Effect of disintegrated sludge recycling on membrane permeability in a membrane bioreactor combined with a turbulent jet flow ozone contactor

Byung-Kook Hwang a, Jae-Hyuk Kim a, Chang Hoon Ahn a, Chung-Hak Lee a,*, Jae-Yoon Song b, Young-Hyun Ra b

a School of Chemical and Biological Engineering, Seoul National University, Seoul, Republic of Korea
b OXEN Tech, Gangnam-gu, Seoul, Republic of Korea

1. Introduction

Membrane bioreactor (MBR) processes are among the most economical and promising technologies for domestic and industrial wastewater treatment. Nevertheless, several drawbacks exist in the MBR process, such as abiotic fouling and biofouling, which prevent the technology from gaining wider applicability (Al-Halbouni et al., 2008). In addition, although the MBR process generates lower amounts of biomass (or excess sludge) than does the conventional activated sludge process, due to a relatively high solids retention time (SRT), the excess sludge produced in the MBR creates a concurrent sludge management problem (Oh et al., 2007). Therefore, the development of an efficient means to reduce excess sludge in the MBR remains a formidable challenge.

One solution for the excess sludge problem is the installation of a pretreatment system prior to the MBR. Various pretreatment methods have been reported for the reduction of excess sludge, including thermal, ultrasonic, mechanical, alkaline, and oxidative technologies (Muller et al., 1998; Low and Chase, 1999; Bougrier et al., 2006). Among these approaches, ozonation is considered to be one of the most efficient means because of its high sludge disintegration and solubilization capability (Muller, 2000; Yasui and Shibata, 1994; Lee et al., 2005). Furthermore, it has been reported that the ozonated sludge has the ability to improve the membrane permeability.
permeability of the mixed liquor (Oh et al., 2007; Huang and Wu, 2008; Williams and Pirbazari, 2007). In particular, Huang and Wu (2008) presumed the re-flocculation mechanism with EPS structure in the MBR by ozonation. However, the exact mechanism of the flux enhancement has yet to be elucidated.

In this study, we established an MBR hybrid system combined with a turbulent jet flow ozone contactor (TJC), which maximizes the ozone mass transfer and, thus, enhances the energy efficiency for sludge reduction. In the TJC-MBR hybrid system where the whole disintegrated excess sludge was recycled to MBR, we investigated mechanisms of the membrane flux enhancement in two ways: i) classification and quantification of EPS [loosely bound (LB) and tightly bound (TB) EPS] and ii) quantitative and qualitative analysis of bio-cake structures formed on the membrane surface using confocal laser scanning microscopy (CLSM) and image analysis techniques.

2. Materials and method

2.1. TJC-MBR system

Fig. 1 presents a schematic of the control- and TJC-MBRs. Two MBRs, each having a working volume of 50 L, were operated in parallel at room temperature 25 °C. The same synthetic wastewater (see Table 1 for its composition) was fed into both MBRs, except that, after being disintegrated by the TJC system, all of the excess sludge was recycled only to the TJC-MBR. In each MBR, one bigger membrane module (area: 0.38 m²/module) or one or two smaller modules (area: 0.05 m²/module) were mounted vertically (Fig. 1). The bigger modules were used to maintain the HRT, whereas the smaller modules were taken out of the reactor one by one at the specified operating time to analyze bio-cakes formed on the membranes. Each module was made of polyvinylidene fluoride (PVDF) hollow fiber membranes (Ge-Zenon, US) having a pore size of 0.04 µm. A coarse bubble diffuser was placed at the bottom of each MBR to maintain a dissolved oxygen concentration of 2.5–3.0 mg/L. The filtration was stopped when the TMP reached 30 kPa. All the membrane modules in each reactor were operated continuously at the constant flux of 13 L m⁻² h⁻¹. The permeates from all the modules in each reactor were combined and a portion of the combined permeate was recycled to each MBR to maintain a constant working volume of 50 L so that the hydraulic retention time (HRT) was maintained at 10 h in each MBR.

Once a day, 2.5 L of the mixed liquor in both MBRs was removed to maintain a sludge retention time (SRT) of 20 days. The excess sludge from the TJC-MBR was collected and preserved at 4 °C until the total accumulated volume reached 30 L (Nagare et al., 2008). The collected sludge was disintegrated in the TJC system in a batch mode and then 2.5 L of this treated sludge was recycled to the TJC-MBR once a day; in contrast, the excess sludge from the Control-MBR was discarded.

