Flexible hybrid membrane treatment systems for tailored nutrient management: A new paradigm in urban wastewater treatment

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The integration of onsite, decentralized, and satellite wastewater treatment systems into existing urban water infrastructure is an attractive option for recovering water and nutrients locally for multi-purpose reuse. To facilitate wastewater treatment and reuse, tailored to local needs, a hybrid membrane treatment process is proposed that couples sequencing batch reactors with a membrane bioreactor (SBR-MBR). In this study, we explored the flexibility and robustness of this hybrid membrane system at a demonstration-scale under real-world conditions by tightly managing and controlling operation conditions to produce effluent of different qualities for multipurpose reuse. Results suggest that an SBR-MBR treatment configuration is flexible, robust and resilient to changing operating conditions. The hybrid system was capable of producing different effluent qualities within 1 week of changing operating conditions, with no adverse effects on membrane performance. This work reinforces the need for a new paradigm of water reclamation and reuse and introduces a new treatment concept facilitating tailored nutrient management for a sustainable urban water infrastructure.

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1. Introduction

Aging wastewater infrastructure, freshwater scarcity, population growth and urbanization, as well as climate change are drivers to advance the science and technology of recovering the resources present in domestic wastewater, a predominantly untapped resource harboring energy, nutrients, and fresh water [1,19,27]. As a dominant driver for water reuse, water scarcity is becoming increasingly prevalent on a global scale. Furthermore, more than half the Earth's human population lives in urban areas, creating increasing pressure and depletion of local water supplies [21]. In addition to water scarcity and urban population growth, decaying urban water infrastructure and inadequate end-of-pipe reuse strategies call into question whether the current paradigm of centralized wastewater treatment is capable of coping with the water supply challenges of the 21st century [10,23]. Thus, water planners and engineers must look beyond traditional methods of water supply (e.g., structural developments and inter-basin water transfers) and adopt an integrated, whole system approach to managing water assets [2]. These assets must include locally available reclaimed water as a strategic supply for balancing urban water use, meeting short-term needs, and improving long-term supply reliability [6].

Recently, onsite, satellite, and decentralized treatment systems (henceforth referred to collectively as distributed) have gained attention for reclaiming used water in the urban environment [10]. Furthermore, advancements in membrane bioreactor technology have made the concept of sewer mining, or scalping, feasible for distributed installations across the urban area where demand for local reuse of water exists [6,8]. Source separation of black-water, urine, and grey-water has also been shown to be reliable and economically viable when implemented for new developments [27,29,7]; however, retrofitting existing infrastructure for source-separated treatment might be economically impractical because the majority of the wastewater infrastructure is already in place. Nonetheless, retrofitting water infrastructure for new treatment approaches ultimately must be evaluated on a case-by-case basis, and planners and local communities must weigh the benefits of infrastructure improvements. In order to prevent freshwater shortages within urban centers, these benefits may include water reclamation and reuse by integrating new treatment strategies into existing water infrastructure.

While the advantages and disadvantages of centralized versus decentralized treatment systems have been discussed elsewhere [18], we target a specific niche of the urban water infrastructure aimed at facilitating distributed reuse within clustered housing developments and apartment complexes. Sequencing batch reactors (SBRs) and membrane bioreactors (MBRs) are uniquely suited for treating wastewater in decentralized settings. Unlike conventional activated sludge processes that employ several dedicated unit processes, SBRs are ideal because their operation is managed through...
time; thus, operation and processes conditions are highly flexible to accommodate load variations, which are typical in decentralized settings [25,26]. MBRs are also ideal for decentralized systems because effluent can only be produced once passed through a membrane barrier. MBRs also reject the majority of bacteria (e.g., when ultrafiltration membranes are employed), can achieve some virus reduction, and are currently accepted as the most appropriate technology where high quality effluent is required for reuse [8,9].

A hybrid SBR-MBR treatment configuration has been described in the literature and evaluated at the laboratory-scale [4,15]. The main advantage of this configuration is the ability to use small exchange ratios (i.e., the ratio between the combined reactor volume and the volume exchanged in each cycle); thereby leading to high nitrogen removal rates [15]. Furthermore, the flexibility of an SBR combined with consistent permeate production via membrane filtration may be an optimal configuration for distributed wastewater treatment within the urban environment.

