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Evaluation of a gravity flow membrane bioreactor for treating municipal wastewater

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Abstract

The biomass concentrator reactor (BCR), a gravity-flow membrane bioreactor (MBR) design, was evaluated for use in treating a municipal wastewater stream. The BCR operates with less than 2.5 cm of pressure head and uses a 3–4 mm thick tortuous path membrane with pore size ranging from 18 – 28 μm to achieve solids separation. A two-stage, aerobic/anoxic reactor was evaluated for removal of chemical oxygen demand (COD), ammonia, total nitrogen, and solids separation. The reactor was fed 72 L/day, a hydraulic retention time of 9.3 hours, and had a solids retention time of 20 days. The influent COD was reduced by 93% while influent ammonia was reduced below 0.1 mg/L and total nitrogen was reduced by 53.7%. A lack of readily biodegradable COD limited denitrification and thus total nitrogen removal. The reactor solids were retained completely in the reactor by the membrane for the duration of testing.

Keywords

Membrane bioreactor; Nitrification; Denitrification; Wastewater treatment; Gravity flow; Biomass concentrator reactor

1. Introduction

Traditional wastewater treatment systems are dominated by large, centralized facilities that remove organic matter and nutrients from water. Design engineers size unit processes based on estimated parameters such as flow rates and mass loadings. Once construction is completed, the resulting infrastructure is often too rigid to accommodate changing conditions that may occur in the served area (Tjandraatmadja et al., 2005). Hence, demands

for a higher quality effluent and the need for a more flexible, controllable treatment process have led to the rise of membrane bioreactors (MBRs).

MBRs were developed in the 1960s, but have become very popular in recent years (Mallevialle et al., 1996; Le-Clech et al., 2006). These reactors have several advantages over conventional designs that make them desirable: a reduced footprint, improved effluent quality, total control of biomass wasting, flexibility to operate at a wide range of variables, and flexibility in scale (Cicek et al., 2001; Le-Clech 2010). Hence, MBRs have become preferred in areas where water and land are scarce and where pollutant removal and water recovery/reuse are important. General acceptance of MBRs has been hindered by several disadvantages such as higher capital cost as compared with traditional treatment because of the high cost of the membrane units; higher energy consumption owing to the pumping requirements through the low porosity membranes and extra aeration; membrane clogging that requires wastewater pretreatment; and membrane chemical cleaning or replacement because of fouling (Cicek et al., 2001; Metcalf and Eddy, 2003; Le-Clech et al., 2006; Le-Clech, 2010). These reasons have limited the implementation of MBRs to instances where the need to recover the water outweighs the energy and monetary costs.

An innovative MBR design, the biomass concentrator reactor (BCR), was developed to significantly reduce the energy and cost requirements of an MBR system, while still maintaining complete biomass retention. Co-developed and patented by the University of Cincinnati and the U. S. Environmental Protection Agency (Patent No. 6821425 issued November 23, 2004), the BCR has a 3–4 mm thick membrane with pore size ranging from 18 – 28 μm constructed from packed polyethylene beads. Flow through the membrane is driven entirely by gravity with a head differential of less than 2.5 cm between the retentate and permeate sides of the membrane. The large pore size of the membrane along with the tortuous, depth filtration mechanism effectively retains the biomass and significantly reduces the pressure required for water to percolate.

The BCR was first developed to treat groundwater contaminated with gasoline (Zein et al., 2004). Several studies confirmed its effectiveness in removing hydrocarbons and oxygenates such as methyl-t-butyl ether, while completely retaining biomass under different conditions and loadings (Zein et al., 2004; Zein et al., 2006; Capodaglio et al., 2010; Medella et al., 2011). Scott et al. (2013) used the BCR system to treat municipal wastewater. Side-by-side evaluation of a nitrifying system and a nitrifying-denitrifying hybrid system using a synthetic wastewater resulted in a chemical oxygen demand (COD) reduction of over 90% and almost complete nitrification in both systems. In addition, the BCR was able to reduce the total nitrogen in the hybrid system by 67% (Scott et al., 2013). The goal of the research reported in this paper is to replicate their results with the hybrid system using actual municipal wastewater in order to assess the impact of inert suspended solids and non-biodegradable organic matter on membrane performance.

