Startup of a Partial Nitritation-Anammox MBBR and the Implementation of pH-Based Aeration Control

Stephanie Klaus[1,2*], Rick Baumler[2], Bob Rutherford[2], Glenn Thesing[3], Hong Zhao[3], Charles Bott[2]

ABSTRACT: The single-stage deammonification moving bed biofilm reactor (MBBR) is a process for treating high strength nitrogen waste streams. In this process, partial nitritation and anaerobic ammonia oxidation (anammox) occur simultaneously within a biofilm attached to plastic carriers. An existing tank at the James River Treatment Plant (76 ML/d) in Newport News, Virginia was modified to install a sidestream deammonification MBBR process. This was the second sidestream deammonification process in North America and the first MBBR type installation. After 4 months the process achieved greater than 85% ammonia removal at the design loading rate of 2.4 g NH_4^+/m^2·d (256 kg NH_4^+/d) signaling the end of startup. Based on observations during startup and process optimization phases, a novel pH-based control system was developed that maximizes ammonium removal and results in stable aeration and effluent alkalinity. *Water Environ. Res.*, **89**, 500 (2017).

KEYWORDS: deammonification, anammox, sidestream treatment, MBBR.

doi:10.2175/106143017X14902968254476

Introduction

Centrate from dewatered anaerobically digested solids can comprise 15 to 25% of the total incoming nitrogen load for a water resource recovery facility, but only represents about 1% of the total incoming flow. By treating the centrate in a sidestream system, the facility can reduce the nitrogen load on the mainstream process and, by doing so, provide more cost-effective and more efficient overall nitrogen removal (Jetten et al., 2001). The combination of partial nitritation and anaerobic ammonia oxidation (anammox), commonly known as deammonification, is an economical option for sidestream treatment because of decreased aeration energy requirements, no required external carbon, and decreased sludge production over traditional nitrification/denitrification (Ahn, 2006).

In the first step of deammonification, aerobic ammonium oxidizing bacteria (AOB) convert approximately 57% of the incoming ammonia to nitrite according to eq 1 (Grady et al., 2011).

$$NH_4^+ + 2.457O_2 + 6.716HCO_3^- \rightarrow 0.114C_5H_7O_2N + 2.509NO_2^- + 1.036H_2O + 6.513H_2CO_3 \tag{1}$$

In the second step, anaerobic ammonium oxidizing bacteria (AMX) convert the remaining ammonium and nitrite to nitrogen gas and a small amount of nitrate according to eq 2 (Strous et al., 1998).

$$NH_4^+ + 1.32NO_2^- + 0.066HCO_3^- + 0.13H^+ \rightarrow 0.26NO_3^- + 1.02N_2 + 0.066CH_2O_{0.5}N_{0.15} + 2.03H_2O \tag{2}$$

The deammonification reaction requires a net consumption of alkalinity (inorganic carbon). There are two components to the inorganic carbon (IC) demand for deammonification: production/consumption of hydrogen ions by AOB/AMX and incorporation of IC into the biomass of AOB and AMX. pH in a sidestream deammonification reactor is mainly governed by alkalinity consumption by AOB, which is a function of aeration intensity or aerobic fraction. Another factor influencing pH is CO_2 stripping due to aeration. Ammonium oxidizing bacteria consume alkalinity, while AMX produce a small amount for a net consumption of approximately 4.0 g $CaCO_3$/g NH_4^+ removed (theoretical according to stoichiometry of eqs 1 and 2). Nitrite oxidizing bacteria (NOB) (if present) do not significantly contribute to alkalinity requirements. Centrate/filtrate has a theoretical alkalinity to ammonia ratio of 3.57 $CaCO_3$/NH_4^+-N

---

[1] Civil and Environment Engineering Department, Virginia Tech, Blacksburg, VA 24060.

[2] Hampton Roads Sanitation District, 1434 Air Rail Ave., Virginia Beach, VA 23455.

[3] Kruger Inc., 4001 Weston Pkwy, Cary, NC 27513.

[*] Hampton Roads Sanitation District, 1434 Air Rail Ave., Virginia Beach, VA 23455; e-mail: sklaus@vt.edu.

(based on an assumed 1:1 molar ratio of HCO_3^-:NH_4^+ coming out of the digester) (Metcalf and Eddy, 2014). The alkalinity/NH_4^+-N ratio in the centrate dictates the percentage of NH_4^+ that can be removed without the addition of supplemental alkalinity.

Deammonification can take place in a single reactor or in two separate reactors. In a two-reactor configuration, partial nitritation occurs in an aerobic reactor followed by anammox occurring in an anoxic reactor (Van Dongen et al., 2001) A number of full-scale single reactor configurations are in operation including upflow granular sludge reactors (Abma et al., 2007), moving bed biofilm reactors (MBBRs) (Christensson et al., 2013; Rosenwinkel and Cornelius 2005), and sequencing batch reactor with an AMX selection device (Wett, 2007). In a deammonification MBBR, the conversion of ammonium takes place in the biofilm attached to the plastic media in which AOB exist on the exterior of the biofilm, while AMX exist deeper within the biofilm in an anoxic environment. This process is also characterized by temperatures of 25 to 35 °C, continuous flow, continuous aeration, and a hydraulic retention time (HRT) of approximately 24 hours (Lackner et al., 2014).

The biggest concern during startup of a deammonification MBBR is AMX inhibition by nitrite because AMX cannot initially consume all of the nitrite being produced by AOB. While AMX appear to be inhibited by the nitrite ion itself, AOB and NOB are susceptible to inhibition by nitrous acid (HNO_2) (Lotti et al., 2012; Strous et al., 1999). Once the AMX capacity is equal to or greater than AOB activity, the limiting factor becomes IC limitation of AOB (Wett and Rauch, 2003). In order to maintain a stable pH and avoid alkalinity limitation an aeration control strategy that takes into account alkalinity is critical. Meeting these operating requirements necessitates utilizing an automatic control system to make continuous process adjustments in order to ensure process reliability. For various deammonification reactor configurations there exists a need for reliable control systems that meet the above objectives, utilize robust sensors, and minimize operator input.