Other hybrid MBR or SBR concepts have been described in the literature [5,16], both of which reliably achieved complete nutrient removal. However, if optimal recovery of water and dissolved nutrients in domestic wastewater is desired, treatment systems must be designed for flexibility in order to meet effluent qualities that can be tailored for seasonal or site specific uses (i.e., nutrients may be retained in the effluent depending on need) rather than just meeting effluent discharge standards. Thus, the concept of tailored nutrient management can be viewed as an alternative to complete nutrient removal. For example, reclaimed water is a medium to convey valuable dissolved nutrients (i.e., nitrogen and phosphorous) for purposes such as urban landscape irrigation, while also meeting other reclaimed water qualities when irrigation is not desired.

The mechanisms facilitating flexibility of the biological treatment processes are the result of tight regulation by microorganisms at the transcriptional level. For example, the expression of carbon and polyphosphate metabolic pathways by activated sludge microbial communities in an enhanced biological phosphorus removal (EBPR) reactor has been shown to exhibit a dynamic range of expression levels during a normal anaerobic/aerobic cycle in response to external acetate, oxygen and phosphate concentrations [11]. The processes of nitrification and denitrification have also been also been shown to be tightly regulated at the transcriptional level [12,17].

The main objective of this study was to investigate the flexibility and robustness of a hybrid SBR-MBR system treating 27 m³ of domestic wastewater per day (7200 gpd) from a clustered housing development to facilitate tailored nutrient management in an urban setting. Under the experimental design, the performance and feasibility of a distributed hybrid treatment process to properly capture temporal variations in influent wastewater characteristics under real-world conditions were tested. We further tested the treatment process with regard to two disparate treatment strategies by altering the number and duration of discrete aeration periods of the SBR. Each treatment strategy aimed at achieving the same carbon removal efficiencies, while resulting in different total nitrogen and total phosphorus effluent qualities.

2. Experimental protocols

2.1. Site layout and SBR-MBR process description

The SBR-MBR system receives raw sewage from a 250-unit student-apartment complex at the Colorado School of Mines (Golden, Colorado) through a sanitary sewer diversion (Fig. 1a). At the site, influent wastewater flows through a 9.5 m³ (2500 gal) underground holding tank where a submerged grinder pump transfers a 1.14 m³ (300 gal) hourly batch of raw, screened (2 mm) sewage to one of the two SBRs at approximately 113 L/min (30 gpm). Each SBR tank is mixed with a 2.4 horsepower (HP) submersible centrifugal pump with bi-directional jet nozzles for mixing. The SBR-MBR system consists of two parallel-operated SBR tanks that work in series with two parallel-operated membrane tanks. The volume of activated sludge in a single SBR tank ranges between 9.7 and 10.8 m³ (2560-2850 gal) and the volume of a single MBR tank is 7.27 m³ (1921 gal). Total system volume is thus approximately 33.9 m³ (~8950 gal). Each membrane tank contains one submerged PURON® PSH30 hollow fiber module (Koch, Wilmington, Massachusetts). The total surface area of each membrane cassette is 30 m², the nominal pore size is 0.05 μm, and each of the nine membrane bundles in each cassette is individually air-scoured through a diffuser nozzle located at the center of each fiber bundle.

Each batch of wastewater is processed over a 2-hour treatment cycle. During the first hour (React phase), the batch of raw wastewater is isolated in the SBR tank, and during the second hour (React/Filtration phase), mixed liquor suspended solids (MLSS) is exchanged between the SBR and the two MBRs. In this investigation, temporal variations of an anaerobic or anoxic fill phase (Fill), an intermittently aerated phase (React) (total time for Fill plus React phases is 60 min), and a 60 min intermittently aerated discharge phase (React/Filtration) we used. No MLSS is exchanged between an SBR and two MBRs during the combined 60-min Fill and React phases (Fig. 1b). However, during the React/Filtration phase, MLSS is continuously pumped from the SBR to the MBRs at approximately 76 L/min (20 gpm) into each of the two MBRs. Each membrane module produces approximately 12 L/min (3.2 gpm) permeate and excess concentrated MLSS from the membrane tanks is returned by gravity to the SBR from which it was received via a spillway and a control valve (Fig. 1b).