2. Materials and Methods

2.1 Reactor Design

The BCR was set up as shown in Supplementary Figure 1 and its detailed description can be found in Scott et al. (2013), as the system is almost identical to the hybrid unit therein reported. The membrane was a commercially available filter (Porex Corp., Atlanta, GA), which consisted of a pleated, cylindrical module, approximately 15.5 cm in diameter and 25.1 cm in length with a surface area of approximately 0.62 m². The reactor volume (30 L) was equally split between the aerobic and anoxic sections with approximately 2 L occupied by the membrane and the permeate collection section within the membrane module.

Wastewater was continuously fed into the anoxic compartment to facilitate denitrification, while a second pump kept a fixed recycle ratio by returning mixed liquor from the anoxic compartment back up to the aerobic section where the membrane separated the solids from the water. The retained biomass was funneled toward the anoxic stage at a rate equal to the recycle ratio minus the feed flow rate of the influent wastewater. Permeate exited the reactor through a side port located in a vertical tube that allowed for water level control inside the reactor.

The BCR was operated at a solids retention time (SRT) of 20 days (i.e., 1/20th of the volume of the reactor was wasted each day). This waste volume was taken equally from both sections. Wastewater was pumped into the reactor at a flow rate of 72 L/d. The recycle ratio was initially set at 3 times the influent flow rate (3Q) and was increased to 5 times the influent flow rate (5Q) on day 247.

2.2 Wastewater Pumping System

The wastewater was taken from a 48" diameter sewer main running under the University of Cincinnati west campus in Cincinnati, Ohio, USA. Upstream of the connection were several restaurants, dorms, classroom buildings, and office buildings as well as runoff collection drains. All of these sources were assumed to have contributed to the wastewater stream, though the exact contributions of each are unknown and most likely varied widely over time.

The pumping system consisted of a large storage tank, a primary settling tank, and several pumps. Wastewater was pumped from the sewer main into the storage tank and, subsequently, up to the lab and into the settling tank. Within the settling tank, the wastewater was allowed to separate, with the solids and grit settling to the bottom and the fats and oils floating to the surface. A port located just below the middle of the tank was used to supply the wastewater to the reactor.

2.3 Reactor Startup and Operation

The BCR was seeded with biomass from the aeration tank of a local municipal wastewater treatment plant. Approximately 4 L of mixed liquor was added to the reactor along with deionized tap water. The reactor start-up period lasted 170 days. The time needed to attain steady state was lengthy due to operational problems and perceived limited denitrification in the anoxic zone. Initially, we attributed this limited denitrification to either poor mixing in

the anoxic section or short-circuiting between the aerobic and anoxic compartments. We later realized that the poor nitrogen removal was due to variations in feed Chemical Oxygen Demand (COD) strength and periods where the feed biodegradable COD was insufficient to achieve the desired level of denitrification (see Discussion section). Since fortifying the wastewater would defeat the goals of this work, we did not alter its quality. After addressing these problems, we started the 137-d experimental period.

The membrane was cleaned four times per week using a jet of recycled mixed liquor from the oxic section of the reactor. This process helped prevent buildup in-between the pleats of the membrane where the scouring intensity of the air bubbles was insufficient to keep the surface clean. The system was initially run without a buffer; however, the pH in the system varied significantly and was often below 7, which is not optimum for nitrification. Consequently, on day 136, a Na_2CO_3 solution (16 mg/L) was pumped into the oxic section at a flow of $\sim 1 \text{ L/d}$ to maintain a pH between 7.4 and 8. Alkalinity was adjusted as needed thereafter. The system was operated until the head differential across the membrane exceeded 2.5 cm, above which the membrane was replaced. The fouled module was chemically cleaned, first soaking for 24 hours in 10% nitric acid solution followed by a 10% Clorox solution.

2.4 Sampling Collection

During the start-up period grab samples were collected twice per week from the influent, effluent, and oxic and anoxic mixed liquors. On day 102, a composite sampler was put in place so that influent and effluent were monitored over 24-hour periods by collecting 20 mL of each stream every 15 minutes. The apparatus was placed in a refrigerator at 4°C to limit further biological activity. Every 24 hours, the sampling bottles were replaced and kept at 4°C until analysis could be performed. Mixed liquor samples were grabbed from the 2 BCR compartments and analyzed along with the stored composite samples 3 times per week.