2.2. TJC system

The laboratory-scale TJC (OXEN Tech Co., Ltd., Seoul, Korea) comprised five major compartments (Fig. 1): i) an ozone and liquid contactor (working volume: 30 L) equipped with a baffle plate, ii) a venturi injector, iii) a circulation pump, iv) oxygen.

Table 1 – Composition of synthetic wastewater.

<table>
<thead>
<tr>
<th>Component</th>
<th>Concentration (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glucose</td>
<td>160</td>
</tr>
<tr>
<td>Bacto Peptone</td>
<td>120</td>
</tr>
<tr>
<td>Yeast Extract</td>
<td>16</td>
</tr>
<tr>
<td>(NH₄)₂SO₄</td>
<td>104</td>
</tr>
<tr>
<td>KH₂PO₄</td>
<td>33</td>
</tr>
<tr>
<td>MgSO₄·7H₂O</td>
<td>32</td>
</tr>
<tr>
<td>MnSO₄·4H₂O</td>
<td>2.8</td>
</tr>
<tr>
<td>FeCl₃·6H₂O</td>
<td>0.1</td>
</tr>
<tr>
<td>CaCl₂·2H₂O</td>
<td>3.2</td>
</tr>
<tr>
<td>NaHCO₃</td>
<td>220–500</td>
</tr>
<tr>
<td>Soluble COD</td>
<td>280</td>
</tr>
</tbody>
</table>
and ozone generators and \( \text{v) an off-gas ozone destructor.} \) Oxygen was converted into ozone through an ozone generator (VMUS-08E, AZCO Industries, Ltd., Langley, Canada). The excess sludge withdrawn from the TJC-MBR was fed to the TJC for sludge reduction in batch mode. The sludge was recycled, passing through the orifice of the venturi injector (5 mm in diameter) at a flow rate of 30 (±1) L min\(^{-1}\) to generate hydrodynamic cavitation. As long as the mixed liquor was being circulated, the ozone gas was suctioned automatically and continuously into the contactor via the venturi injector as a result of the development of negative pressure. During TJC operation, samples were removed periodically from the TJC to monitor the degree of sludge decomposition, which was subdivided into sludge disintegration, solubilization, and mineralization. The optimal ozone contact time for excess sludge treatment was 18 min. At the end of this time, the ozone transfer efficiency was ca. 99% and the input ozone dosage per MLSS was 12.9 mg-O\(_2\) g-MLSS\(^{-1}\).

Hydrodynamic cavitation can be defined as the generation, subsequent growth, and collapse of cavitation bubbles, releasing high wave energy at the surface of the bubbles for a very short time. The cavitation bubbles implode within milliseconds under the high pressure surrounding them, releasing tremendous amounts of energy in the form of shock waves (Kumar and Pandit, 1999). Highly oxidizing radicals (e.g., HO\(_2\)) can be generated from these waves, analogous to those formed through advanced oxidation processes (AOPs) (Benito et al., 2005).

A dimensionless cavitation number is used to characterize the process; under ideal conditions, hydrodynamic cavitation is generated when its value is less than 1 (Kumar and Pandit, 1999). In the TJC system, hydrodynamic cavitation was achieved by forcing the excess sludge through an annular opening that combined a narrow entrance orifice with a much larger exit orifice (Fig. 1); the cavitation number for the TJC system was calculated to be within the range 0.24–0.25. Moreover, the baffle plate at the lower part of the contactor could smash the fluid, causing a certain portion of it to flow back, thereby reinforcing the turbulent flow. In this way, the TJC system maximized both the disintegration of the sludge particles and the ozone transfer efficiency.

### 2.3. Analytical methods

The mixed liquor suspended solids (MLSSs) and mixed liquor volatile suspended solids (MLVSSs) were measured using standard methods (APHA et al., 2005). The total COD (TOC) and soluble COD (SCOD) of the mixed liquor were measured using a DR4000 spectrophotometer (Hach Co., Denver, CO, USA) and corresponding reagent kits. The ozone concentration in the gas phase was monitored using an ozone gas analyzer (H1-S, IN USA Corp., Norwood, MA, USA). The particle size distribution of the microbial flocs (i.e., sludge particles) in the mixed liquor was monitored using a particle size analyzer (MasterSizer/E, Malvern Instruments, Ltd., Malvern, UK). The zeta potential (ζP) of the sludge particles in the supernatant of the mixed liquor was determined using an electrophoretic light scattering method (ELS-Z, Otsuka, Japan).