To ensure continuous operation of the system, the SBR-MBR cycles are configured such that while one SBR is in the Fill/React phase, the second SBR is in the React/Filtration phase. Correspondingly, the MBRs continuously receive and return MLSS to the SBR which is in React-Filtration phase. Under this process configuration, settle and decant phases typical of conventional SBR operation are obviated due to the continuous exchange and rapid filtration of MLSS between SBRs and MBRs. In this investigation, the SBRs were operated at approximately 5000 mg/L MLSS and the MBRs at approximately 8000 mg/L MLSS. Waste activated sludge (WAS) was discharged from system via the spillway (Fig. 1b) at the start of each React/Filtration phase, at a flow rate of approximately 41.6 L/min (11 gpm) for 0.8–1.0 min per cycle. Sludge retention time (SRT) was calculated based on total amount of biomass and adjusted weekly using WAS flow duration. During the investigation, the SBR-MBR system was operated at an SRT of 35–40 days.

Based on the batch volume, the SBR-MBR control system calculates the amount of water to be processed by matching the batch volume with either an optimum net flux of 24 L m⁻² h⁻¹ or a peak net flux of 45.5 L m⁻² h⁻¹. Furthermore, permeate backflush is automated and performed every 6 min for 20 s during optimal flux and every 3 min for 20 s during peak flux. Chemical maintenance cleaning was performed every 1–2 months according to the manufacturer’s instructions.

2.2. Description of treatment strategies

The advanced control system implemented in the SBR-MBR system is designed for flexible use and easy manipulation of desired operating parameters. Membrane performance (i.e., transmembrane pressure (TMP) and permeability) and nutrient removal efficiency (e.g., chemical oxygen demand (COD), total nitrogen (TN), and total phosphorous (TP)) were evaluated by adjusting the duration and frequency ofoxic and anoxic periods during each 2-hour treatment cycle (Fig. 2).
For strategy 1, in order to limit the amount of time required for denitrification during the fill period, a 10 min anoxic phase was implemented at the start of the treatment cycle followed by 4 intermittent aeration periods in each SBR. This sequence is depicted in Fig. 2a. The first aeration period proceeded immediately after the 10 min anoxic phase, followed by intermittent periods of aeration for the duration of the treatment cycle. In strategy 2, we aimed to increase the amount of time required for denitrification by implementing a 20 min anoxic/anaerobic period at the start of the treatment cycle. Thus, the 10-min fill period remained the same; however, the first aeration period was delayed for an additional 10 min, thereby prolonging anoxic/anaerobic conditions (i.e., 20 min anaerobic conditions). Furthermore, a single and longer aeration period was implemented for each React and React-Filtration phase during strategy 2 (Fig. 2b). Each cycle was concluded with a 25 min long anoxic period.

2.3. Data acquisition and analytical procedures

In order to ensure stable operating conditions for each treatment cycle and treatment strategy, supervisory control and data acquisition (SCADA) system was utilized to control and measure a wide range of environmental and process parameters. The most important
parameters measured were fluid flow rates, dissolved oxygen (DO) concentration, TMP, mixed liquor pH and temperature, SBR and MBR total suspended solids concentrations, permeate conductivity, and MLSS levels in the SBRs. The DO concentration was set at a maximum of 1 mg/L for both treatment strategies and was controlled via a process logic controller (PID) for DO probes and aeration devices. Thus, the bioreactor blowers operated at maximum capacity (~1.8 m³/min) in order to reach the 1 mg/L DO set point. If DO concentration increased above the 1 mg/L set point, the bioreactor blowers would decrease flow as shown in Fig. 2.

