastewater treated by MBR. Water savings results are not shown by life cycle stage or by scale scenario since the recycled water displacement of potable water dwarfs the operational water impacts of MBRs. Water consumption during operation includes water usage in upstream electricity and chemical production. Displaced potable water covers water that would be delivered to the consumer along with drinking water loss during distribution. In this analysis, the net water avoided was never greater than the volume of displaced water given upstream MBR water requirements for material, chemical and energy generation. However, it is theoretically possible that the volume of displaced potable water could exceed the volume of demanded potable water with sufficient surplus electricity produced, which would avoid the water demand for electricity generation. The amount of surplus electricity produced can be increased through optimizing climate conditions, reactor temperature, and methane recovery options, as discussed in Section 3.2.

AeMBR and AnMBR LCC results are shown for the multi family land use type per cubic meter of WWT in Fig. 4. Costs for each cost component on a functional unit basis for all land use types are provided in Cashman et al. (2016). The LCC analysis includes infrastructure, WWTP operation, recycled water delivery, and avoided drinking water. Costs decreased significantly as scale increased, with large reductions in cost seen starting at the 1MGD scale. AnMBR costs are notably higher than AeMBR costs. The main driver for this is the increase in operation and maintenance (O&M) labor needed to operate the anaerobic reactor and, to a lesser extent, anaerobic reactor capital costs. Across all scenarios, O&M and capital costs combined were responsible for roughly 90% or more of the overall cost of the system. The share of expenses due to O&M decreased with increasing scale, while the share for capital costs increased. Scenarios with lower density service areas were costlier than scenarios of the same scale with higher density service areas due to increased infrastructure to deliver recycled water over greater distances. Overall, LCC results were much more sensitive to variations in scale than variations in population density.

3.2. Climate and methane recovery scenarios

Fig. 5 illustrates annual CED and GWP for various climate and methane recovery scenarios of the 1MGD psychrophilic AnMBR. Net CED benefits were realized for all scenarios except Cincinnati, Ohio winter scenarios (ambient temperature=6°C) with only biogas flare and no reactor surface insulation. The largest CED benefits were from displaced potable water and recovery of methane in headspace. Less key CED benefits were for inclusion of reactor surface insulation and recovery of methane from permeate. The greatest energy impact was from heating of influent under Cincinnati winter conditions. The optimal scenario investigated from an energy perspective was the psychrophilic AnMBR operated in warm climate (e.g., Miami, FL, annual average ambient temperature=26.4°C) with methane recovery via CHP and both headspace and dissolved methane recovery. GWP results were similar to those seen for CED with the exception of permeate methane emissions. Notable GHG savings were realized through capture of methane in the MBR permeate. Cost results were not sensitive to regional considerations, as costs were driven by capital, material and labor requirements rather than operational energy requirements.

3.3. Summary of findings

This study found that for all scales investigated, the psychrophilic AnMBR reactor operated at 20°C resulted in the most net energy benefits compared to the other investigated systems. The AnMBR GWP and CED was found to be highly sensitive to reactor temperature and climate. This study supports findings that AnMBRs operated at lower reactor temperatures in warmer climate represent a promising technology for decreasing the environmental impacts of WWT systems. Potential CED and GWP benefits of the psychrophilic AnMBR increased when operating in warm climates and when the dissolved methane in the permeate was recovered. When examining the CED results standardized to a cubic meter of water treated, all scenario CED impacts decreased as the scale increased in the same scenario categories. While the AnMBR operating at ambient temperature resulted in notable energy and GHG benefits, the AnMBR costs remained higher than the AeMBR under all scenarios. The study also found that all impacts decreased comparatively as the population density increased; however, population density was not a main driving parameter in determining the

environmental impacts such as GWP and CED. In other words, the difference among the land use type was not significant compared to other factors such as the heating of the influent or the climate. Cost, however, was less sensitive to regional considerations, and was most strongly influenced by the scale of the reactor, with economies of scales realized for the larger MBR scales. Cost was also less influenced by operational energy requirements, and more dependent on capital, material and labor requirements.