The molecular weight distribution of the soluble components was measured using gel filtration chromatography (GFC, Waters, Milford, MA). A column (Ultrahydrogel 120, Waters, Milford, MA) was employed in conjunction with a refractive index detector (RID, Waters 2414). Filtered, degassed, and distilled water incorporating 0.01 mM NaNO\(_3\) was used as the mobile phase at a flow rate of 0.6 mL min\(^{-1}\). Samples were filtered through 0.45-μm filters prior to injection into the GFC.

### 2.4. Extraction and analysis of LB-EPS and TB-EPS

A two-step heat extraction method was adopted to extract LB-EPS and TB-EPS from the sludge sample (Li and Yagn, 2007). A mixed liquor (50 mL) from the MBR was dewatered through centrifugation (Mega 17R, Hanil Science Industrial, Ltd., Korea) at 4000g for 5 min. The sludge pellet in the 50-mL centrifugal tube was resuspended in 0.05% NaCl solution at 50 °C and then immediately sheared by a vortex mixer (G-560, Scientific industries, Inc., NY, USA) for 1 min. The sludge suspension was then centrifuged at 4000g for 10 min, and the organic matter in the supernatant was regarded as the readily extractable EPS or the LB-EPS of the sludge mass. For further TB-EPS extraction, the sludge pellet was resuspended in 0.05% NaCl solution to its original volume of 50 mL. The sludge suspension was heated at 60 °C in a water bath for 30 min and then it was centrifuged at 4000g for 15 min. The organic material in the supernatant was regarded as the TB-EPS. The LB-EPS and TB-EPS extracts were further analyzed for their protein and polysaccharide contents, measured using the modified Lowry and phenol/sulfuric acid methods, respectively.

### 2.5. Dead-end filtration

A stirred batch cell (8200, Amicon, USA) was used to filter resuspensions of sludges with various fractions of LB-EPS. The fresh polyvinylidene fluoride (PVDF) membrane (pore size: 0.2 μm) was rinsed prior to filtration to remove any impurities. For all dead-end filtration tests, the TMP was fixed at 0.3 bar using N\(_2\) gas; the flux was measured under a stirring speed of 180 rpm.

### 2.6. Bio-cake staining and image analysis

A fraction (length: 3 cm) of the membrane fibers located in the middle section of the membrane module were removed and the bacterial cells and EPSs on the bio-cake were stained using a mixture of fluorescent dyes specific to each component. SYTO 9 (Molecular Probes, Eugene, OR, USA; excitation (ex) = 488 nm; emission (em) = 515/30 nm), a cell-permeable nucleic acid dye, was used to visualize all cells. Concanavalin A (ConA) lectin (Molecular Probes, Eugene, OR, USA; ex = 568 nm; em = 600/50 nm) conjugated with tetramethylrhodamine isothiocyanate (TRITC) was used to bind to the α-mannopyranosyl and α-glucopyranosyl sugar residues (polysaccharides) in the bio-cake. After incubating the stained bio-cake for 30 min in the dark at room temperature, the excess staining solution was removed by washing with phosphate-buffered saline (PBS) after each staining step.

Specimens of the stained bio-cakes were immediately observed using CLSM. A scanning confocal system (C1plus, Nikon, Japan) was used to obtain images of the bio-cakes nondestructively. The bio-cakes were observed using a 60× objective lens.
Table 2 – Characteristics of the mixed liquors in the Control-MBR, TJC-MBR, and disintegrated sludge.

