

Evaluating RO performance with biological pretreatment of graywater

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ABSTRACT

Reverse osmosis (RO) is a treatment technology that may be used for the recovery of graywater terrestrially as well as the recovery of wastewater on board long duration, manned space flights. As with terrestrial RO applications, concentration polarization and membrane fouling lead to decreased productivity and increased energy demands with time. Physiochemical and biological pretreatment options can enhance the performance of the RO system; also, biological pretreatment has the advantages of low energy and consumable requirements as compared to other physiochemical pretreatment options. To determine the degree to which the incorporation of biological treatment enhances RO performance, a series of bench-scale experiments were performed. Observed resistances indicate that biological pretreatment alleviated the degree of non-reversible fouling while also improving the rate of permeate flux in a downstream RO process to further treat water. Results also indicated that enhanced urea hydrolysis, pH reduction, and carbon oxidation serve as the primary benefits of biological pretreatment of space-based graywater. The observations of this study may also benefit engineers considering the treatment of graywaters for terrestrial water-reuse applications.

Key words | flux, graywater, resistance, reverse osmosis, space

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INTRODUCTION

As urbanization, droughts, and falling water tables continue to strain existing water resources, the need for creative water management practices will increase (Levine & Asano 2004; Maurer *et al.* 2006), and engineers and scientists must seek means to relieve stress on water supplies. One approach attracting attention is water reuse. Levine & Asano (2004) go as far as to say that ‘society no longer has the luxury of using water only once’. By simply viewing wastewater as a valuable commodity with reuse potential rather than a substance of little or no value, the amount of water available for use on the mission is vastly increased.

Where you find mankind, you will also find water. Even as mankind travels beyond the confines of Earth, water will join in the travel. The same terrestrial challenges presented by water reuse also occur in spaceflight travel. For long

duration space missions, the importance of wastewater recovery and reuse increases with mission duration (Jones 2007). Recovery and treatment of wastewater to potable water standards provides a viable alternative to direct re-supply, due to the reduction in both mass and stowage demands. For the purposes of wastewater collection and treatment, one possible waste stream that can be generated based on the specific mission is the early planetary base wastewater (EPBW). The EPBW (Verostko *et al.* 2004) is composed of urine flush water, humidity condensate, and hygiene waste. The hygiene waste stream contains a large amount of surfactants, and the humidity condensate contains several organic compounds that account for a small majority of organic carbon in the EPBW. The urine waste stream provides the vast majority of nitrogen and organic

carbon, and trace materials to the final composition of the EPBW. EPBW has a higher pH and higher concentrations of dissolved organic carbon (DOC), total nitrogen (TN), sodium, and ammonium ($\text{NH}_4^+\text{-N}$) (Verostko *et al.* 2004) than traditional wastewater, resulting in a waste stream with unique water qualities.

Reverse osmosis (RO) is a technology well suited for the treatment of EPBW. RO can effectively remove substances of concern at the molecular level such as bacteria, viruses, organic compounds, and salts (Mulder 1996). Unlike other physiochemical treatment processes that would require a constant supply of chemicals to achieve treatment of the EPBW, the only consumables required for the treatment combination of a bioreactor and an accompanying RO system are additional membrane modules and an oxygen supply.

Pretreatment options can enhance RO system performance by reducing the degree of concentration polarization and fouling (Mulder 1996) and also by decreasing the bulk concentration of constituents in the water. Potential pretreatment techniques include both physiochemical and biological treatment processes. Physiochemical pretreatment options include low pressure membrane processes (i.e. microfiltration) and pH adjustment. Physiochemical separation technologies improve RO performance by reducing the mass loading of wastewater constituents to the RO membrane, which decreases the degree of concentration polarization and fouling (Yoon & Lueptow 2005). The use of pH adjustment chemicals can also prevent precipitate fouling (i.e. scaling) within the RO system. Although physiochemical pretreatment options may be effective, they require consumables; thus, physiochemical pretreatment alone is not ideally suited for EPBW recovery. Alternatively, the use of bioregenerative pretreatment options can reduce organic mass loading and maintain a more stable effluent pH while simultaneously reducing TN and pH to downstream processes all while possessing a low consumable demand (Garland *et al.* 2005; Jones 2007). Also, the EPBW requires a relatively low amount of oxygen compared to crew requirements to make gas supply a major operational limitation. Thus, the use of biological systems is well suited for pretreatment of EPBW. Potential biological pretreatment options include aerobic oxidation, nitrification, and denitrification.

The purpose of this research was to quantify the impact of biological pretreatment on a graywater waste stream (EPBW), as well as to evaluate the impact on RO operation. RO membrane performance was assessed by measuring permeate flux, membrane resistance, fouling, and salt rejection. The results of this study can benefit engineers that are considering the treatment of graywaters for both space and terrestrial water reuse applications by evaluating the impact of biological treatment on RO performance for an EPBW model wastewater ersatz solution.

MATERIALS AND METHODS

Two biological treatment systems were employed to assess the impact of biological treatment of EPBW graywater generated during space travel on subsequent RO membrane performance. The first biological treatment system used was an aerobic oxidation–nitrification reactor, which treated an EPBW made up of three ersatz solutions to represent hygiene, urine, and humidity condensate. The second biological treatment system, a denitrification–nitrification reactor system, treated an EPBW made from real urine and ersatz hygiene and humidity condensate streams. The work herein evaluates the impact of biological treatment of each of the four EPBW streams on RO performance.

A bench-scale aerobic oxidation–nitrification fixed-film bioreactor was constructed to generate biologically treated water for the RO experiments (Landes 2007; Syron & Casey 2008; Downing *et al.* 2010; Jackson *et al.* 2009). The bioreactor was inoculated with aerobic heterotrophs and *nitrosomonas* for organic and ammonium oxidation, respectively. The advanced membrane bioreactor was constructed of acrylic with Silastic® hollow fiber membranes on the interior to physically separate the aqueous and air phases. The hollow fiber membranes served as the attachment surface, for biological growth. The reactor contained 150 membranes, each 0.91 m long with inner and outer diameters of 0.102 and 0.216 cm, respectively. The bioreactor was 25.4 cm tall with an inside diameter of 10.2 cm, with a resulting working volume of 1,557 mL. The reactor was configured so that the aqueous and air phase were both in up-flow mode. The Silastic® membranes allowed for gas to diffuse from the air phase to the aqueous

phase within the reactor (Suzuki *et al.* 1993). The bench-scale setup contained a feed tank and effluent tank, a raw feed pump (operated at 0.1 mL/min) and a recycle pump (operated at 2.0 mL/min). Facility air was used in all experiments with an operating lumen pressure of 2.5 psi.