Weekly influent grab samples were collected at the beginning of each treatment cycle, immediately after the 2 mm fine-screen, during peak flow periods (i.e., between 8 AM and 1 PM). Permeate samples were collected at the end of the treatment cycle from a permeate collection tank with a hydraulic residence time of ~30 min. With the exception of total COD, all influent samples were filtered using 0.45 μm Pall Co. Super-450 filters to measure soluble COD (sCOD), TN, TP, ortho-P, NH₄⁺, and NO₃⁻ concentrations. TN is defined as inorganic nitrogen (NH₄⁺+NO₂⁻+NO₃⁻) plus soluble organic nitrogen using persulfate digestion Test N Tube method. Ammonia was measured via the Nessler method. Nitrate was measured via the dimethylphenol method. Ortho-P and TP were measured with the ascorbic acid method with acid persulfate digestion. Alkalinity was measured using the titration method 8203 (model 16900, Hach, Loveland, CO). Dissolved organic carbon (DOC) was measured using a Sievers 5310 TOC analyzer (GE Analytical Instruments, Boulder Colorado). Calcium, iron, magnesium, and potassium were measured by inductively coupled plasma-atomic emission spectroscopy (ICP-AES) (Optima 5300, Perkin-Elmer, Fremont, CA).

To evaluate membrane performance, the permeability (K) of the membranes was calculated using the recorded TMP and the constant optimal flux (j) of 24 L m⁻² h⁻¹ of the membrane.

\[
K = \frac{j}{\text{TMP}} \quad \text{[L m}^{-2} \text{h}^{-1} \text{bar}^{-1}] 
\]

(1)

In order to compare membrane permeability over the course of the operation period, a permeability temperature correction (K₂₀) was applied:

\[
K_{20} \text{ c} = K_{1} \times 1.025^{20} \text{ c}^{-1} \quad \text{[L m}^{-2} \text{h}^{-1} \text{bar}^{-1}] 
\]

(2)

While this correction is not comprehensive, as reported by Judd [13], the use of temperature normalization is justified in order to compare time series data across different seasonal temperatures (Jiang et al., 2005 [14]).

2.4. Statistical analysis

All data acquired through the SCADA system were extracted and processed using the R software (http://www.r-project.org). Statistical analyses were also conducted in R, including hypothesis testing, outlier identification, and principal component analysis. The latter was carried out in the vegan package, version 2.0-3 [22].

3. Results and discussion

3.1. Start up

The start-up period (days 1 through 89) of the hybrid SBR-MBR process lasted approximately 3 months. It was operated under the strategy 1 operating conditions and was intended to identify and correct any control system errors and mechanical malfunctions. After stable conditions with respect to operating parameters and effluent water quality were achieved, strategy 1 conditions were maintained for additional 122 days (days 90 through 212). From the 213th day, the system was operated under strategy 2 conditions.

Water quality of the SBR-MBR influent and membrane permeate during strategies 1 and 2 operating conditions is summarized in Table 1. Influent quality to the hybrid systems was consistent and within the range of typical water quality values of domestic sewersheds [25]. Influent pH values varied between 6.6 and 8.8 with an average of 7.5.

3.2. Treatment performance under strategies 1 and 2

Treatment strategies 1 and 2 are markedly different in two respects. First, the anaerobic period under strategy 2 is twice as long (20 min) as the anaerobic period under strategy 1 (10 min), and second, strategy 2 consists of two aerobic steps per cycle totaling 60 min, while strategy 1 consists of 4 aerobic steps per cycle totaling 45 min. Despite the inherent differences between strategy 1 and strategy 2, COD removal for both conditions was very low (97.2%) (Table 1) and was not significantly different (p = 0.864, Welch–Satterthwaite method Two Sample t-test).

Nutrient removal efficiencies for strategy 1 were calculated and are summarized in Table 1. Results acquired during this operating period indicate very high removal of COD, moderate removal of TN, and low removal of TP. During testing under strategy 1 conditions, the average mixed liquor pH was 7.5. The aeration scheme for strategy 2 was adopted on the 213th day of the testing period. This configuration involved a 20-min anaerobic period at the start of the cycle, and the final anoxic step was extended to 25 min, thereby preventing any carryover of dissolved oxygen to the beginning of the next SBR cycle. Nutrient removal efficiencies for strategy 2 are also summarized in Table 1. The removal efficiencies of TN and TP substantially increased (92.9% TN and 82.1% TP removal) under strategy 2. During testing under strategy 2 conditions, the average mixed liquor pH was also 7.5.