<table>
<thead>
<tr>
<th>Parameter (units)</th>
<th>Control-MBR</th>
<th>TJC-MBR</th>
<th>Disintegrated sludge</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>6.9–7.6</td>
<td>6.8–7.2</td>
<td>5.7–6.2</td>
</tr>
<tr>
<td>Total COD (mg/L)</td>
<td>9050 (±150)</td>
<td>9220 (±170)</td>
<td>8480 (±180)</td>
</tr>
<tr>
<td>Soluble COD (mg/L)</td>
<td>80 (±20)</td>
<td>72 (±18)</td>
<td>2460 (±70)</td>
</tr>
<tr>
<td>MLSS (mg/L)</td>
<td>7650 (±350)</td>
<td>7820 (±260)</td>
<td>4840 (±140)</td>
</tr>
<tr>
<td>MLVSS (mg/L)</td>
<td>5950 (±310)</td>
<td>6020 (±280)</td>
<td>2830 (±170)</td>
</tr>
<tr>
<td>ZP of sludge in supernatant (mV)</td>
<td>–20 (±3)</td>
<td>–8 (±2)</td>
<td>12 (±3)</td>
</tr>
<tr>
<td>Soluble COD in permeate (mg/L)</td>
<td>14 (±5)</td>
<td>17 (±4)</td>
<td>–</td>
</tr>
<tr>
<td>Removal efficiency of soluble COD (%)</td>
<td>95 (±3)</td>
<td>94 (±4)</td>
<td>–</td>
</tr>
</tbody>
</table>

Values in parenthesis are standard deviations (n = 10).
TJC-MBR sludge (d = 213 µm) was ca. 20% larger than that of the Control-MBR sludge (d = 167 µm).

The ZP of the disintegrated sludge in its supernatant had a positive value (+12 mV), whereas both the MBR sludges in their supernatants were negatively charged (Table 1). Interestingly, the TJC-MBR sludge was less negatively charged (ZP = −8 mV) than the Control-MBR sludge (ZP = −20 mV). These ZP values suggest that when the positively charged disintegrated sludge was recycled into the TJC-MBR, it neutralized the negatively charged sludge particles in the TJC-MBR, allowing them to agglomerate through electrostatic attraction. As a result, the fraction of smaller particles (10–200 µm) in the TJC-MBR was reduced, whereas that of larger particles (200–500 µm) was augmented (Fig. 3). Because an increase in the microbial floc size enhances the membrane permeability in the MBR (Hwang et al., 2007), this phenomenon might be one of the key factors affecting the enhancement of filterability in the TJC-MBR system, as indicated in Fig. 2.

3.4. Changes in the concentrations of LB-EPS and TB-EPS through the TJC

It is generally recognized that EPS is closely associated with the physiological states of activated sludge as well as the architecture of bio-cake and, thus, it affects the membrane permeability in the MBR (Chang and Lee, 1998; Yun et al., 2006; Hwang et al., 2008). EPS in mixed liquors is composed of soluble EPS and bound EPS. The bound EPSs exhibits a dynamic-fractioned structure and is made up of LB-EPS and TB-EPS (Shao et al., 2009). Li and Yagin (2007) also suggested that an excess of LB-EPS might deteriorate cell attachment and weaken the microbial floc structure, leading to a poor sludge-water separation. Therefore, we extracted soluble and bound EPS (LB- and TB-EPS) from the mixed liquors in both MBRs and disintegrated sludge, respectively, and then analyzed and compared them both qualitatively (Fig. 4) and quantitatively (Fig. 5).

First, we used GFC to analyze all types of EPS, as presented in Fig. 4. For all the samples of LB-EPS (Fig. 4a), we observed peaks at a retention time of 5.6 min. The peak at the retention time of 6 min, which was clearly observed in the Control-MBR, had, however, nearly disappeared in the analyses of the TJC-MBR and disintegrated sludge. In particular, the peak intensity at the retention time of 5.6 min for the disintegrated sludge was drastically reduced, suggesting that most of the LB-EPS had been decomposed by the TJC. Because all the decomposed sludge containing a negligible amount of LB-EPS was recycled to the MBR and could be recombined with the existing sludge, the LB-EPS content in the TJC-MBR sludge ultimately became less than that in the Control-MBR sludge. Unlike the case for the LB-EPS, the level of TB-EPS in the disintegrated sludge was reduced, but it did not become negligible (Fig. 4b). On the contrary, some peaks of TB-EPS representing the TJC-MBR in which the disintegrated sludge was recycled had a bit higher intensities than those representing the Control-MBR. This phenomenon implies that recycling of the disintegrated
sludge could change both the LB-EPS and TB-EPS contents in the TJC-MBR.