The EPBW ersatz was prepared according to Verostko *et al.* (2004). The ersatz recipe accounts for inorganic and organic fractions of humidity condensate, hygiene water, and urine. The procedure used by Verostko *et al.* (2004) to make the ersatz solutions calls for the use of Geroon TC-42 (previously Igepon TC-42) to make the hygiene water condensate; however, the EPBW ersatz included an alternative to the Geroon TC-42. This alternative contained 8.11% sodium laureth sulfate (SLES), 7.10% cocoamphodiacetate, and the remainder was distilled, deionized (DDI) water (Jackson & Morse 2005). The influent and effluent of the system was monitored for pH, dissolved oxygen, DOC, TN, nitrite, nitrate, and ammonia. As the effluent was collected for RO experiments, it was transferred to a separate container (triple rinsed with DDI) and stored at 4 °C until enough effluent was collected to perform an RO experiment.

The Texas Tech University Water Recovery System (TTU-WRS) operated for 6 years. The system was composed of a denitrifying packed bed attached-growth reactor located upstream from a nitrifying tubular reactor. A recycle line linked the two reactors supplying the packed bed with the required terminal electron acceptor for treatment. Additional information regarding the performance and system specifications of the TTU-WRS may be found in the literature (Muirhead *et al.* 2003; Jackson *et al.* 2004; Jackson & Morse 2005). The system was operated at an influent flow rate of 0.4 mL/min, and a recycle ratio of 10. The TTU-WRS treated the EPBW stream containing real urine rather than the synthetic urine solution used by the aerobic oxidation-nitrification reactor.

A common bench-scale membrane test is the batch internal recycle membrane test (BaIReMT) (DiGiano *et al.* 1999, 2000). In the BaIReMT test procedure, the waste concentrate and permeate streams are returned to the feed tank. The BaIReMT has the distinct advantage of requiring only 15 L of wastewater (DiGiano *et al.* 2000) for each experiment. A mass balance on the feed tank identified the loss of mass within the system. Several assumptions were

made. First, it was assumed that no biological or chemical reactions occurred within the system. Realistically, the bio-reactor system would need to be modified with a filtration unit to prevent the escape of bacteria from the system to downstream processes. Second, the permeate flux and solute rejection were constant over time. Membrane performance may be modeled using the boundary layer resistance (BLR) model (Mulder 1996; Kumar *et al.* 2006). The BLR model defines flux as a function of transmembrane pressure (ΔP), dynamic viscosity (μ) and total membrane resistance (R_{Tot}).

The SEPA CF II[®] cross-flow, cell membrane system was used to evaluate RO performance using flat sheet RO membranes with an active surface area of 140 cm² (AK RO membrane by GE Osmonics). The system contained an internal recycle loop circulating flow at 3.5 L/min. The recycle pump was a Hydra-Cell[®] diaphragm pump from Wanner[®] Engineering. The maximum inlet pressure for the recycle pump was 250 psi. To avoid exceeding the maximum inlet pressure, the recycle tank was placed upstream of the recycle pump. The addition of the recycle tank deviates from the BaIReMT proposed by DiGiano *et al.* (1999, 2000). Flow and pressure were controlled by a bypass valve downstream from the recycle pump. The bypass line leads directly to the recycle tank. At the concentrate exit, a needle valve refined the operating pressure. At the intersection of the recycle and waste concentrate lines, a needle valve allowed for adjustment of the waste concentrate flow rate and the system recovery.

The test procedure contained three steps: set the membrane, determine the membrane resistance, and assess pure water flux. To set the membrane, DDI water was processed and the only resistance in the BLR model was due to the membrane. The membrane was set when the permeate flux changed less than 4% over a 12 h period. The second step quantified the total resistance as well as solute rejection and flux decline. Fifteen liters of experimental water was processed by the RO system. After 250 h of operation, wastewater processing was stopped. The resistance determined from this step was the sum of the resistances from the membrane, concentration polarization, and fouling. The impact of biological pretreatment was assessed by comparing the resistance due to fouling for the different water types. The last step removed the reversible resistance.

Any remaining wastewater was drained from the system and DDI water was processed. After initial flushing of the system with 2 L of DDI water, the system processed DDI water at 30% recovery. The permeate flux was monitored until the recycle line reached a temperature of 45 °C. The resistance due to non-reversible fouling was determined by subtracting the membrane and reversible resistances from the total resistance.

Measured water quality parameters quantified performance of the bench-scale biological reactors and the RO. pH was measured using benchtop meters and liquid filled pH probes by Thermo Orion (Beverly, Massachusetts) and HACH (Loveland, Colorado). Prior to analysis, samples were filtered using 0.45 µm glass fiber filters. For DOC and TN, the filtered sample was spiked with concentrated hydrochloric acid to lower sample pH below 2 for preservation. DOC/TN analysis was performed using a Shimadzu TOC/TN analyzer. Ions were determined using ion chromatography (DX-600 system by Dionex Corporation (Sunnyvale, California)). An IonPac® AS14A column was used to separate the following anions: chloride, nitrite, and nitrate. Cation (ammonium, sodium) concentrations were also determined by ion chromatography (IonPac® CS16 column) after filtration.

The Shon *et al.* (2004) extraction technique was employed to determine carbon adsorption to the membrane surface. A membrane coupon was placed into a 0.1 N NaOH solution to extract the organic matter off the membrane, which was acidified and then analyzed for DOC. This method is limited because 100% extraction efficiency is not likely and the extraction method tends to under-predict carbon sorption.