2.5 Analytical Methods

Each composite influent and effluent sample was split into aliquots to measure the concentrations of COD, ammonia, nitrate, nitrite, and total Kjeldahl nitrogen (TKN). Influent and both mixed liquors were also analyzed for total suspended solids (TSS) and volatile suspended solids (VSS). All samples were analyzed in triplicate following Standard Methods (APHA, 1998). These methods were COD method 8000 by Hach, ammonia method 4500 (Orion 9512HPBNWP Ammonia Electrode), nitrate and nitrite method 4110B (Dionex LC20 Ion Exchange Chromatograph), TKN method 8075 by Hach, and TSS/VSS method 2540D (Hach, 1992). The pH (Oakton WD-35801-00 pH Electrode) of the aerobic and anoxic mixed liquors was measured daily.

3. Results and Discussion

During the startup period, the wastewater was high in ammonia, between 20 and 40 mg/L as $\text{NH}_3\text{-N}$, and had the expected amount of COD ranging from 150 to 400 mg/L (Sup. Figs. 2–5). However, less than half of that COD was readily biodegradable, which is not entirely unexpected when working with real wastewater (Grady et al., 2011). The low BOD plus the

higher than expected ammonia meant that denitrification was carbon-limited and almost never reached the expected removal. Confirmation of this carbon limitation came when a large, one-day spike in the COD on day 204 (up to ~700 mg/L) reduced the effluent nitrate concentration to the approximately the expected value.

3.1 Chemical Oxygen Demand

COD was measured in the influent and effluent (see Fig. 1). The influent COD was generally low with a few high spikes when compared to a typical municipal wastewater. When compared to the data collected during startup, the COD was much lower because of a drastic change in wastewater production: the end of the spring school term at the University of Cincinnati occurred 2 days before the experimental period began and the autumn term did not begin until day 290. Activity on campus dropped significantly during summer, highlighted by the vacancy of most of the dorms and apartments that would have contributed to the wastewater stream. The restaurants and classrooms also generated significantly lower usage, leaving the majority of the flow to come from the office buildings and rainfall runoff. Erratic spikes in the COD are understandable given that the wastewater comes from a small area and could be influenced very easily by a number of different factors. On day 290, the new school year began and an increasing trend in the influent COD can be seen in the days just prior to classes starting.

Throughout the entire period of observation, the effluent COD was consistently below 20 mg/L with an average of 7.3 ± 1.5 mg/L, regardless of the corresponding influent concentration. The consistency of this removal highlights the flexibility of the BCR in handling a varying-strength waste stream. Other MBRs have shown comparable results when dealing with variations in the waste stream. Using a similar, low strength wastewater, Ueda and Hata (1999) reported achievement of COD removal down to 3.7 mg/L, while Rosenberger et al. (2002) reported a COD reduction from a high strength wastewater to below 40 mg/L. In both cases, influent fluctuations had little impact on the effluent quality.

The average COD removal rate was approximately 93%. This rate is consistent with the results of Scott et al. (2013) and other MBR systems, which range from 80 to 99% (Ueda and Hata, 1999; Rosenberger et al., 2002; Kimura et al., 2008). The switch to real waste from the synthetic feed used in the previous BCR research was expected to have caused a decrease in the removal efficiency because of potentially non-degradable components in the wastewater. The high removal efficiency indicated that either the wastewater did not contain large quantities of non-degradable COD or that the BCR was effective in retaining this fraction of the COD. MBRs have been shown to be effective in retaining non-degradable or slowly degrading COD, allowing the increased diversity of the culture in the reactor to eventually remove it (Rosenberger et al., 2002). Two minor increases in effluent COD were observed on days 190 and 204. The increase on day 190 was the result of contamination in the effluent line, which was purged after it was discovered. This buildup only occurred in the tubing used in the composite sampler and was likely due to a combination of contamination of the tubing prior to installation, the large surface area of the tubing, and the reduced flow of the composite sampler. Contamination of a membrane system is common anytime nutrients and a surface for attachment are available (Flemming, 1997; Matin et al., 2011).

The contamination was not observed prior to the composite sampler installation and was not reported in any of the previous BCR studies. As a precaution, the tubing was flushed every 3–4 weeks afterward. The increase on day 204 was likely due to the large spike in the influent COD coupled with the failure of the membrane.