The SBR-MBR system demonstrated very low removal of phosphorous when operated under strategy 1 conditions (29% removal of phosphate and 33% removal of TP). Yet, phosphorus removal under strategy 2 conditions was significantly greater than strategy 1 conditions (p = 4.52 × 10⁻⁷, Welch–Satterthwaite method Two Sample t-test), achieving an average removal of 82% TP. EBPR is accomplished through sequential anaerobic/aerobic cycling of activated sludge, also referred to as feast/famine conditions [24]. Under anaerobic conditions (i.e., in the absence of oxygen and nitrate), polyphosphate accumulating organisms (PAOs) utilize polyP stores and release soluble phosphorus. In turn, PAOs accumulate PHAs to be later used under aerobic conditions. Under aerobic conditions, PAOs accumulate polyphosphate within their cells and oxidize polyhydroxyalkanoates (PHAs) stores for energy [24]. The significant increase in phosphorus removal during strategy 2 is thus attributed to the extended anaerobic periods at the beginning and end of each two-hour treatment cycle. Furthermore, phosphorus removal did not occur immediately following the treatment transition but was established within 1 week (Fig. 3). However, phosphorus removal was not consistent over the course of strategy 2. This is likely the result of fluctuating sCOD loading over the course of a day, which leaves insufficient available carbon for denitrification during off-peak flow periods (i.e., evening and early morning hours) to completely remove nitrate heterotrophically (Fig. S1). Phosphorus removal is therefore inhibited when nitrate is present in the mixed liquor during off peak flow periods and may explain the inconsistency of phosphorous removal efficiency across strategy 2. Nonetheless, these results demonstrate the feasibility of gaining enhanced biologic phosphorus removal (EBPR) in a decentralized setting within a relatively short acclimation period (i.e., weeks), which has significant practical application for next-generation urban water treatment systems aiming to tailor water quality to local needs.

Under strategy 2 conditions, the SBR-MBR removed 92.9% of TN compared to only 69.9% removal under strategy 1 conditions. Due to three outlying removal values that were observed during strategy 1,
Table 1
Summary of influent and effluent parameters and percent removal during strategies 1 and 2 operating conditions. Additional water quality parameters from ICP analysis can be found in Table S1 in the Supplementary Content.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Steady state—strategy 1 (days 90 through 212)</th>
<th>Steady state—strategy 2 (days 213 through 334)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Influent (n = 18)</td>
<td>Effluent (n = 18)</td>
</tr>
<tr>
<td>COD mg/L</td>
<td></td>
<td>478.1 ± 110</td>
<td>–</td>
</tr>
<tr>
<td>sCOD mg/L</td>
<td></td>
<td>172.1 ± 30</td>
<td>11.1 ± 4.4</td>
</tr>
<tr>
<td>Total N mg/L</td>
<td></td>
<td>57.2 ± 12.0</td>
<td>15.5 ± 13.2</td>
</tr>
<tr>
<td>NH₄-N mg/L</td>
<td></td>
<td>38.0 ± 6.3</td>
<td>&lt; D.L.</td>
</tr>
<tr>
<td>NO₃-N mg/L</td>
<td></td>
<td>0.48 ± 0.26</td>
<td>9.40 ± 4.1</td>
</tr>
<tr>
<td>Total P mg/L</td>
<td></td>
<td>9.6 ± 4.2</td>
<td>6.01 ± 2.3</td>
</tr>
<tr>
<td>PO₄-P mg/L</td>
<td></td>
<td>6.5 ± 3.3</td>
<td>4.18 ± 2.2</td>
</tr>
<tr>
<td>Alkalinity mg/L as CaCO₃</td>
<td></td>
<td>189.1 ± 50</td>
<td>48.6 ± 11.6</td>
</tr>
</tbody>
</table>

D.L.— < Below detection limit.