Kumar *et al.* (2006) derived the gel layer model (Equation (1)) to predict increases in resistance and decreases in flux. J is the permeate flux, R_{Tot} is total resistance, t is time and A/B are lumped parameters. The lumped parameters A and B can be determined experimentally by plotting the change in resistance over change in time against permeate flux. The parameters A and B were determined by linear regression. Knowing parameters A and B , J may be substituted with P_{net}/R_{Tot} in Equation (1), where R_{Tot} is defined as the sum of membrane, concentration polarization, and fouling resistances, resulting in Equation (2). A fourth order Runge-Kutta solution was used to solve

for R_{Tot} (Chapra & Canale 1988).

$$\frac{dR_{Tot}}{dt} = AJ - B \quad (1)$$

$$\frac{dR_{Tot}}{dt} = A \frac{P_{net}}{R_{Tot}} - B \quad (2)$$

RESULTS AND DISCUSSION

The impact of biological treatment on the EPBW and the ersatz water quality is illustrated in Figure 1. As shown in Figure 1(a), the DOC and NH_4^+ -N in the EPBW was reduced approximately 90 and 43%, respectively, following biological treatment. NO_3 -N and NO_2 -N were produced during nitrification and the TTU-WRS effluent water typically contained approximately 170 mg/L NO_3^- -N, resulting in a pH drop from 8.5 to approximately 7. A statistical analysis was not performed to account for differences in data in Figure 1, because the two systems operate differently. As for the aerobic oxidation-nitrification system, the influent DOC, total dissolved solids (TDS), and TN were less concentrated (Figure 1(b)) than the EPBW influent water quality (Figure 1(a)). As such, the effluent of the aerobic oxidation-nitrification system was of a higher quality than the nitrification-denitrification system. Overall, the effluent of the aerobic oxidation-nitrification system contained 30 mg/L DOC, 200 mg/L TN, approximately 175 mg/L of NH_4^+ -N and 125 mg/L of NO_3^- -N.

A mass balance on the feed tank was employed to identify potential mass losses in the EPBW over time, which could impact the interpretation of experimental results. The mass balance results for TDS and DOC are presented (Figure 2). The lines in Figure 2 represent the values anticipated by the mass balance model, and the symbols represent the actual water quality data. The BaIReMT model over-predicts permeate TDS concentration while under-predicting brine concentration at the end of the experiment, which is related to the model's assumptions of constant solute rejection, constant permeate flux, and no reactions. The model also over-predicts the feed tank concentration, one possibility for this over-prediction is that it is a result of

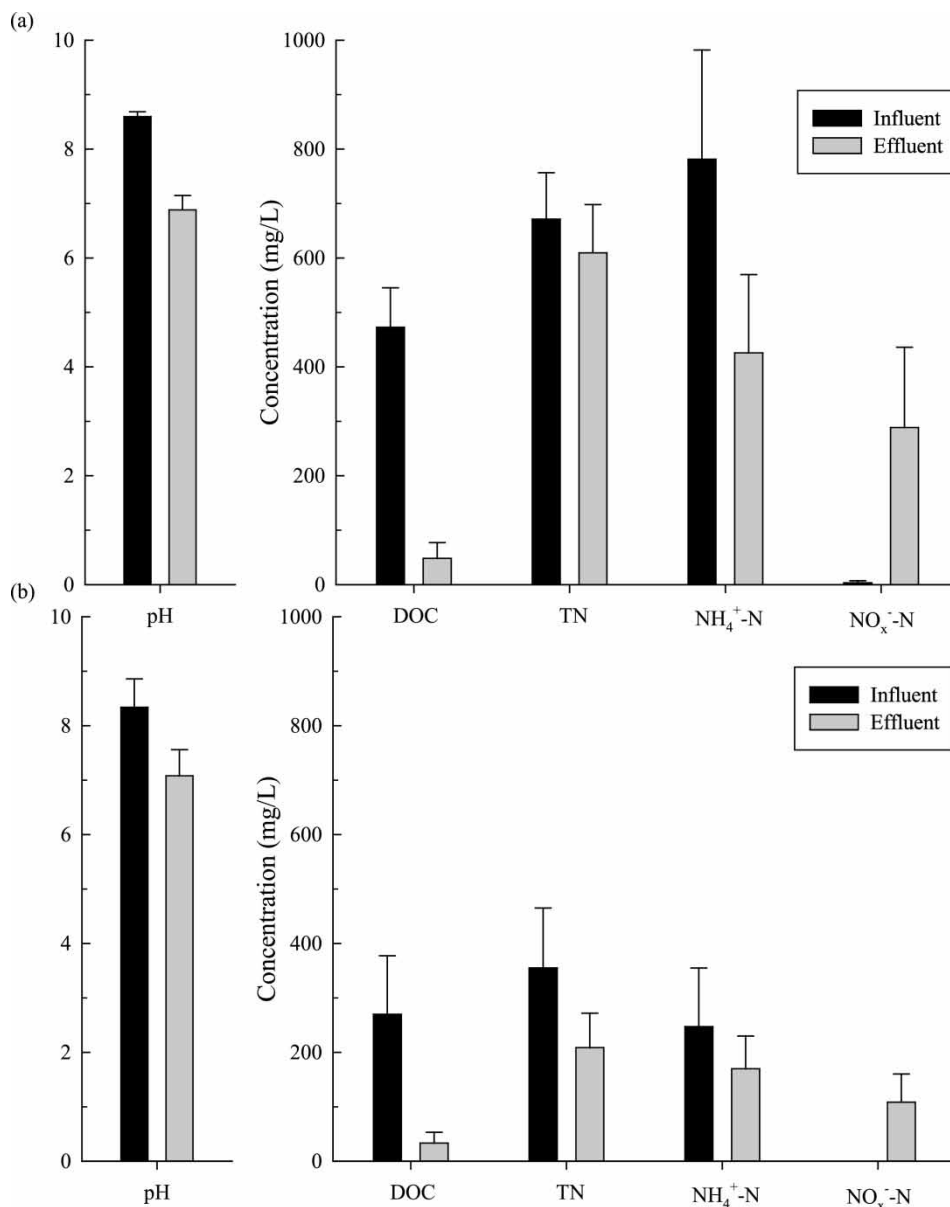


Figure 1 | Change in (a) EPBW ersatz water quality due to aerobic oxidation-nitrification system and (b) EPBW water quality due to denitrification-nitrification (TTU-WRS) system.

incomplete dispersion of the wastewater constituents in the feed reservoir.