NOB repression is key to deammonification systems because NOB compete with AMX for substrate and space within the biofilm. If all of the nitrate production in the reactor is due to AMX activity, then the nitrate production ratio (eq 4) will be around 11% based on stoichiometry (eq 2). If the nitrate production ratio is any higher than 11%, then it can be assumed that the excess nitrate production is due to NOB activity. Strategies for NOB repression in sidestream systems include high free ammonia (FA) concentration (Anthonisen et al., 1976), low dissolved oxygen concentration (Wiesmann, 1994), high temperature (Hellinga et al., 1998), and transient anoxia (Kornaros et al., 2010).

pH-based aeration control is the basis for the DEMON process, an intermittently aerated deammonification sequencing batch reactor (SBR) in which the length of the aerated and non-aerated phases is controlled by a low and high pH setpoint (Wett, 2007). This process takes advantage of the high accuracy of pH sensors to control within a 0.05 fluctuation in pH based on alkalinity consumption during the aerobic phase and alkalinity production during the anoxic phase. A typical value for the low pH setpoint is 6.8 (Lackner et al., 2014). A deammonification MBBR process can be operated with intermittent aeration (Ling, 2009; Zubrowska-Sudol et al., 2011); however, continuous aeration is preferred due to simplicity of operation, more accurate readings of online signals, and elimination of the need for mechanical mixing during non-aerated phases. Continuous aeration also reduces nitrous oxide (N_2O) emissions (Christensson et al., 2013)

Another control method for a deammonification MBBR described in Christensson et al. (2013) relies on ammonia removal ratio (eq 3) and nitrate production ratio (eq 4) in the reactor to adjust continuous aeration. Where EQ stands for equalization basin (influent) concentration, and Reactor is the process (effluent) concentration.

$$NH_4^+\text{ removal} = \frac{\text{EQ}\,NH_4^+ - \text{Reactor}\,NH_4^+}{\text{EQ}\,NH_4^+} \times 100 \tag{3}$$

$$NO_3^-\text{ production} = \frac{\text{Reactor}\,NO_3^-}{\text{EQ}\,NH_4^+ - \text{Reactor}\,NH_4^+} \times 100 \tag{4}$$

The ratios are calculated from online sensor values and the dissolved oxygen (DO) set-point is incrementally increased or decreased to maintain optimum operating conditions. The optimal operating condition is for the ammonia removal to be in the range of 80 to 90% and for nitrate production to be below 12%.

There are a few publications on operation of full-scale deammonification MBBR systems (Christensson et al., 2013; Lackner et al., 2014; Rosenwinkel and Cornelius, 2005) but none give detailed information on optimization of controls. While it is recognized that pH-based aeration control is essential for operating the DEMON® process (Wett, 2007), this is the first full-scale deammonification MBBR to be operated with pH-based aeration control.

The objective of this paper is to demonstrate that pH-based aeration control optimizes performance in a sidestream deammonification MBBR and to provide detailed information on startup strategy.

Methods and Materials

Deammonification MBBR Installation. The James River Treatment Plant is a 76 ML/d facility located in Newport News, Virginia. Anaerobically digested waste activated sludge and primary sludge was dewatered using centrifuges and the centrate was sent to an equalization basin. An existing below-grade tank was modified for the installation of the sidestream deammonification MBBR (ANITA Mox™, Kruger Inc., Cary, North Carolina). Centrate was pumped from the equalization basin to the deammonification MBBR for treatment and the effluent was recycled back into the primary clarifiers. Airflow rate to the reactor was controlled and measured by a modulating, motor-actuated control valve. Mechanical mixers kept the media in suspension and the tank completely mixed during periods of non-aeration. Centrate pump speed was controlled by a variable frequency drive and was measured by a flow meter to meet a flow setpoint ranging from 75 to 250 L/min. Two deep-tank electric immersion heaters were used during startup to maintain the tank temperature at 30 °C. A blend of trace metals was added based on micronutrient requirements for bacterial growth (Grady et al., 2011) to prevent micronutrient deficiencies in both AOB and AMX populations.

Instrumentation and Control. Online sensors from YSI Inc. (Yellow Springs, Ohio) were used to monitor NH_4^+, NO_3^-, pH, DO, specific conductivity, and temperature in the deammonification MBBR. NH_4^+ and temperature were also monitored in the equalization basin. Ion selective electrode (ISE) probes were used to monitor NH_4^+ and NO_3^- and included an additional sensor for potassium correction of NH_4^+.

There were three aeration control modes available: Fixed DO control, ammonia-based floating DO control, and pH-based control. Airflow rate to the deammonification MBBR was controlled and measured by a modulating, motor-actuated control valve. The valve receives a command from the Distributed Control System (DCS) and sends a position signal in return. Airflow can be either operated continuously or intermittently. In both continuous airflow mode and intermittent control mode, the airflow can be controlled to meet an airflow setpoint (measured by a flow meter upstream of the valve) or to meet a DO setpoint (measured by a process probe). A low pH setpoint ranging from 6.3 to 6.6 was programmed to safeguard against running out of alkalinity in the deammonification reactor. When the low pH setpoint was reached, the airflow shut off while centrate feed continued to allow the system to recover. In fixed DO control mode, cascading proportional integral derivative (PID) control was used to control airflow to the reactor based on a DO setpoint, and then valve position was PID controlled based on the airflow setpoint. In ammonia-based floating DO control, the DO setpoint was adjusted to keep ammonia removal and nitrate production ratios within optimum ranges (Christensson et al., 2013). In pH-based aeration control, the DO or airflow setpoint was adjusted to meet a pH setpoint.