Second, to determine the fine details of these changes in the LB- and TB-EPS, we performed quantitative analysis of the EPS (Fig. 5). The soluble EPS concentrations were similar—13 and 14 mg/g VSS for the Control-MBR and TJC-MBR, respectively—whereas the concentration of total bound EPS (i.e., LB-EPS + TB-EPS) increased slightly (Fig. 5a) from 88 mg/g VSS (Control-MBR) to 99 mg/g VSS (TJC-MBR). Breaking the total bound EPS into LB- and TB-EPS (Fig. 5b), we found that the LB-EPS in the disintegrated sludge became negligible (2 mg/VSS), whereas the TB-EPS remained at a considerable level (41 mg/VSS) even after the ozone treatment with the TJC. When the disintegrated sludge was recycled, the LB-EPS decreased by ca. 50%—from 26 mg/g VSS (Control-MBR) to 14 mg/g VSS (TJC-MBR)—whereas the TB-EPS increased from 62 mg/g VSS (Control-MBR) to 85 mg/g VSS (TJC-MBR). These changes coincided well with those observed through GFC analysis (Fig. 4).

To elucidate the causes of these interesting phenomena, we measured the ZPs of sludge particles in the supernatants of both the MBRs, respectively. Table 2 reveals that the ZP reversed from −8 mV (TJC-MBR) to +12 mV (disintegrated sludge) after ozonation. Although the pH of mixed liquor can affect zeta potential of microbial sludge, we thought that a slight difference in the pH (~1) between the mixed liquors in TJC-MBR and the disintegrated sludge could not necessarily reflect such a significant change of zeta potential (~20 mV). Consequently, we suggested that the ozonation could not only readily destroy the LB-EPS that was exposed outward but also reverse the surface charge of the ozonated sludge.

Based on these analytical data, we hypothesize that positively charged disintegrated sludge particles featuring negligible amounts of LB-EPS would combine readily with the LB-EPS of the negatively charged sludge particles in the MBR and, consequently, the existing LB-EPS would be incorporated within the sludge particles to become TB-EPS. This phenomenon would explain not only why the ZP increased from −20 mV for the Control-MBR to −8 mV for TJC-MBR but also why the content of LB-EPS decreased, whereas that of TB-EPS increased, in the TJC-MBR in which the ozonated sludge (e.g., disintegrated sludge) was recycled totally.

Furthermore, this hypothesis led us to suspect that the reduced amount of LB-EPS and the increased ZP (e.g., the decreased absolute value of ZP) in the TJC-MBR might be closely related to the slower TMP rise up (e.g., less biofouling) in the TJC-MBR than in the Control-MBR, as revealed in Fig. 2. Therefore, we investigated the effect of the LB-EPS on the membrane permeability in more detail.

### 3.5. Effect of the LB-EPS on membrane permeability

A portion (50 mL) of excess sludge from the TJC-MBR was divided into two fractions: a sludge pellet of microorganisms containing only the TB-EPS (first fraction) and an extracted solution of the LB-EPS only (second fraction). We repeated this procedure three times to obtain three sets, each of the sludge pellet with TB-EPS and the extracted solution of the LB-EPS. Next, three LB-EPS solutions containing different LB-EPS concentrations were prepared with the second fractions, namely 1x, 1/2x, and 1/3x suspensions (1x: the whole amount of LB-EPS extracted from the 50 mL of excess sludge). The three LB-EPS solutions were added into three volumetric flasks, each containing the first fraction, and then each flask was diluted to the same final volume (50 mL).

Filtration using the stirred cells was conducted for the three suspensions with different LB-EPS concentrations to investigate the effect of the LB-EPS content on membrane fouling (e.g., membrane permeability). As indicated in Fig. 6, all the fluxes declined rapidly within the first 5 min, reaching a steady state after 10 min. At the steady state, the fluxes for the three feed suspensions were ca. 24, 29, and 35 L m⁻² h⁻¹, respectively, depending on the LB-EPS concentrations (1x, 1/2x, and 1/3x). In other words, the lower the content of LB-EPS (1/3x), the higher the steady state flux (35 L m⁻² h⁻¹). Although the three suspensions are different in their total EPS content (max. 10%, Fig. 5b), the difference in the total EPS could not necessarily reflect the gap of the steady state flux (max. 45%, Fig. 6) by exactly that extent.

However, this result was expected because the lowest LB-EPS content (1/3x) gave rise to the greatest mean particle size (d = 110 μm), whereas the highest LB-EPS content (1x)
provided the smallest particle size (\(d = 92\) \(\mu\)m) among the three feed suspensions. Li and Yagn (2007) suggested that the LB-EPS could make cells slippery as a result of the higher content of water bound to the cells, which might inhibit re-flocculation.