After 100 h of operation, there was a steady loss of DOC from the system, likely a result of an organic matter cake layer formation on the surface of the membrane. However, a fraction of the DOC loss could be due to microbial oxidation over time or sorption to other parts of the experiment apparatus. For each of the four wastewater sources (untreated EPBW with real urine, treated EPBW

with real urine, untreated EPBW ersatz, and treated EPBW ersatz), the rate of DOC loss was calculated, which was assumed to equal the rate of DOC accumulation on the membrane surface. Both of the untreated water sources exhibit DOC accumulation at a rate between 0.017 and 0.021 mg/cm² h whereas the rate of accumulation for the treated wastewater sources ranged between 1.3×10^{-4} and 1.92×10^{-4} mg/cm² h. The higher rate of carbon adsorption for the untreated cases was reasonable as the untreated

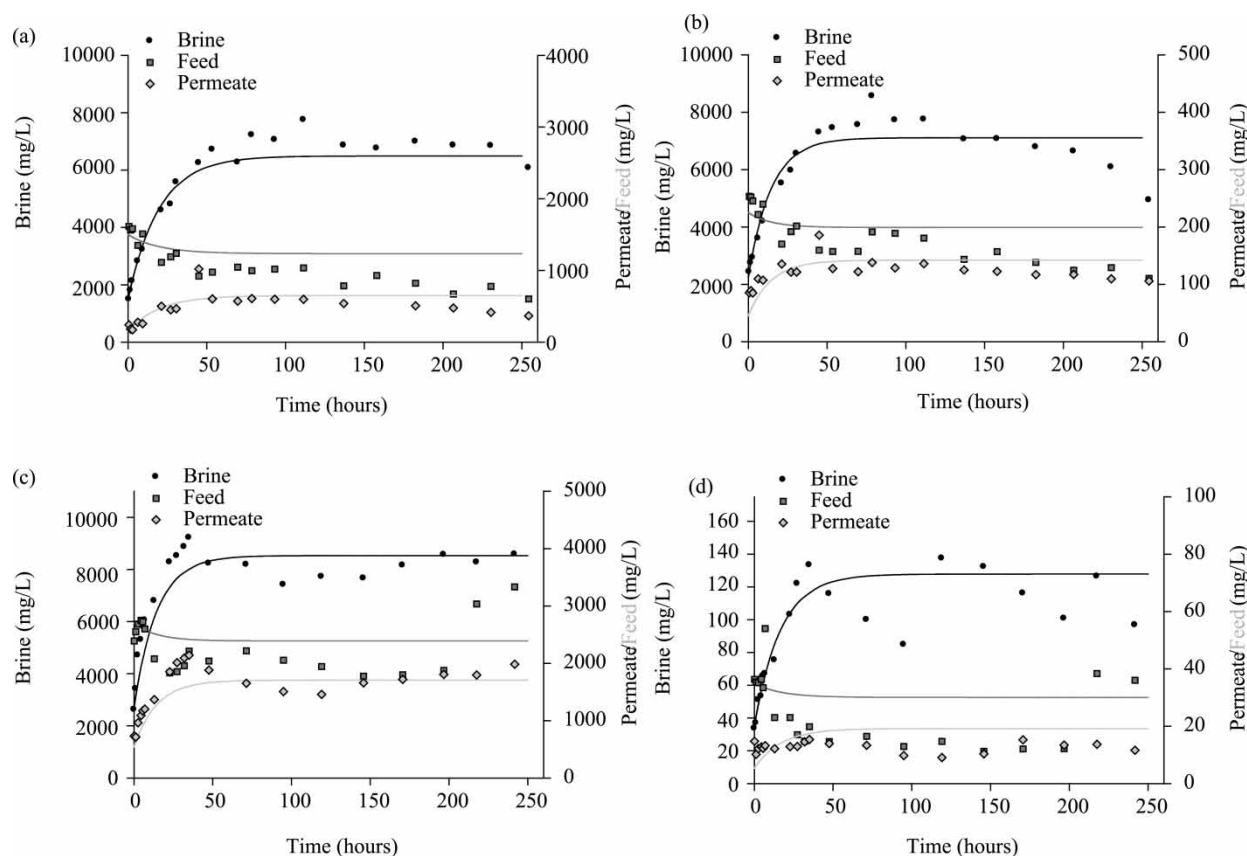


Figure 2 | Mass balance graphs for untreated EPBW (a) TDS and (b) DOC and untreated ersatz EPBW (c) TDS and (d) DOC.

DOC (250–600 mg/L) was typically an order of magnitude more than the treated DOC (<50 mg/L).

Figure 3 presents the results for the membrane carbon extraction method. The data indicates that the membrane used to treat the biologically treated effluent exhibited greater carbon extraction than untreated virgin membrane. The treated membranes exhibited a 40% increase in carbon extracted, which is inconsistent with the results of the mass balance method. Researchers believe the lower molecular weight (MW) material tends to foul by adsorption to the membrane and subsequently lowers extraction efficiency (Mulder 1996). Typically, biologically treated effluent exhibits a bimodal particle size distribution (PSD) where the majority of the refractory organic material has a MW greater than 10,000 Daltons (Barker & Stuckey 1999). The high MW material has a greater tendency to form a cake layer on the surface of the membrane, which is more easily extracted than the low MW material lodged within the membrane pores. This phenomenon explains why

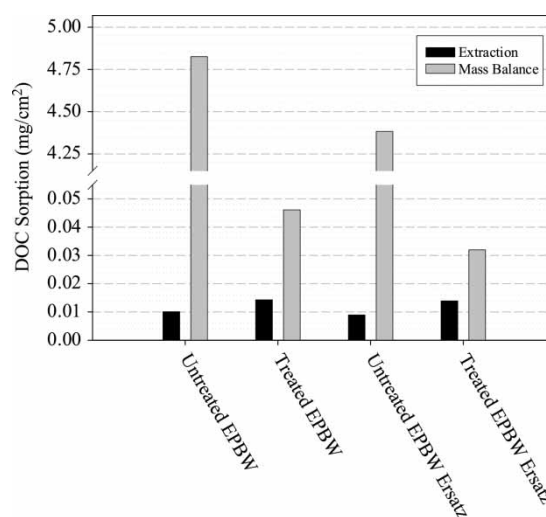


Figure 3 | Comparison of carbon sorption techniques.

membranes exposed to the biologically treated water sources extract more carbon and indicates that the untreated EPBW membrane was less receptive to cleaning than the

treated EPBW membrane, which was consistent with observed resistances and flux recovery obtained.