Fig. 3. Time series influent and effluent (permeate) COD, ortho-phosphate, and nitrogen species (ammonia and nitrate) concentrations collected weekly from the Mines Park water reclamation test site. The time series is divided between strategy 1 and strategy 2 operating conditions. Upon transition from strategy 1 to strategy 2, no change in COD removal was detected. Phosphorus and nitrogen removal efficiencies significantly increased.

nonparametric statistical analysis was chosen to test if strategy 2 was able to achieve significantly greater TN removal rates than strategy 1. Results revealed that strategy 2 achieved significantly greater TN removal rates over strategy 1 ($p=4.1 \times 10^{-5}$, using the Wilcoxon Sign Rank test). Due to the possibility that the outliers (i.e., 22.0%, 13.2%, and 44.6% TN removal) might contribute to a type 1 error, the outliers were removed and new test statistics were computed. The new mean TN removal under strategy 1 was thus calculated to be 78.6%. By satisfying the assumptions for normality and repeating the Welch–Satterthwaite method Two Sample t-test, there is evidence to conclude that the true mean TN removal of strategy 2 is greater than strategy 1 ($p=5.74 \times 10^{-5}$).

Overall, the effluent quality with regard to TN and TP removal under strategy 1 is not suitable for surface water discharge. Furthermore, the TN and TP removal process was observed to be variable. For example, the variation of TN concentration in the permeate stream was relatively high (average of 15.5 mg/L and standard deviation of 13.2 mg/L). This may have been due to slightly lower COD loading during the summer months (June–August) when occupancy of the housing complex was at its lowest—low COD loading may have limited denitrification in the SBRs. However, evidence is insufficient to support this hypothesis, as there is no significant difference in influent COD loading during the periods of strategies 1 and 2 treatments ($p=0.1714$, Welch–Satterthwaite Two Sample t-test). Rather, a better explanation for the poor nutrient removal rates can be partly attributed to the carryover of DO from the end of each treatment cycle to the start of the subsequent treatment cycle, as illustrated in Fig. 2a (i.e., denitrification was delayed until DO was consumed first). In addition to DO carryover from one treatment cycle to the next, the relatively short time period allocated for denitrification during the anoxic/Fill period (10 min), further limited denitrification inside the SBRs. The effects of incomplete nitrate removal are thus two-fold: (1) incomplete removal of NO₃⁻–N prevents the formation of anaerobic conditions required for enhanced biological phosphorus removal [20,25] and (2) partially inhibited denitrification allows for the retention of additional nitrate in the SBR-MBR permeate. Therefore, an aeration scheme that limits denitrification during the SBR Fill phase is a viable option in a
scenario where the retention of nutrients in the effluent is desired, such as for landscape irrigation. Overall, treatment strategy 2 demonstrated stable removal efficiencies; however, in order to achieve more strict effluent discharge requirements for TN and TP, supplemental carbon and coagulant addition may be necessary. Further optimization of the anaerobic zone duration and subsequent aeration cycles may also enhance the nutrient removal process.

The main advantage of the SBR-MBR configuration is that permeate can be produced while the reactor is still in reaction phase. Thus, is it possible to manipulate the process such that the desired nutrients are present only during filtration. Strategies 1 and 2 are thus two examples of how operating parameters (i.e., anaerobic zone duration, oxygen delivery, and number of delivery intervals) can be manipulated in order to tailor effluent nutrient concentrations for seasonal or sight-specific use.

3.3. Membrane performance

Average permeate turbidity over the course of the study period was 0.07 NTU. In order to further evaluate membrane performance over the course of the study, daily average MLSS concentration (g/L), reactor temperature (°C), TMP (bar), and hourly membrane permeability L.m⁻².h⁻¹.bar⁻¹ at 20 °C were plotted as a function of time, as illustrated in Fig. 4. Membrane tank MLSS concentrations fluctuated between 6500 and 9000 mg/L. Reactor temperature showed a clear seasonal trend, with values ranging between 15 °C in the winter and 25 °C in the summer. TMP was relatively constant and stable from May to November (i.e., day 90 to approximately day 277), with averages of 0.084 bar and 0.086 bar for membranes 1 and 2, respectively, at which point an increase in TMP was observed. The sudden increase in TMP is interpreted in terms of two possible variables, MLSS concentrations and reactor temperature. To explore this relationship, Principal Component Analysis (PCA) correlation biplot (Fig. 5) was used to visualize and interpret the massive multivariate dataset (>300,000 observations) collected by the SCADA system during the period of the study. Specifically, a PCA correlation biplot was used in conjunction with the dataset to (1) explain how observations, or principal component scores, cluster relative to each other, and (2) elucidate how each operating parameter (e.g., TMP, MBR MLSS concentration, and sludge temperature), or arrow, controls the locations of those observations. Furthermore, the angle between arrows in the biplot indicates their correlations such that projections in the same direction are positively correlated, projections in opposite directions are negatively correlated, and orthogonal projections have a correlation close to