Bench-Scale Activity Tests. Bench-scale maximum activity tests were performed on a biweekly basis on the seed media, new media, and bulk liquid individually to monitor AMX, AOB, and NOB activity. One liter of seed media and one liter of new media were collected from the deammonification MBBR for each test. Ammonium oxidizing bacteria and NOB activity was measured under aerobic conditions for the seed and new media while AMX activity was measured under anoxic conditions. The bulk liquid test was only performed under aerobic conditions because it was assumed the amount of AMX activity in the bulk was negligible. Temperature was controlled to match the temperature in the full-scale reactor. Dissolved oxygen was monitored and manually controlled to above 4 mg/L in the aerobic sample to ensure that oxygen was not a limiting factor. For the anoxic test, the reactor was sparged with nitrogen gas to remove as much oxygen as possible and covered with a Styrofoam lid. The nitrogen gas contained 380 ppm (atmospheric concentration) carbon dioxide to prevent a drastic increase in pH. pH was monitored and manually controlled using NaOH and CO_2 to stay within the range of 6.5 to 7.5. Samples were taken at 5- to 30-minute intervals for 5 to 7 samples and analyzed for NH_4^+, NO_3^-, and NO_2^-. Ammonium oxidizing bacteria rates were determined from NOx production, NOB from NO_3^- production, and AMX from both NH_4^+ and NO_2^- consumption. NO_2^-/NH_4^+ and NO_3^-/NH_4^+ ratios were calculated for the AMX rate experiments to be compared to the stoichiometric values of 1.32 and 0.26, respectively (eq 2).

Biomass Concentration Measurements. The weight of the biomass per square meter of surface area was measured every 2 weeks for both the seed and new media. For this measurement, nine seed pieces and nine new pieces of media were selected at random from the tank. Media samples were dried at 105° C for 2 hours. The dried samples were weighed and the biomass removed by placing the carriers in a 25 mg/L disodium EDTA [ethylenediaminetetraacetic acid] solution and shaking vigorously. High-pressure tap water was then applied to each media individually to ensure that no dry biofilm remained. The media was then dried for more than 2 hours at 105°C. The difference in initial and final weight was used to calculate the biomass on the carriers. The difference in initial and final weight was used to calculate the biomass on the carriers.

Performance Monitoring. Samples for on-site monitoring were collected daily from the deammonification MBBR and equalization basin, immediately filtered through 0.45 micron filter membranes, and analyzed using HACH (Loveland, Colorado) TNT kits and a HACH DR-2800 spectrophotometer. The equalization basin samples were analyzed on-site for NH_4^+ only as NO_3^- and NO_2^- were assumed to be close to zero. Samples from the deammonification MBBR were analyzed on-site for NH_4^+, NO_3^-, and NO_2^-. NH_4^+ and NO_3^- values were used to calibrate the ISE probes as necessary. Grab samples from the two locations were also analyzed off-site for the following parameters using standard methods: Total Kjeldahl Nitrogen (TKN), total suspended solids (TSS), volatile suspended solids (VSS), chemical oxygen demand (COD), soluble chemical oxygen demand (sCOD), total phosphorus (TP), orthophosphate (OP), and alkalinity.

Results and Discussion

Startup Summary. Design parameters (Table 1) were based on the average influent (centrate) characteristics shown in Table 2. The design flow to the tank, based on centrate production was 284 m^3/d. The total volume of the tank was approximately 393 m^3 for an HRT of 33 hours at the design flow rate. Because this installation was a retrofit, the HRT was determined by the volume of the existing tank. The expected NH_4^+ removal was 204 kg N/d based on 80% removal at the design loading rate of 256 kg N/d. To achieve this removal, 133 m^3 of media was required in the deammonification reactor, which equated to a fill percentage of 32.2%, 10% of which was pre-colonized (seed) media from an established sidestream deammonification MBBR process in Sjölunda Wastewater Treatment Plant (WWTP), Malmö, Sweden. The percentage of seed media was based on previous startups that have used anywhere in the range of 2 to 15% (Christensson et al., 2013). Seeding with 10% pre-colonized

Table 1—Design parameters.

Parameter	Units	Value
Centrate flow	m^3/d	284
Centrate NH_4^+ load	kg N/d	256
Expected NH_4^+ removal (80%)	kg N/d	204
Tank volume	m^3	393
Total media fill	%	32.2%
Seed media	%	10%
Design NH_4^+ surface area load	$g\ N/m^2 \cdot d$	2.37
Design NH_4^+ volumetric load	$kg\ N/m^3 \cdot d$	0.64

Table 2—Average influent characteristics.

Parameter	Units	Value
NH_4^+-N	mg NH_4^+-N/L	890 ± 89
Alkalinity	mg $CaCO_3$/L	3400 ± 287
Alk/NH_4^+-N	mg $CaCO_3$/mg NH_4^+-N	3.83 ± 0.14
COD	mg COD/L	407 ± 74
sCOD	mg COD/L	283 ± 65

media from the Sjölunda WWTP was chosen to reduce startup time of the deammonification reactor. There are contradicting views on the importance of seeding reactors during startup. According to Christensson et al. (2013) and Lemaire et al. (2011), seeding decreases startup time, while Kanders et al. (2014) and Ling (2009) argue that seeding is not necessary. Regardless of seeding influence on startup time, seeding provides the benefit of immediate nitrite consumption by AMX, which allows for a higher initial ammonia load and reduced risk of nitrite inhibition (Kanders et al., 2014).