Upon comparing the two carbon sorption techniques, the mass balance method generates values greater than the extraction method. The results of the comparison were expected as the mass balance method tends to over-predict and the extraction method tends to under-predict. The results support the premise that biological oxidation of organic matter was occurring within the RO system. For untreated EPBW and ersatz, the majority of the DOC loss was due to microbial oxidation within the RO system rather than sorption to the membrane. The original assumption regarding microbial oxidation is not valid.

The mass balance for $\text{NH}_4^+\text{-N}$ is shown in Figure 4 and $\text{NH}_4^+\text{-N}$ decreased in the non-biologically treated brine. The average rate of ammonia loss was about 0.95 mg/L per h, which was likely due to volatilization as the brine pH was above 9.0. The fraction of $\text{NH}_4^+\text{-N}$ ion and NH_3 gas is highly dependent on the pH of solution (Tchobanoglous et al. 2003) and the pK value for the distribution of $\text{NH}_4^+\text{-N}$ and NH_3 gas is 9.25 (Tchobanoglous et al. 2003). As NH_3 gas dominates at a pH above 9.25, the loss of $\text{NH}_4^+\text{-N}$ from solution was likely due to volatilization and turbulence. In comparing $\text{NH}_4^+\text{-N}$ mass balance for untreated and treated EPBW the $\text{NH}_4^+\text{-N}$ brine concentration did not exhibit the same loss as the untreated EPBW. The average pH for the treated EPBW experiment was 6.5. As expected, the low pH prevented $\text{NH}_4^+\text{-N}$ volatilization. Also, the $\text{NH}_4^+\text{-N}$ concentration for the last two points increases due to an increase in mixing in the

feed tank. $\text{NH}_4^+\text{-N}$ represents 18% of TDS and the loss of $\text{NH}_4^+\text{-N}$ could account for the mass balance over-prediction of TDS.

In addition to monitoring the fate of water quality compounds during the experiment, the permeate flux was measured during each experiment (Figure 5). To address the variability from one membrane to the next, the permeate flux was normalized to the DDI water flux. The treated ersatz experiment only has one replicate. Generally, the flux was expected to maintain a downward trend as fouling perpetuates; however, the flux for the untreated EPBW reaches a minimum and then increases. The loss of $\text{NH}_4^+\text{-N}$ could explain this phenomenon. It is presumable that as $\text{NH}_4^+\text{-N}$ is lost from the system, the concentration polarization resistance decreases at a rate greater than the accumulation of fouling resistance; therefore, the flux increases. The same trend was not observed in the treated EPBW experiments, suggesting the loss of $\text{NH}_4^+\text{-N}$ reduces concentration polarization.

Figure 6 shows the observed resistances for the four water sources. The duration of each experiment was approximately 250 h. Notice that the membrane resistance remains fairly constant for the raw EPBW, treated EPBW, and raw ersatz experiments. The membrane resistance for the treated ersatz, is much lower (approximately 70%), which may be explained by the fact that this membrane came from a separate production lot. The reversible resistance for untreated ersatz is greater than that from the untreated EPBW, which is expected because the ersatz recipe contained greater DOC and TN than the urine

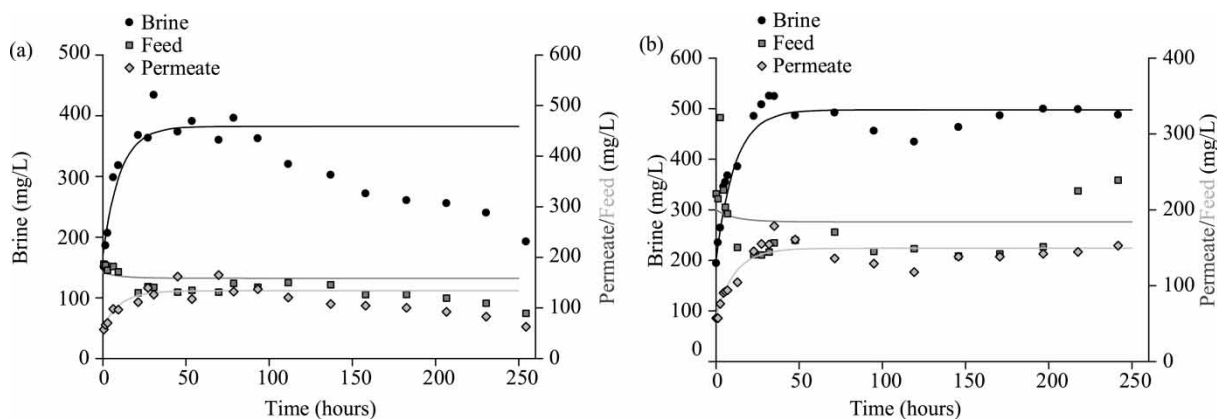


Figure 4 | $\text{NH}_4^+\text{-N}$ mass balance for untreated EPBW (a) and treated EPBW (b).