Conclusions

When selecting or managing a water treatment approach, it is crucial to perform a comprehensive systems analysis to understand environmental and cost trade-offs of different options. In a system level evaluation of the transition from centralized to decentralized treatment systems producing recycled water, the environmental impacts and costs did not always result in net benefits when considering a variety of scales, technologies, population densities, and climates. The importance of system level analysis was illustrated through indirect benefits provided by displaced drinking water compared to direct impacts of the technology such as operational costs or benefits (e.g., energy recovered from wastewater).

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Figure 1.
Study system boundaries. Main sub processes of AnMBR and AeMBR operation are highlighted.

Figure 2. AeMBR and AnMBR population density scenario analysis of (a) CED per year, and (b) GWP per year. Results calculated for a mesophilic AnMBR operated at 35 °C and for a psychrophilic AnMBR operated at 20 °C.

Figure 3.

AeMBR and AnMBR multi-family land use comparison by life cycle stage of (a) CED per year, (b) CED per cubic meter of WWT, (c) GWP per year, and (d) GWP per cubic meter of WWT. Results calculated for a mesophilic AnMBR operated at 35 °C and for a psychrophilic AnMBR operated at 20 °C.

Figure 4.
Comparative MBR system cost for multi-family land use scenario on basis of costs per cubic meter of wastewater treated.

Fig. 5. 1MGD psychrophilic AnMBR climate and methane recovery scenario analyses for multi family land use type. Results shown for a) GWP per functional unit and b) CED per functional unit. GWP scenario results are shown in descending order. Parameter combinations are defined by the scenario letter and defined in Table 2.

Table 1

Scale and population density land area scenarios for AeMBR and AnMBR systems.

Population density and scale land area scenarios						
Population density	Land use type	500ppl served	1000ppl served	10,000ppl served	50,000ppl served	100,000ppl served
		0.05MGD (189m³/d)	0.1MGD (379m³/d)	1MGD (3785m³/d)	5MGD (18,927m³/d)	10MGD (37,854m³/d)
100,000 #ppl/sqm	High density urban	0.005sqm (0.013sqkm)	0.01sqm (0.026sqkm)	0.1sqm (0.26sqkm)	0.5sqm (1.29sqkm)	1sqm (2.59sqkm)
50,000 #ppl/sqm	Multi family	0.01sqm (0.026sqkm)	0.02sqm (0.052sqkm)	0.2sqm (0.52 sqkm)	1sqm (2.59sqkm)	2sqm (5.18sqkm)
10,000 #ppl/sqm	Single family	0.05sqm (0.13sqkm)	0.1sqm (0.26sqkm)	1sqm (2.59sqkm)	5sqm (12.9sqkm)	10sqm (25.9sqkm)
2000 #ppl/sqm	Semi-rural single family	0.25sqm (0.65sqkm)	0.5sqm (1.29sqkm)	5sqm (12.9sqkm)	N/A	N/A

Notes: d=day; MGD=million gallons per day; ppl=people; sqkm=square kilometer; sqm=square mile.

Table 2

Climate and methane recovery scenarios for 1 MGD the psychrophilic AnMBR.

Scenario	Ambient temperature (°C)	Influent temperature (°C)	Reactor temperature (°C)	Reactor insulation	Methane recovery option		Methane permeate recovery
					Flared	Converted to electricity only	CHP
A	6.0	17.9	20.0				
B	6.0	17.9	20.0				
C	6.0	17.9	20.0				
D	6.0	17.9	20.0				
E	6.0	17.9	20.0				
F	6.0	17.9	20.0				
G	6.0	17.9	20.0				
H	6.0	17.9	20.0				
I	6.0	17.9	20.0				
J	6.0	17.9	20.0				
K	26.4	26.4	26.4				
L	6.0	17.9	20.0				
M	6.0	17.9	20.0				
N	26.4	26.4	26.4				
O	26.4	26.4	26.4				
P	26.4	26.4	26.4				
Q	26.4	26.4	26.4				
R	26.4	26.4	26.4				

Note: CHP – combined heat and power