3.2 Nitrogen

Several forms of nitrogen were monitored in the influent and effluent: ammonia, nitrate and nitrite, and TKN. The aerobic section was intended for nitrification, while denitrification took place in the anoxic stage. The $\text{NH}_3\text{-N}$ results are shown in Fig. 2a. The influent $\text{NH}_3\text{-N}$ fluctuated between 5 and 60 mg/L with an average of 20.8 ± 9.6 mg/L. This fluctuation did not correlate to changes in the influent COD, but the ammonia did follow the same pattern as the COD in relation to the changes in the wastewater source. During the period of observation, the effluent $\text{NH}_3\text{-N}$ concentration did not exceed 0.6 mg/L and was almost always below 0.1 mg/L. Furthermore, variations in influent $\text{NH}_3\text{-N}$ concentration had no apparent effect on the effluent concentration, even when the influent surged to 60 mg/L. Influent and effluent TKN concentrations averaged 25.4 ± 8.9 and 1.2 ± 0.4 mg/L, respectively, which resulted in an overall TKN removal efficiency of 95% (Fig. 2b). Ammonia represented 80% of the total TKN load.

The effluent concentration of $\text{NO}_3\text{-N}$ was 13.7 ± 7.0 mg/L while the $\text{NO}_2\text{-N}$ was consistently below 0.1 mg/L. The $\text{NO}_3\text{-N}$ removal was limited compared to the removal effectiveness observed for ammonia and TKN. As previously discussed, the combination of low influent COD, which had a small fraction of readily biodegradable carbon, and high influent $\text{NH}_3\text{-N}$ led to less than expected denitrification. Figure 3 presents three data sets: effluent $\text{NO}_3\text{-N}$, theoretical maximum effluent $\text{NO}_3\text{-N}$ based on influent TKN and no denitrification, and theoretical effluent $\text{NO}_3\text{-N}$ assuming no carbon limitations. The theoretical concentrations were determined using Eq. 1 and 2 (Metcalf and Eddy, 2003):

$$\text{NO}_x = \text{TKN}_{\text{Inf}} - \text{NH}_{3,\text{eff}} - 0.035 \times \frac{P_x}{Q} \quad (1)$$

$$\text{IR} = \frac{\text{NO}_x}{\text{NH}_{3,\text{eff}}} - 1 \quad (2)$$

where all the forms of nitrogen were expressed as N; P_x , Q , and IR represent the daily wasted biomass, daily flow rate, and internal recycle ratio, respectively; and 0.035 g N/g VSS is substituted for 0.12 g N/g VSS based on TKN digestion of the biomass. These calculated concentrations represent upper and lower boundaries for $\text{NO}_3\text{-N}$, and, as shown in the figure, measured values remained between the two margins, being closer to the upper, or nitrification only, limit. Observed data are below the nitrification only values, which points to denitrification limited by the readily biodegradable COD. We compared $\text{NO}_3\text{-N}$ removed to influent COD in Fig. 4; clearly surges in COD led to higher $\text{NO}_3\text{-N}$ removal. While other factors may affect $\text{NO}_3\text{-N}$ disappearance, this graph provides reasonable evidence that the

COD heavily influenced nitrate metabolism and was likely the limiting factor. The recycle ratio between the anoxic and aerobic zones was changed on day 247 from 3Q to 5Q. The intention was to evaluate whether nitrogen removal affected the BCR performance. Because of the limited COD concentrations, reactor hydraulics did not affect denitrification.

The total nitrogen removed was calculated using Eq. 3:

$$\%N_{removed} = \frac{(TKN_{inf} - NH_{3,eff} - NO_{3,eff} - NO_{2,eff})}{TKN_{inf}} \times 100 \quad (3)$$

where all the forms of nitrogen were expressed as N. The average removal was $46.3 \pm 17.1\%$, ranging from 10 to 90%. The wide range and high standard deviation are consequences of the low COD and fluctuating influent concentrations. Such nitrogen elimination is low for a nitrifying/denitrifying process and is more in line with a purely nitrifying system (Scott et al., 2013; Kimura et al., 2008).

3.3 Total and Volatile Suspended Solids

The TSS and VSS were measured in both the oxic and anoxic sections as well as in the influent. Influent solids were monitored to verify that the settling/degritting tank was working properly and that large amounts of non-biodegradable material were not fed into the system. The TSS concentration from both sections and the total biomass in the reactor are shown in Fig. 5. As expected, solids followed the COD trend, since biomass depends on organic matter to grow. The figure clearly indicates a drop in TSS and total biomass during the low COD period, with a recovery exhibited after the COD started to increase again around day 285. The VSS was 77% of the TSS, which is consistent with reported values for systems operated on municipal wastewater (Metcalf and Eddy, 2003). Influent TSS was only about 100 mg/L with 60% being VSS, which suggests good particle separation.