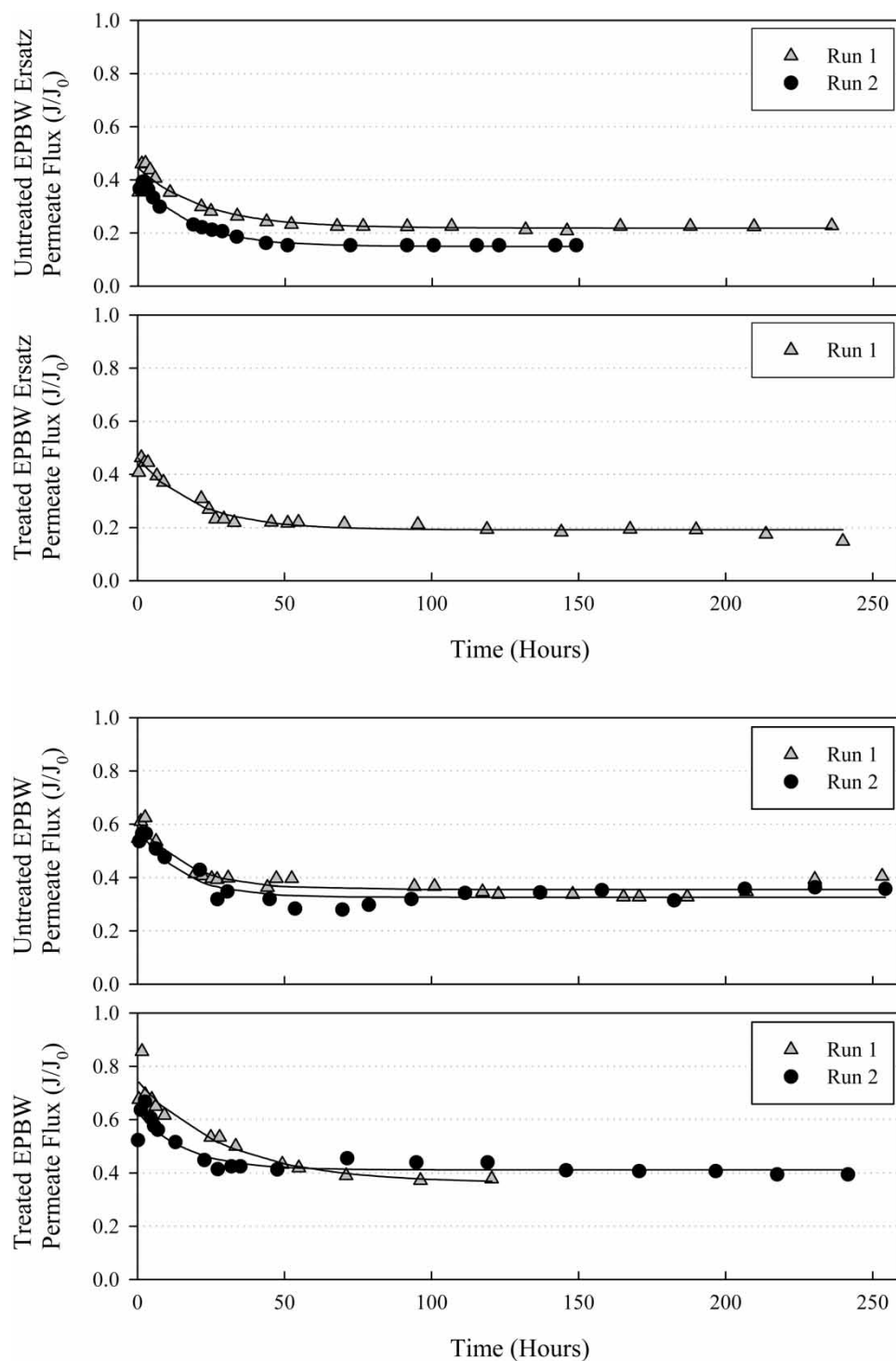


Figure 5 | Treated and untreated EPBW ersatz and EPBW permeate flux.

based EPBW. The DOC and TN concentrations for the untreated ersatz recipe are 500 and 800 mg/L, respectively. The DOC and TN concentrations for the untreated EPBW

are 250 and 325 mg/L, respectively. Also, the treated ersatz experiences a 20% drop in reversible resistance compared to untreated ersatz. The treated EPBW appears to

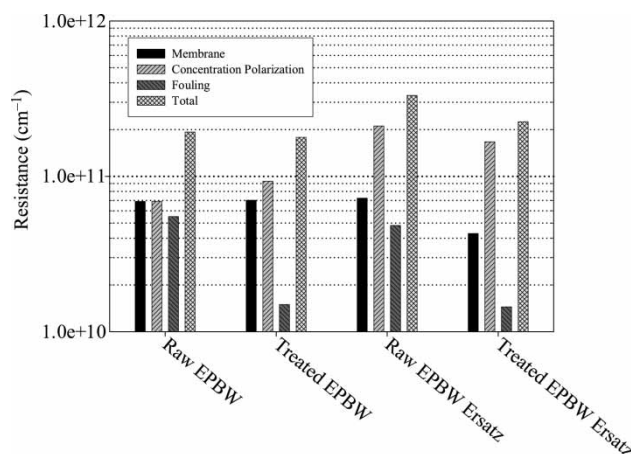


Figure 6 | Resistances.

have a 34% higher reversible resistance than raw EPBW. The increase in resistance seems counterintuitive considering the treated wastewater has lower DOC and TN concentrations. However, the observed reversible resistance

could be artificially low owing to the loss of ammonia via volatilization. For the untreated EPBW and the untreated ersatz, the biological treatment yields a 72 and 70% decrease in non-reversible fouling resistance, respectively. To determine if the degree of non-reversible fouling changes over the duration of the experiment, resistances were plotted for untreated ersatz after 150 and 250 h. The resultant non-reversible fouling resistance for 150 and 250 h was 2.7×10^{10} and 4.83×10^{10} N-s/m³, respectively. The additional 100 h increases non-reversible fouling resistance by 78% suggesting that non-reversible fouling occurred over the 100 h difference in experiment duration.

Figure 7 shows the regressions used to determine the lumped parameters A and B whereas Figure 8 shows the modeled change in resistance with change in time for treated and untreated ersatz. As expected, the untreated ersatz resistance converged to a higher value, and converges faster than the treated ersatz resistance. The R_{Tot} for treated

