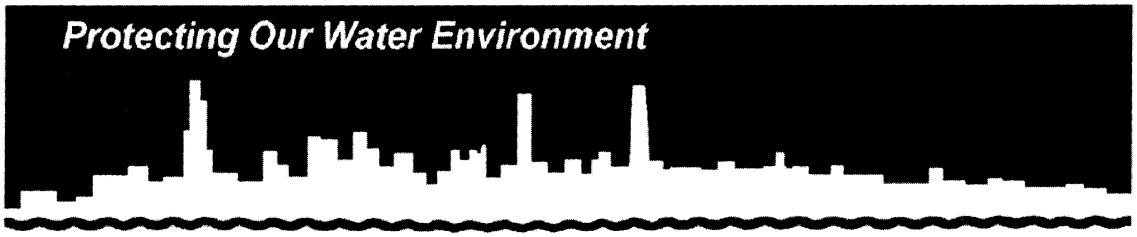

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SIDESTREAM NITROGEN REMOVAL AT THE JOHN E. EGAN WATER RECLAMATION PLANT

March 2012

Metropolitan Water Reclamation District of Greater Chicago
100 East Erie Street Chicago, Illinois 60611-2803 (312) 751-5600

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By

Joseph A. Kozak
Supervising Environmental Research Scientist

Heng Zhang
Environmental Monitoring and Research Manager

Catherine O'Connor
Assistant Director of Monitoring and Research
Environmental Monitoring and Research Division

Monitoring and Research Department
Thomas C. Granato, Director

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TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

TABLE OF CONTENTS (Continued)

LIST OF TABLES

LIST OF FIGURES

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DISCLAIMER

Mention of proprietary equipment and chemicals in this report does not constitute endorsement by the Metropolitan Water Reclamation District of Greater Chicago.

INTRODUCTION

The John E. Egan (Egan) Water Reclamation Plant (WRP) conveys its nitrogen (N) rich centrate to the North Side WRP. This practice has historically caused odor problems in the sewer lines. Common odorous N compounds include ammonia (NH_3), methyl amine, indole, skatole, and trimethyl amine (Metcalf and Eddy, 2003). The Egan WRP operates three solid bowl centrifuges which thicken digester draw from four heated mesophilic anaerobic digesters. Generally, the centrate is pumped to the North Side WRP via a sewer line for treatment, but on rare occasions, digester draw is combined with the centrate prior to pumping. This digester draw is pumped to North Side WRP when the centrifuges are not operating. Additionally, the sewer line is flushed on weekends when the centrifuges are idle; the centrifuges are typically operated Monday morning through Friday afternoon.

As part of the North Side Master Plan, the Egan WRP centrate flows and loads were determined based on intensive sampling and analysis over a fourteen-day period in March 2005 (CTE/AECOM, 2007). The Monitoring and Research Department also performed biweekly sampling and analysis of Egan WRP centrate during August 2009 (Patel, 2010). Finally, the operations data for 2010 were collected for both centrate and plant influent. The flow, concentration, and loading data from these different sources are summarized in Table 1. If the centrate is returned to the plant headworks without being treated, the plant's flow would increase by 1.0 percent for 2010 using the Master Plan data. However, the total Kjeldahl nitrogen (TKN), total suspended solids (TSS), five-day biochemical oxygen demand (BOD_5), and total phosphorus (TP) loads would increase by 45.9 percent, 17.1 percent, 0.98 percent, and 13.2 percent, respectively, for 2010 using the Master Plan data. Of all the parameters, the TKN load variations would have the most significant effect on the current nitrification capacity of Egan's aeration batteries currently in operation if the centrate were recycled to the plant headworks and may affect the plant's National Pollutant Discharge Elimination System permit compliance for ammonia N (NH_3-N).

Because of excess odor in the centrate sewer line passing through the James C. Kirie (Kirie) WRP service area and to minimize the negative impact on operations at Egan, centrate from the dewatering process could be treated to significantly reduce the TKN load such that the recycle of treated centrate will not impose harm on the secondary treatment plant. Centrate treatment at the Egan WRP is one of the management options in the Kirie Odor Strategy Plan. There are several biological and physical processes for treating dewatered sludge supernatant. However, physical-chemical processes, such as breakpoint chlorination, ion exchange, and air stripping, are often not feasible because of technical, regulatory, and cost considerations (van Loosdrecht and Salem, 2006; United States Environmental Protection Agency [USEPA], 2008). Therefore, biological N removal (BNR) is a more promising alternative.