Startup of the deammonification MBBR was limited by AMX due to their slow growth rate and sensitivity to nitrite. The objective of the startup was to reach the design ammonia loading rate as quickly as possible without allowing ammonia, pH, or nitrite to reach inhibitory levels. Therefore, to achieve faster startup, the process must be closely monitored for ammonia, nitrite, nitrate, and pH. Additionally, aeration and ammonia loading, the main control variables, must be monitored. Aeration in the deammonification MBBR can be either continuous or intermittent. Intermittent aeration is typically utilized during startup, transitioning to continuous aeration for long-term operation. This is because during startup AMX rates are too low to meet loading provided by continuous feed, however once anammox activity rates increase, the system can transition to continuous feed. Typical DO values range from 0.5 to 1.5 mg/L (Christensson et al., 2013). Because ammonia concentrations in the centrate were dependent on digester performance, the only way to control ammonia loading was through the influent flow rate.

During startup, nitrite was controlled below 40 mg NO_2-N/L using intermittent aeration. There is a large amount of variation in reported inhibitory concentrations of nitrite to anammox ranging from 30 to 50 mg NO_2-N/L reported by Fux et al. (2004) to 275 mg NO2-N/L reported by Kimura et al. (2010). The goal was to maintain alkalinity above 150 mg $CaCO_3$/L; however, beause alkalinity data were collected on a weekly basis, alkalinity would sometimes drop as low as 80 mg $CaCO_3$/L causing suspected limitations for AOB. One hundred fifty milligrams of $CaCO_3$/L corresponded to an approximate effluent NH_4^+-N concentration of 50 to 100 mg/L, which was desirable to prevent substrate limitation to both AOB and AMX. During startup, NH_4^+ loading to the deammonification reactor was controlled to keep the NH_4^+ concentration below 350 mg NH_4^+-N/L to prevent inhibition by FA. At an ammonium concentration of 300 mg NH_4^+-N/L, pH of 8.0 and temperature of 30 °C, the FA concentration was 25 mg NH_3-N/L. Anthonisen et al. (1976) demonstrated that AOB were inhibited by FA concentrations in the range of 10 to 150 mg NH_3-N/L while NOB were inhibited at a range of 0.1 to 1 mg NH_3-N/L. Although high pH and ammonia concentrations are undesirable for optimal reactor performance, these conditions provide an effective strategy for temporarily controlling NOB proliferation though NOB adaptation to FA has been reported (Turk and Mavinic 1989; Wong-Chong and Loehr 1978). It is known that AOB are inhibited at a pH around 6.3 however the IC concentration will most likely be the limiting factor so the recommended pH setpoint is in the range of 6.6 to 6.8 (Wett and Rauch, 2003).

Performance During Startup. The deammonification reactor performance for 241 days in terms ammonium and total inorganic nitrogen (TIN) removal, temperature, effluent nitrite concentration and nitrate production ratio are shown in Figure 1. Startup was considered complete after 120 days when the reactor was treating all of the available centrate at the design loading rate. For the first week following seeding, the influent centrate was fed intermittently and then transitioned to continuous feed. An immediate reduction in ammonia of above 60% was realized resulting from AMX activity on the seed media. During the 120 days of startup, the ammonia loading rate was limited by AMX activity and was gradually increased as ammonia and nitrite removal increased (Figure 2). For the first 2

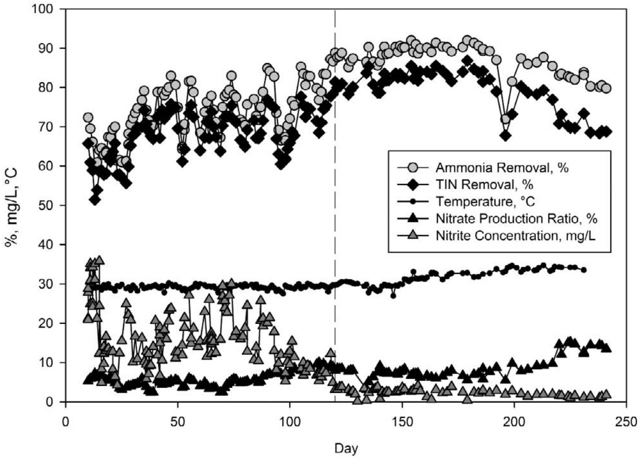

Figure 1—Ammonium and TIN removal percentages, temperature, nitrite in effluent, and nitrate production ratio. Vertical line indicates end of startup on Day 120.

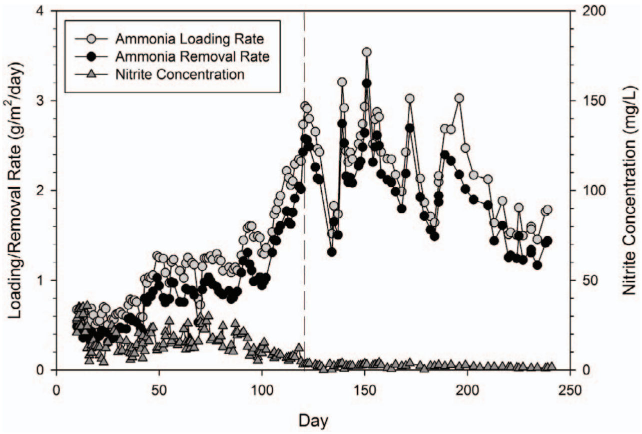

Figure 2—Ammonium loading, ammonium removal, and nitrite concentrations in the effluent. Vertical line indicates end of startup on Day 120.