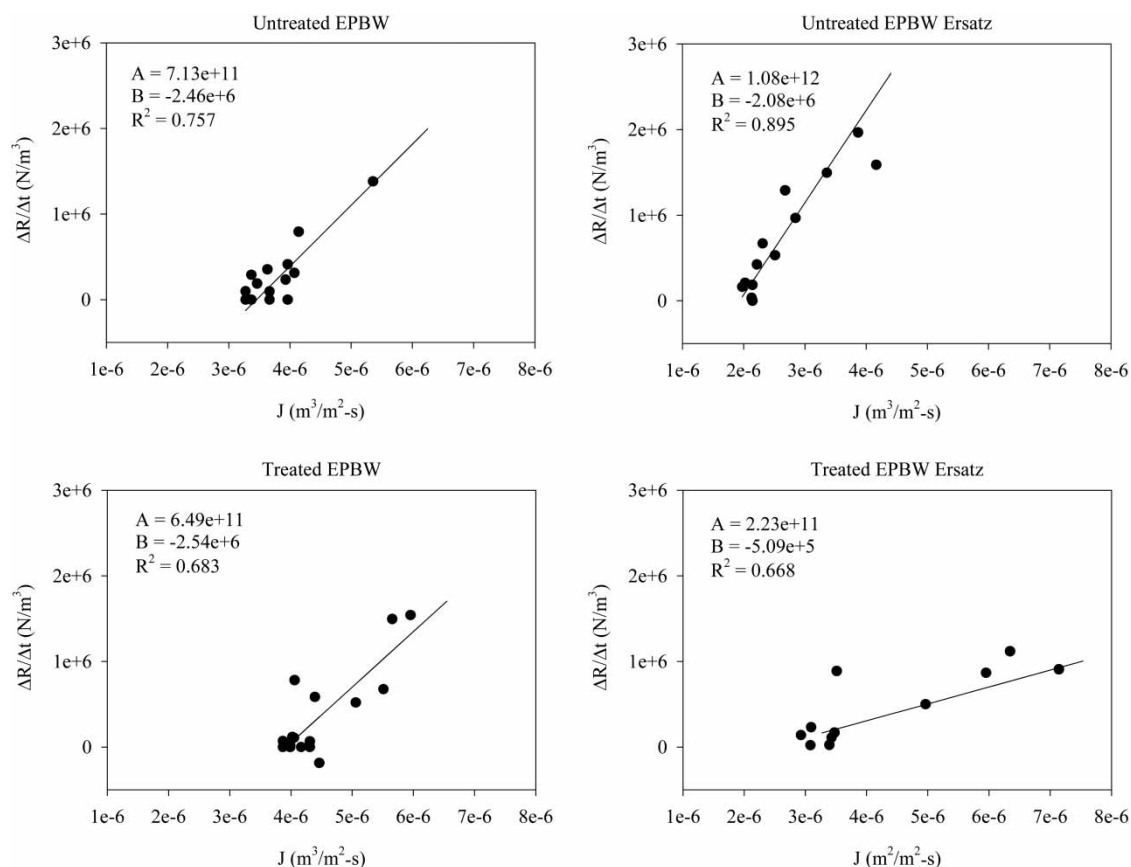


Figure 7 | Determination of parameters A and B for the gel layer model.

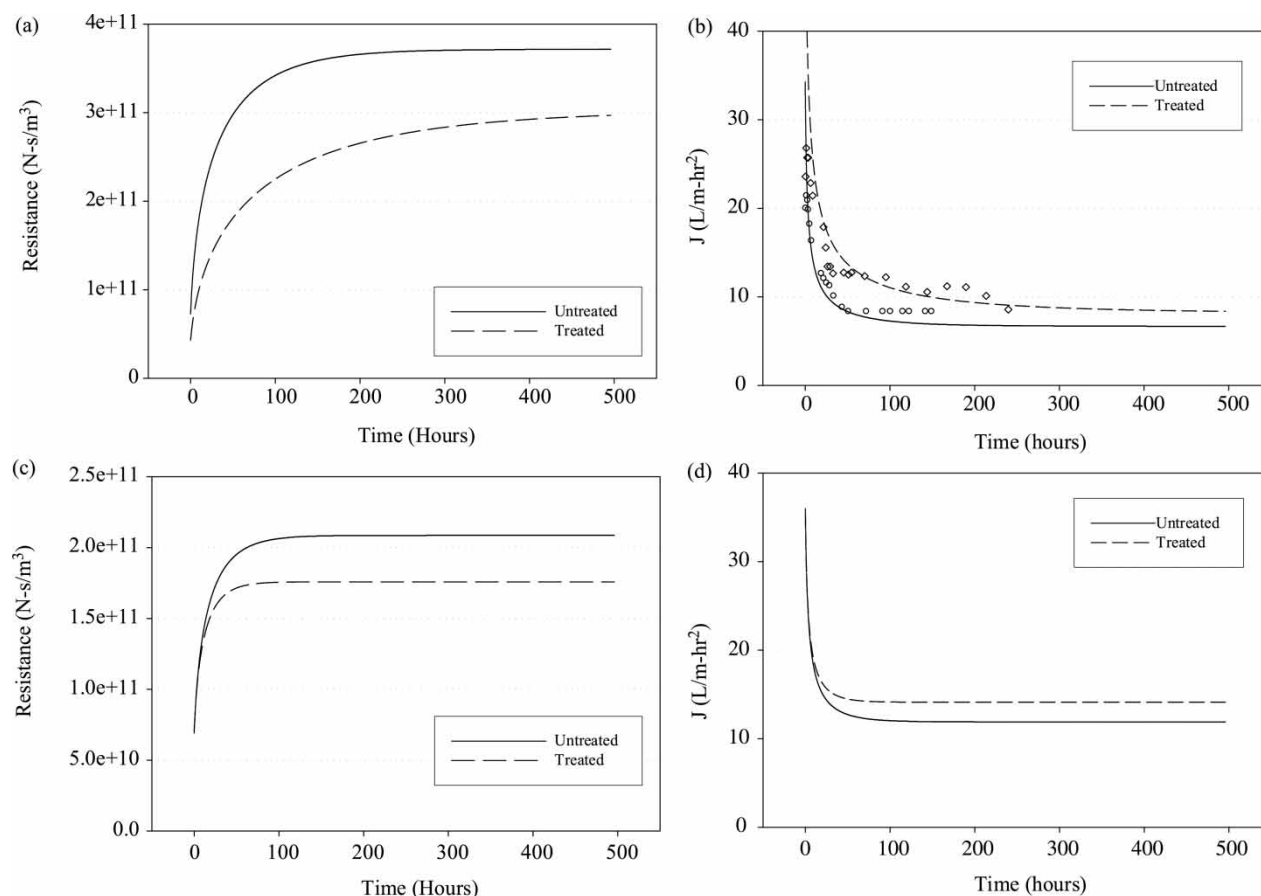


Figure 8 | Modeled EPBW ersatz resistance (a) and flux (b) and the modeled EPBW resistance (c) and flux (d).