Fig. 4. MBR performance as a function of time during treatment strategies 1 and 2. Panel (a) shows the fluctuations in MBR MLSS in g/L over the course of the study period. Panel (b) shows the change in temperature in °C over the course of study period. Panels (c) and (d) show TMP (bar) and membrane permeability at 241 m².h⁻¹.bar⁻¹, respectively. Stars in panel (d) indicate when membrane maintenance cleaning was performed.
zero. Results illustrated in Fig. 5 reveal that the combined proportion of variance explained by the first pair of axes is 85.5% (60.2% and 25.3% for PC1 and PC2, respectively), indicating that the first two principal components accurately represent the data in 2-dimensional space [3]. Also in Fig. 5, the observations are binned by season such that points colored in black represent spring/fall months, red points represent summer months, and green points represent winter months. Results show that summer observations (red) cluster together, winter (green) observations are clearly separated from summer, and spring/fall observations (black) lie in between. The separation of results along this gradient is a result of seasonal temperatures and TMP, which project outward from each other along axis 1—indicating a strong, negative correlation. Along axis 2 there is no visible trend for the impact of MLSS concentration on TMP. Furthermore, in order to quantify the significance of these results Pearson product–moment correlation coefficients were calculated. Results indicate that the effect of reactor temperature on TMP was highly significant ($p = 2.2 \times 10^{-16}$), while the variations in TMP did not significantly correlate with membrane tank MLSS concentrations ($p = 0.1969$, at a 95% confidence interval). Therefore, these results highlight and reinforce the importance of water temperature as a major operating parameter affecting TMP and membrane performance for distributed systems at the scale reported in this study.

As a result of the sudden TMP increases that were due to seasonal temperature shifts (i.e., during November and December), membrane permeability was also adversely affected, declining to approximately 100 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ by late-November. In response to the declining membrane permeability, a membrane maintenance cleaning was conducted but only partially restored permeability for the duration of the studied period. Inspection of the membrane modules post-study period revealed that membrane clogging, or sludging, was evident and likely explains the rise in TMP and decline in membrane permeability. The cause of the membrane sludging is likely due to increasing MLSS viscosity from decreasing temperature during colder temperatures as reported by Judd [13]. Winterization of the SBR-MBR before the following winter (enclosure in an insulated barn) resolved the operational challenges of temperature effects on TMP.

In decentralized settings the effects of temperature on membrane performance must be considered and proper infrastructure (i.e., thermal insulation) must be in place to ensure stable operation. Nonetheless, in our study membrane permeability averaged 284 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ for membrane cassette 1 and 256 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ for membrane cassette 2, and is comparable to other studies operating a reactor under a similar flux range [4,28].

And lastly, the transition from treatment strategy 1 to strategy 2 did not have any apparent negative impacts on membrane performance (i.e., permeability or TMP). This indicates that the manipulation of anaerobic and aerobic periods within SBR treatment cycles does not have adverse effects on membrane fouling. Therefore, an SBR-MBR hybrid treatment system is capable of achieving different effluent qualities within the same operating footprint, without compromising membrane performance, and it offers new options for water supply planning and water reuse within the urban environment.

4. Summary

This study reports the first decentralized/satellite application of a hybrid sequencing batch membrane bioreactor (SBR-MBR) implemented in an urban setting and operated in a tailored water reuse mode. Our results reveal that a flexible wastewater treatment system can be used to achieve different effluent quality without compromising the performance of membrane operation. Thus, a flexible treatment system has great potential to tailor effluent qualities to local urban water demands, such as for landscape irrigation, household service water, stream flow augmentation, or groundwater recharge. Further research must aim at assessing the level of predictability, resilience of system performance to changing treatment conditions, and long-term robustness of flexible treatment systems.

Author disclosure statement

No competing financial interests exist.

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Appendix A. Supplementary materials

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.memsci.2013.06.021.

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