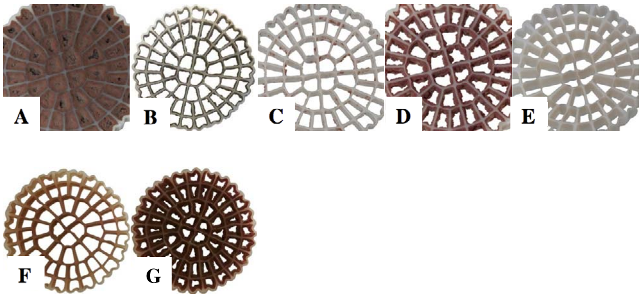

Figure 3—Biofilm development photos: (A) original seed media prior to placement in reactor; (B) seed media on Day 10; (C) sheared seed media on Day 64; (D) typical seed media on Day 64; (E) new media on Day 64; (F) new media after completion of startup on Day 120; (G) seed media after completion of startup on Day 120.

months during startup, the aeration control strategy was intermittent aeration using either an airflow or DO setpoint. This encouraged the growth of AMX by providing a distinct anoxic period and gave more control over the production of nitrite. During intermittent DO control the setpoint ranged from 1.0 to 1.3 mg/L. As AMX activity in the reactor increased, the length of the anoxic period was gradually reduced from 100 to 30 minutes with the intent of transitioning to continuous aeration. During the first 2 months of startup, biomass concentration measurements and visual inspection of the media indicated that a large amount of biomass had been sheared off of the seed media (Figure 3B compared to Figure 3A) and activity measurements indicated that there was no AMX activity on the new carriers (Figure 4). This may have been due in part to high shear forces from the mechanical mixers used during anoxic periods. For about 25% of the seed media (based on visual inspection), the AMX biomass was sheared almost completely off the media (Figure 3C). Soon after the aeration control strategy was switched from intermittent to continuous, biomass on the media increased and AMX was detected on the new carriers (Figure 3F). During startup, NO_2^--N levels were as high as 35 mg/L and then stayed below 3 mg/L after startup was complete (Figure 1). NH_4^+ removal ranged from 60 to 85% and TIN removal ranged from 50 to 80% (Figure 1).

Activity Tests. Bench-scale activity tests were used to monitor the progress of AMX development on the new and seed media throughout startup as well as determine the presence of NOB in the deammonification MBBR. Results of the activity tests are shown in Figure 4 and Figure 5. Initially, the AMX rates and biomass density on the seed media were low due to shearing of the biofilm upon placement in the reactor as discussed previously (Figure 5). Ammonium oxidizing bacteria activity was first detected on the new media 93 days after seeding (Figure 4) as indicated by anoxic nitrite consumption. On Day 114, anoxic ammonia consumption was detected in addition to nitrite consumption in a ratio indicative of AMX stoichiometry (eq 2). Throughout startup, no NOB activity was detected on the new or seed media. Ammonium oxidizing bacteria activity remained fairly constant on both the new and seed media. According to AMX stoichiometry (eq 2), for every one mole of NH_4^+ consumed, 1.32 mole of NO_2^- is consumed and 0.26 mol of NO_3^- is produced. The new media AMX ratios for NO_2^-/NH_4^+ and NO_3^-/NH_4^+ were 1.17 ± 0.37 and 0.16 ± 0.03, respectively. The seed media AMX ratios for NO_2^-/NH_4^+ and NO_3^-/NH_4^+ were 1.14 ± 0.13 and 0.15 ± 0.05, respectively. The ratios in the activity tests were lower than the stoichiometric ratios most likely due to heterotrophic denitrification, as was evident in the full-scale deammonification MBBR by NO_3 production ratios less than 11% (Figure 1).

It should be noted that as the biofilm became thicker on both the seed and new media, it became increasingly difficult to inhibit AMX activity during the aerobic activity tests. Even at DO concentrations above 6 mg/L, the biofilm thickness limited diffusion of oxygen into the inner layers of the biofilm, thus never completely inhibiting AMX activity. This was evident in the tests as NH_4^+ was being removed that did not end up as NO_x and could not be explained by assimilation. To compensate, the AOB rates were adjusted assuming no NOB activity, which is acceptable because there was no evidence of NOB activity in the full-scale reactor and the small amount of nitrate produced in the bench scale tests could be explained by AMX. It was also assumed that all excess ammonia removal was due to AMX and so NO_x production by AOB was less than it appeared because NO_2^- was being consumed by AMX.

Performance After Startup. Four months (Day 120) after seeding the reactor, all of the available centrate was being treated at greater than 85% NH_4^+ removal at the design loading rate of 2.4 g N/m^2·d, signaling the end of startup (Figure 1 and Figure 2). After startup was complete, the ammonia loading to the reactor was determined by the centrate production and NH_4^+ concentration. The maximum removal rate was 3.5 g N/m^2·d (Figure 2) and the maximum NH_4^+ removal was 92% (87% TIN removal). Similar deammonification processes have achieved 80 to 90% ammonia removal within 2 to 4 months at the design loading rate at three locations in Europe (Christens-

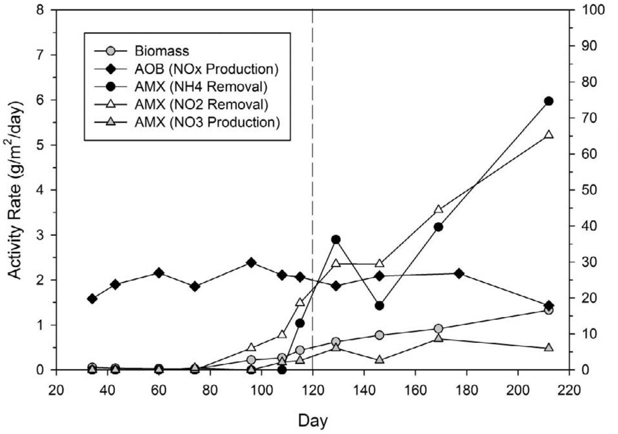

Figure 4—Bench-scale new media activity rates. Vertical line indicates end of startup on Day 120.