ersatz converges to $2.97 \times 10^{11} \text{ N·s/m}^3$ which is 16.5% less than the R_{Tot} for raw ersatz. Additionally, R_{Tot} and permeate flux was plotted with time. The untreated ersatz permeate flux converges to $6.96 \text{ L/m}^2 \text{ h}$ while the treated ersatz permeate flux converges to $8.36 \text{ L/m}^2 \text{ h}$, which is 20% higher than the untreated ersatz permeate flux. The treated EPBW permeate flux ($14.12 \text{ L/m}^2 \text{ h}$) was about 13.6% higher than the untreated EPBW permeate flux ($12.43 \text{ L/m}^2 \text{ h}$). R_{Tot} was plotted against time for the treated and untreated EPBW water sources. Again, the treated EPBW R_{Tot} was less than the untreated EPBW R_{Tot} . The untreated EPBW R_{Tot} converges to $2.09 \times 10^{11} \text{ N·s/m}^3$ and the treated EPBW R_{Tot} converges to $1.76 \times 10^{11} \text{ N·s/m}^3$, which is 16% less than the untreated EPBW R_{Tot} . Results indicate that the higher solute concentration of the ersatz affected performance; however, biological pretreatment improved R_{Tot} and permeate flux. When using the gel layer model, it is

important to consider the limitations of using such a model. First, the gel layer model has been known to overestimate flux decline when colloidal material exists in the wastewater (Kumar *et al.* 2006). Second, the gel layer model predicts the flux to reach an asymptote after a period of time, which may not be the case for a fully functioning system as fouling resistance will increase over time. Perhaps these limitations offset one another as they have opposite impacts on the model.

Figure 9 shows the membrane rejection for both treated and untreated EPBW. As the water is not nitrified, nitrate and nitrite are not expected in the feed; thus, their rejection is not displayed for untreated EPBW. For each of the examined solutes, the rejection is greater in the treated EPBW. For the treated EPBW, the enhanced rejection could be a product of lower osmotic pressure. A high osmotic pressure gradient can cause water to back diffuse from the permeate

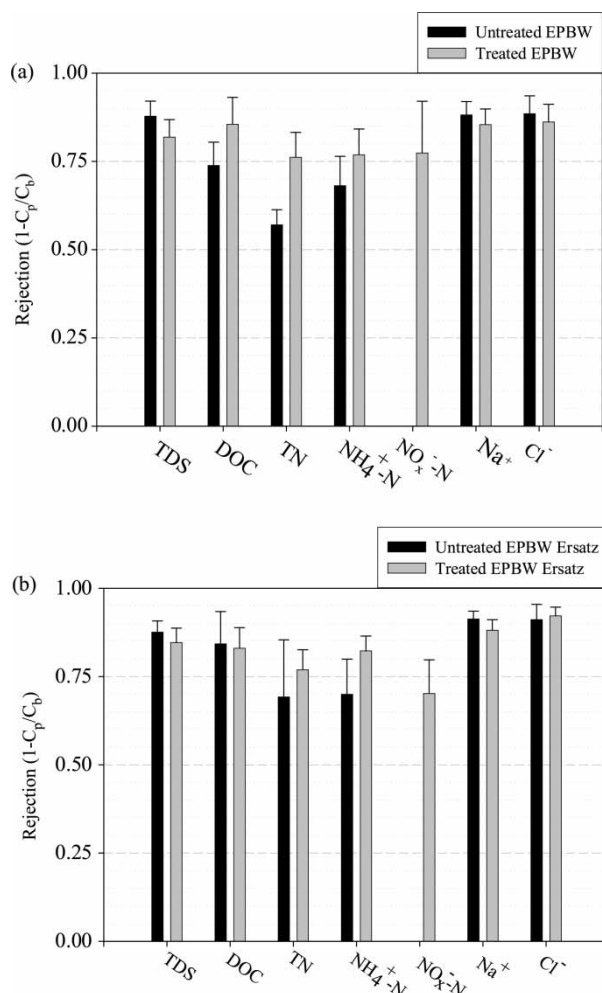


Figure 9 | EPBW and EPBW ersatz compound rejection.

to the brine, effectively increasing the concentration of the permeate. The lower concentration of the treated waste stream negates the back diffusion of water. The refractory DOC in treated EPBW likely has a greater MW, which also aids in rejection (Namkung & Rittmann 1986). Figure 9 shows the treated EPBW exhibits a 32% increase in DOC rejection compared to untreated DOC rejection. Biological pretreatment of EPBW increases TN rejection from 30% for the untreated case, to over 55%. The rejection of TN is greatly benefited by urea hydrolysis. As mentioned previously, urea passes easily through membranes. The urea in solution is biologically converted to NH₄⁺-N and NO_x⁻-N, which are more easily rejected than urea. The net result of biological treatment was an increase in TN rejection by 83%. Rejection was also evaluated for the treated and

untreated ersatz experiments. The NO_x⁻-N rejection is not depicted for the untreated case. Unlike the experiments performed on treated and untreated EPBW, several compounds experienced a decrease in rejection after treatment. Specifically, TDS, DOC, Na⁺, and Cl⁻ rejections were lower for treated ersatz than untreated ersatz. Alternatively, TN and NH₄⁺-N rejection was higher for the treated case.

CONCLUSION

To determine the degree to which the incorporation of biological treatment of graywater enhances RO performance, a series of bench-scale experiments were performed. A mass balance and observed solute concentrations helped to determine the fate of the solutes. For the case of the untreated EPBW and ersatz, microbial oxidation of DOC occurs within the RO system. This trend was not observed for the case of treated EPBW and ersatz. The mass balance indicated that NH₄⁺-N partitions into the cabin atmosphere for untreated EPBW. Because the pH of the untreated EPBW lies near the pK value for NH₃, the nitrogen readily partitions into the air phase. Simply comparing the permeate flux for treated EPBW, untreated EPBW, treated ersatz, and untreated ersatz indicates that biological pretreatment does little to improve the permeate flux in the RO system. However, the permeate flux from untreated EPBW was artificially high owing to mass loss. Results of the gel layer model indicate that biological treatment improves permeate flux more than the observed permeate flux indicates. Observed resistances suggest biological pretreatment alleviates the degree of fouling.

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