Returning to Figs. 3, 5 and Table 2, we recognize that the TJC-MBR provided a lower LB-EPS (14 vs. 26 mg/VSS) as well as a lower absolute ZP (\(-8\) vs. \(-20\) mV), but a higher mean particle size (213 vs. 167 \(\mu\)m), relative to those of the Control-MBR. Accordingly, it is evident that a strong correlation exists between the content of LB-EPS, the ZP, and the particle size, all of which are in close association with the membrane permeability. In summary, the TJC system generated an ozonated sludge having a positive charge as well as a negligible amount of LB-EPS. As a result, when this ozonated sludge was recycled and combined with the negatively charged sludge in the TJC-MBR, it not only reduced the content of LB-EPS but also increased the size of the sludge particles (e.g., microbial flocs) in the TJC-MBR. Accordingly, the recycling of ozonated sludge retarded the TMP rise-up in the TJC-MBR, as indicated in Fig. 2.

### 3.6 Analysis of the bio-cake structure

As stated above, the recycling of the disintegrated sludge to the TJC-MBR dramatically increased membrane permeability and reduced membrane biofouling (Fig. 2). To identify the main properties of the bio-cake that could be closely associated with membrane permeability (Hwang et al., 2008; Yun et al., 2006), we recorded CLSM images of samples of the bio-cake corresponding to each point in Fig. 2. Table 3 summarizes the textural (porosity) and volumetric (biovolume, average thickness) parameters determined from the 12 different CLSM image stacks of each bio-cake.

For the TJC-MBR, points 2 and 3 suggest that the longer the operating time, the higher the TMP reached (13 vs. 30 kPa), because the biovolume and the average thickness of bio-cake at point 3 (4.3 \(\times\) 10^6 \(\mu\)m^3 and 42 \(\mu\)m, respectively) were much greater than those at point 2 (2.4 \(\times\) 10^6 \(\mu\)m^3 and 25 \(\mu\)m, respectively). Because the porosity difference between points 2 (0.78) and 3 (0.74) was not remarkable, the TMP increase in the TJC-MBR appears to be attributable mainly to growth of the bio-cake in the vertical direction. In other words, the loss of filterability in the TJC-MBR was not caused by the structural change of the bio-cake, but by the simple accumulation of biomass onto the membrane upon increasing the filtration time.

After the same operating period (8 days), however, the Control-MBR already reached the TMP of 30 kPa (point 1 in Fig. 2), whereas the TJC-MBR reached only 13 kPa (Point 2 in Fig. 2). Taking into account the fact that the biovolume and average thickness of the bio-cake at point 2 (2.4 \(\times\) 10^6 \(\mu\)m^3 and 25 \(\mu\)m, respectively) were slightly greater than those at point 1 (2.2 \(\times\) 10^6 \(\mu\)m^3 and 19 \(\mu\)m, respectively), we suspect that the difference in membrane permeability between the two MBRs could be attributed to the difference in architecture of bio-cake: a greater porosity (0.78) of bio-cake in the TJC-MBR than that in the Control-MBR (0.58).

In summary, the recycling of disintegrated sludge with the TJC induced a series of consecutive changes in the mixed liquor and on the bio-cakes in the TJC-MBR: a decrease in the LB-EPS content of the sludge particles, an increase in the size of the sludge particles in the mixed liquor, an increase in the porosity of the bio-cake that accumulated on the membrane, an increase in the membrane permeability, and a decrease in the rate of TMP rise. In essence, we attribute the enhanced membrane permeability in the TJC-MBR to the decreased LB-EPS content of the sludge particles when using the TJC system.

### 4. Conclusions

We have constructed an MBR hybrid system combined with a TJC to investigate the effect of disintegrated sludge recycling on membrane permeability and its mechanism. The TJC system not only generated ozonated sludge containing a negligible amount of loosely bound EPS but also reversed the zeta potential of the sludge from negative to positive. As a result, when this ozonated sludge was recycled and
combined with the negatively charged sludge, the size of the sludge particles (e.g., microbial flocs) increased in the TJC-MBR. Consequently, the bio-fake that formed in the TJC-MBR had greater porosity than that in the Control-MBR, resulting in higher membrane permeability in the TJC-MBR, which we attribute ultimately to the decreased loosely bound EPS content of the sludge particles when using the TJC system.

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**References**