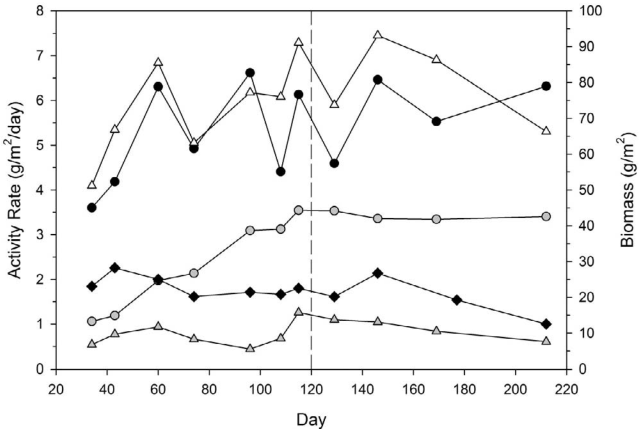

Figure 5—Bench-scale seed media activity rates. Vertical line indicates end of startup on Day 120.

son et al., 2013). When comparing startup times from different water resource recovery facilities, it should be noted that startup time depends on the centrate production and ammonia concentration (i.e., loading rate) and this will vary from facility to facility. Once startup was complete, the goal became maximizing ammonia removal.

Following startup, it was observed that a constant dissolved oxygen setpoint did not protect against running out of alkalinity in the reactor, which resulted in sporadic and dramatic decreases in pH. A low pH setpoint (air shutoff) was set up to safeguard against running out of alkalinity. However, this scenario resulted in the air frequently switching on and off because the system did not naturally maintain a constant pH at a constant DO setpoint as shown in Figure 6. A similar observation was made at a pilot-scale demonstration in which the low pH air shutoff condition was repeatedly triggered as a result of aerating at a constant airflow (Hollowed et al. 2013). As a result of these observations, an aeration control method was added in which airflow was controlled by a constant pH setpoint.

Wett and Rauch (2003) developed a model from full-scale sidestream nitrification data and found that nitrification rates started to slow below 400 mg $CaCO_3$/L and rates reached close to zero at 150 mg/L $CaCO_3$. Guisasola et al. (2007 determined that AOB activities were limited at IC concentrations lower than 150 mg/L $CaCO_3$, while NOB were not limited even at a concentration of 5 mg/L $CaCO_3$. Chen et al. (2012) found that IC limitation of AOB and AMX occurred at 200 mg/L as $CaCO_3$ in a bench-scale deammonification reactor and that AOB activity was more affected than NOB activity. Kimura et al. (2011) concluded that AMX was affected by IC limitation at 5 mg $CaCO_3$/L and, therefore, more sensitive to IC limitation than NOB but not as much as AOB. A review of full-scale sidestream deammonification processes by Lackner et al. (2014) stated that alkalinity was not an important consideration; however, results from this study indicate that alkalinity is the most important consideration for long-term operation of a sidestream deammonification MBBR, which agrees with the work of Wett and Rauch (2003) and Wett (2007).

Comparison of Aeration Control Strategies. In the deammonification MBBR, the ammonium concentration in the effluent corresponded to a given pH and specific conductivity so the three signals can be used interchangeably. It was desirable to maintain a constant pH (i.e., ammonium and specific conductivity) in the effluent to maintain near-complete use of influent alkalinity and the lowest possible ammonium concentration in the effluent. It is known that in order to maximize NH_4^+ removal in a sidestream deammonification process, it is necessary to maximize the utilization of available alkalinity (Wett, 2007). It was difficult to achieve this using DO control alone due to changes in influent ammonium concentration, alkalinity, and oxygen demand in the reactor. Although any of the three signals (pH, specific conductivity, and ammonia concentration) could have been used for aeration control, the pH signal was chosen because it was the most robust sensor (followed by specific conductivity and then ammonia), and it was the best indicator of residual alkalinity. Specific conductivity is an acceptable substitute for control as it is indicative of the alkalinity and

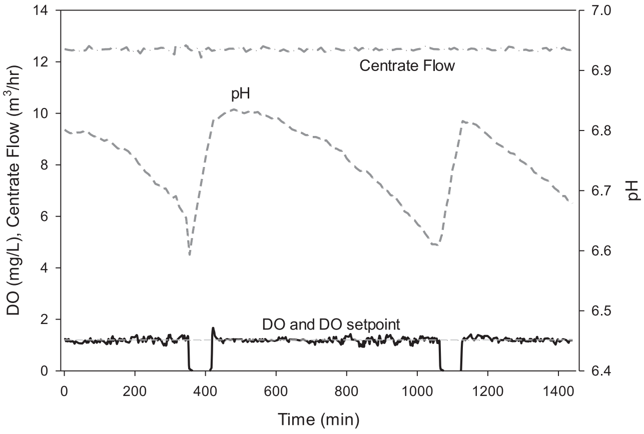

Figure 6—Example of fixed DO control leading to low pH shutoff. In this example, the airflow is programmed to shut off when the pH reaches 6.6 and come back on when pH reaches 6.8.