The aerobic and anoxic sections of the BCR had similar concentrations of TSS during most of the period of study. Large differences were observed only at the beginning and end of the run. Such differences can be attributed to changes in the influent COD concentration during the study. High COD supported a greater amount of denitrification, which results in a slower growth rate compared to aerobic COD utilization, but also provided more COD for aerobic utilization (Metcalf and Eddy, 2003; Grady et al., 2011). Hence, solids rose in the aerobic section compared to the anoxic section. With low influent COD, however, the majority of the COD was used for denitrification, limiting any aerobic COD utilization. As a result, the growth rates converged and the two sections had similar TSS.

3.4 Observed Yield

The observed biomass yield for the experimental period is displayed in Fig. 6. The yield is typically determined based on the cumulative COD removed and cumulative VSS wasted, but this method assumes the reactor is at steady-state. Because the influent COD fluctuated significantly, the VSS never reached steady-state and the fluctuations in VSS needed to be added into the calculations to determine the yield more accurately. The VSS difference, from

the figure, was defined as the VSS wasted intentionally on a given day plus the change in VSS that occurred in the 24 hours since the previous wasting (wasting was conducted daily). The graph reveals three distinct rates that correspond to the periods of high and low COD described earlier. The yields were 0.22, 0.15, and 0.24 g VSS/g COD removed for the three periods, which are all at the lower end for a nitrification/denitrification process (Metcalf and Eddy, 2003; Grady et al., 2011). These results are likely due to a combination of the high SRT and the carbon-limited denitrification process. Scott et al. (2013) also observed a decrease in yield as the SRT increased when comparing nitrifying and hybrid processes. The periods prior to 1,750 g of COD removed and after 2200 g of COD removed contained higher amounts of COD, allowing for growth to occur in both aerobic and denitrifying conditions. During the low COD period (between 1750 and 2200 g of COD eliminated), the COD was utilized mostly under denitrifying conditions, resulting in a lower yield.

3.5 Membrane Performance

The pressure head, the only mechanism for flow through the membrane, was recorded daily to monitor the flow and assess the condition of the membrane. The pressure across the membrane was consistent throughout the testing phase, maintaining a level between 0.5 and 1 cm of pressure head. The membrane clogged once during testing on day 204, having operated for 51 days. The clogging corresponds with the large spike in the influent COD concentration, indicating the large change in concentration led to the failure. Also, the pressure head increased dramatically over a 24-hour period, from 1 cm to 6.5 cm, rather than a slow rise over several days, further supporting the reason for failure. After replacing with another module, the reactor was operated for an additional 117 days before another clogging failure, which occurred on day 318, shortly after the end of the data collection period. This second membrane failure happened more gradually, with an increase in the pressure head of ~0.5 cm per day over 9 days. The length of time between failures demonstrates that the design can perform well for extended periods of time without the need for backflushing or chemical cleaning. However, the two failures were preceded by higher and more variable influent COD levels, indicating that the system may be adversely affected by either high influent levels or high variability or a combination of both. It should be noted that throughout our studies with the BCR that extended over four years, the same membranes were used and regeneration consisted of chemical cleaning with 10% bleach and 10% nitric acid.

4. Conclusion

The BCR was assessed for treating municipal wastewater and was shown to effect excellent removal of carbon and ammonia-nitrogen. Operated by gravity flow at less than 2.5 cm of head, the large pore-size, depth filtration membrane completely separated the solids from the treated water. On average, COD and $\text{NH}_3\text{-N}$ were reduced by 93 and 99%, respectively. Total nitrogen removal was not as effective, averaging approximately 46%, but the carbon content of the influent wastewater limited the system. The results exceeded conventional activated sludge treatment systems and compared favorably with other MBR systems. Overall, the BCR proved flexible, capable of handling large fluctuations in the wastewater stream without compromising effluent quality.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Figure 1.
Influent and effluent COD.

Figure 2.
Influent and Effluent $\text{NH}_3\text{-N}$ (a) and TKN(b).

Figure 3.
Comparison of the measured effluent $\text{NO}_3\text{-N}$ to the theoretical effluent $\text{NO}_3\text{-N}$ and theoretical $\text{NO}_3\text{-N}$ produced.

Figure 4.
Dependence of nitrate removal on influent COD.

Figure 5.

Total suspended solids of the aerobic and anoxic mixed liquor and the influent (a) and total biomass in the reactor (b).

Figure 6.
Biomass yield during the experimental period.