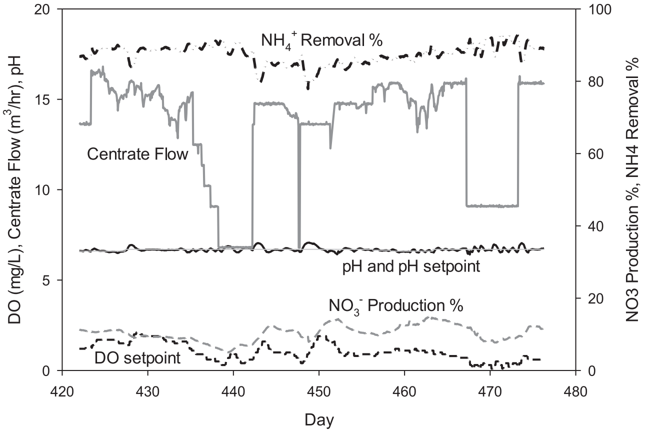

Figure 7—Performance of pH-based DO control.

ammonia concentration. The ammonia ISE probe is not as reliable and does not account for changes in alkalinity.

The airflow control valve could be controlled by an airflow or DO setpoint. In both of these methods, if the pH feedback was less than the pH setpoint (indicating that too much alkalinity was being consumed) the airflow decreased, and if the pH feedback was greater than the pH setpoint, the airflow or DO setpoint increased. The airflow control was accomplished with an appropriately tuned PID controller or logic-based algorithm. If NOB growth occurred, resulting in an increase in the effluent nitrate concentration, the pH setpoint was increased (decreasing the airflow rate) at the expense of ammonia removal until the nitrate production ratio was less than the value that would be expected to be produced by AMX alone (11%). Nitrate production typically increased over the course of days as opposed to hours. Although the pH setpoint adjustment could be automated based on nitrate production ratio, this calculation did not need to be made at the same frequency as DO setpoint.

By controlling aeration based on pH, the alkalinity consumed in the reactor was equal to the alkalinity in the influent, maintaining enough residual alkalinity to avoid IC limitation. pH-based aeration control maximized NH_4^+ removal and resulted in more consistent effluent characteristics (Figure 7) with less operator input than fixed DO control. Fixed DO control required that DO setpoint be manually adjusted to maximize ammonia removal and avoid alkalinity limitation. pH-control also maintained an NH_4^+ residual which prevents AOB or AMX activity limitations, and the subsequent induction of NOB growth. The main advantage of using pH-based DO control is that the controller will maintain a high NH_4^+ removal rate while protecting against running out of alkalinity even with changes in loading. Figure 7 demonstrates that over the course of 2 months, the controller was able to respond to disturbances caused by changes in centrate flow while maintaining an ammonia removal rate in the range of 83 to 92%.

As previously mentioned, DO control is required in order to prevent over-aeration, which inhibits AMX and encourages NOB growth. Floating ammonia-based aeration control maximizes ammonia removal but does not take into account residual alkalinity. pH-based aeration control maximizes ammonia removal and prevents alkalinity limitation using a robust and accurate sensor and is, therefore, the preferred aeration method for sidestream deammonification MBBRs. These observations are summarized in Table 3.

Table 3—Comparison of aeration control strategies.

	Prevents over-aeration?	Maximizes ammonia removal?	Prevents running out of alkalinity?
Air flow control	no	no	no
Fixed DO control	yes	no	no
Ammonia-based floating DO control	yes	yes	no
pH-based aeration control (airflow or DO)	yes	yes	yes

Conclusions

The objective of this study was to demonstrate that pH-based aeration control optimizes performance in a sidestream deammonification MBBR and to provide detailed information on startup strategy. The system reached full design capacity after four months and was consistently achieving 80 to 90% NH_4^+ removal at the design loading rate 2.4 g NH_4^+-N/m^2·d. Anammox bacteria were not detected on the new media in bench-scale testing until 120 days after seeding and no NOB activity was detected during startup. Upset periods have occurred since startup and all were characterized by a short-term increase in nitrate production and NOB activity, and resulted in a period of decreased NH_4^+ removal, until corrective action could be taken. Startup time could potentially have been shorter if continuous aeration had been used earlier by reducing shear in the reactor that resulted from mechanical mixing. pH-based aeration control proved to be an effective, simple, and stable method that was preferred over DO-based aeration control. pH-based control is crucial in a sidestream deammonification MBBR to maximize ammonia removal while protecting against alkalinity (IC) limitations.

Submitted for publication April 15, 2016; revised manuscript submitted September 14, 2016; accepted for publication October 17, 2016.

References

Abma, W. R.; Schultz, C. E.; Mulder, J. W.; Van der Star, W. R. L.; Strous, M.; Tokutomi, T.; Van Loosdrecht, M. C. M. (2007) Full-Scale Granular Sludge Anammox Process. *Water Sci. Technol.*, **55** (8-9), 27–33.

Ahn, Y. Sustainable Nitrogen Elimination Biotechnologies: A Review. (2006) *Process Biochem.*, **41** (8),1709–1721.

Anthonisen, A. C.; Loehr, R. C.; Prakasam, T. B. S.; Srinath, E. G. (1976) Inhibition of Nitrification by Ammonia and Nitrous Acid. *J. Water Pollut. Control Fed.*, **48** (5), 835–852.

Chen, Y.; Li, S.; Fang, F.; Guo, J.; Zhang, Q.; Gao, X. (2012) Effect of Inorganic Carbon on the Completely Autotrophic Nitrogen Removal Over Nitrite (CANON) Process in a Sequencing Batch Biofilm Reactor. *Environ. Tech.*, **33** (23), 2611–2617.

Christensson, M.; Ekström, S.; Andersson Chan, A.; Le Vaillant, E.; Lemaire, R. (2013) Experience from Start-Ups of the First ANITA Mox Plants. *Water Sci. Technol.*, **67** (12), 2677–2684.

Fux, C.; Marchesi, V.; Brunner, I.; Siegrist, H. (2004) Anaerobic Ammonium Oxidation of Ammonium-Rich Waste Streams in Fixed-Bed Reactors. *Water Sci. Technol.*, **49** (11-12), 77–82.

Grady, C.; Daigger, G.; Love, N.; Filipe, C. (2011). *Biological Wastewater Treatment*; IWA Publishing: Boca Raton, Florida.

Guisasola, A.; Petzet, S.; Baeza, J. A.; Carrera, J.; Lafuente, J. (2007) Inorganic Carbon Limitations on Nitrification: Experimental Assessment and Modelling. *Water Res.*, **41** (2), 277–286.

Hellinga, C.; Schellen, A.; Mulder, J. W.; van Loosdrecht, M. C. M.; Heijnen, J. J. (1998) The SHARON Process: an Innovative Method for Nitrogen Removal from Ammonium-Rich Waste Water. *Water Sci. Technol.*, **37** (9), 135–142.

Hollowed, M.; Stec-Uddin, E.; Zhao, H.; McQuarrie, J. (2013) Evaluation of the Anita-Mox Moving Bed Biofilm Reactor Process for Sidestream Deammonification at the Robert W. Hite Treatment Facility, Denver Colorado. *Proc. Water Environ. Fed.*, (4), 389–399.

Jetten, M. S. M.; Wagner, M.; Fuerst, J.; van Loosdrecht, M.; Kuenen, G.; Strous, M. (2001) Microbiology and Application of the Anaerobic Ammonium Oxidation ('Anammox') Process. *Curr. Opin. Biotechnol.*, **12** (3), 283–288.

Kanders, L.; Areskoug, T.; Schneider, Y.; Ling, D.; Punzi, M.; Beier, M. (2014) Impact of Seeding on the Start-Up of One-Stage Deammonification MBBRs. *Environ. Technol.*, **35** (21-24), 2767–2773.

Kimura, Y.; Isaka, K.; Kazama, F.; Sumino, T. (2010) Effects of Nitrite Inhibition on Anaerobic Ammonium Oxidation. *Appl. Microbiol. Biotechnol.*, **86** (1), 359–365.

Kornaros, M.; Dokianakis, S. N.; Lyberatos, G. (2010) Partial Nitrification/Denitrification Can Be Attributed to the Slow Response of Nitrite Oxidizing Bacteria to Periodic Anoxic Disturbances. *Environ. Sci. Technol.*, **44** (19), 7245–7253.

Lackner, S.; Gilbert, E.; Vlaeminck, S.; Joss, A.; Horn, H.; van Loosdrecht, M. (2014) Full-Scale Partial Nitritation/Anammox Experiences—An Application Survey. *Water Res.*, **55**, 292–303.

Lemaire, R.; Liviano, I.; Ekström, S.; Roselius, C.; Chauzy, J.; Thornberg, D.; Thirsing, S.; Deleris, S. (2011) 1-Stage Deammonification MBBR Process for Reject Water Sidestream Treatment: Investigation of Start-Up Strategy and Carriers Design. *Conference Proceedings WEF Nutrient Recovery and Management*, Miami.

Ling, D. (2009) Experience from Commissioning of Full-Scale Deammon® Plant at Himmerfjärden (Sweden). *Proceedings of 2nd IWA Specialized Conference on Nutrient Management in Wastewater Treatment Processes*, Krakow, Poland; pp 403–410.

Lotti, T.; van der Star, W. R. L.; Kleerebezem, R.; Lubello, C.; van Loosdrecht, M. C. M. (2012) The Effect of Nitrite Inhibition on the Anammox Process. *Water Res.*, **46** (8), 2559–2569.

Metcalf and Eddy, Inc. (2014) *Wastewater Engineering Treatment and Reuse*, 5th ed.; McGraw Hill: New York.

Rosenwinkel, K. H.; Cornelius, A. (2005) Deammonification in the Moving-Bed Process for the Treatment of Wastewater with High Ammonia Content. Chem. Eng. Technol., **28** (1), 49–52.

Strous, M.; Heijnen, J. J.; Kuenen, J. G.; Jetten, M. S. M. (1998) The Sequencing Batch Reactor as a Powerful Tool for the Study of Slowly Growing Anaerobic Ammonium-Oxidizing Microorganisms. *Appl. Microbiol. Biotechnol.*, **50** (5), 589–596.

Strous, M.; Kuenen, J.G.; Mike, S.M.J. (1999) Key Physiology of Anaerobic Ammonium Oxidation. *Appl. Environ. Microbiol.*, **65** (7), 3248–3250.

Turk, O.; Mavinic, D. S. (1989) Maintaining Nitrite Buildup in a System Acclimated to Free Ammonia. *Water Res.*, **23** (11), 1383–1388.

Van Dongen, L.; Jetten, M.; Van Loosdrecht, M. (2001) *The Combined SHARON/Anammox Process*; STOWA Report; IWA Publishing: London.

Wett, B. (2007) Development and Implementation of a Robust Deammonification Process. *Water Sci. Technol.*, **56** (7), 81–88.

Wett, B.; Rauch, W. (2003) The Role of Inorganic Carbon Limitation in Biological Nitrogen Removal of Extremely Ammonia Concentrated Wastewater. *Water Res.*, **37**, 1100–1110.

Wiesmann, U. (1994) Biological Nitrogen Removal from Wastewater. *Adv. Biochem. Eng.*, **51**, 113–154.

Wong-Chong, G. M.; Loehr, R. C. (1978) Kinetics of Microbial Nitrification: Nitrite-Nitrogen Oxidation. *Water Res.*, **12** (8), 605–609.

Zubrowska-Sudol, M.; Yang, J.; Trela, J.; Plaza, E. (2011) Evaluation of Deammonification Process Performance at Different Aeration Strategies. *Water Sci. Technol.*, **63** (6), 1168–1176.