Biological N removal in wastewater treatment occurs by two primary mechanisms: (1) biomass synthesis (N assimilation) and sludge wasting, and (2) biological nitrification and denitrification, with only the latter able to achieve high levels of N removal and low effluent concentrations of inorganic N in biological nutrient removal processes treating domestic wastewaters. Nitrification is a two-step process in which one genus of aerobic bacteria oxidize NH_3-N to nitrite N

TABLE 1: JOHN E. EGAN WATER RECLAMATION PLANT CENTRATE, PLANT FLOW, AND SELECTED WATER QUALITY PARAMETERS

	Centrate			Plant Influent
	North Side Master Plan 2005	M&R 2009	2010 Operations Data	2010 Operations Data
Flow (mgd)*	0.27	0.25	0.28	26.9
NH_3-N (mg/L)	953	277	1,078	15.1
NH_3-N load (tpd)	1.07	0.29	1.26	1.69
TKN (mg/L)	1,304	289	ND	28.5
TKN load (tpd)	1.47	0.3	ND	3.2
BOD_5 (mg/L)	183	80	ND	190
BOD_5 load (tpd)	0.21	0.08	ND	21.31
TSS (mg/L)	3,249	695	2,384	191
TSS load (tpd)	3.66	0.72	2.78	21.43
Total P (mg/L)	77	23	ND	6.1
P load (tpd)	0.09	0.02	ND	0.68
pH	ND	ND	7.6	7.3

*Centrifuges only operate five days a week; ND = No data.

(NO_2-N) followed by another genus which oxidizes NO_2-N to nitrate N (NO_3-N). In biological denitrification, organic carbon is oxidized using nitrate and/or nitrite as the electron acceptor to reduce these N species to inert N gas (N_2). For domestic wastewater treatment, 15 to 30 percent of influent N can be removed via biomass synthesis and sludge wasting (USEPA, 2010). Nitrification-denitrification processes then become an option to remove additional N from the wastewater.

Biological N removal has been used by a number of technologies to treat NH_3 rich streams such as the Egan centrate. The following BNR technological processes can be used: (1) Inexpensive Nitrification Process; (2) Bio-Augmentation Batch Enhanced Process; (3) Single-Reactor High-Activity Ammonia Removal Over Nitrite Process; and (4) Anaerobic Ammonia Oxidation.

Inexpensive Nitrification Process

The Inexpensive Nitrification (InNitri) process treats the NH_3-laden water, most often dewatered sludge supernatant, in a separate nitrification reactor to reduce the NH_3 load and increase the nitrifier population prior to recycle (USEPA, 2008). When the main stream reactor is limited due to low sludge retention times (SRTs), the nitrifiers are often washed out of the system (Philips and Kobylinski, 2007). However, by seeding that main system with nitrifiers from the InNitri reactor, the main SRT can be reduced, which can thus reduce overall capital and operations and maintenance costs. The sidestream reactor can be small in size and operated at an elevated temperature compared to the main reactor to encourage nitrifier growth. The process has been pilot tested in the United States in Arizona (Warakomski et al., 2007).

Bio-Augmentation Batch Enhanced Process

The Bio-Augmentation Batch Enhanced (BABE) process is a variation on the InNitri process. In BABE, the reactor is a batch reactor system that is fed return sludge from the main activated sludge system along with the dewatered sludge supernatant. This batch reactor is operated first aerobically to nitrify the sidestream followed by anoxic conditions to denitrify the sidestream. This means that alkalinity lost during the nitrification process will be partially recovered during denitrification, thereby eliminating the need to neutralize the BABE reactor effluent prior to recycle. Additionally, the BABE reactor effluent contains nitrifiers, which will enhance the population in the main system much like InNitri (USEPA, 2008). Full-scale testing has been done in the Netherlands (Philips and Kobylinski, 2007).

There are a number of other bio-augmentation processes that have been developed in Europe similar to BABE. Examples include the Mainstream Autotrophic Recycle Enabling Enhanced N-removal (MAUREEN), the Bio-Augmentation R (regeneration) (BAR) process, the Aeration Tank 3 (AT-3) Process, and the Biofilm Activated Sludge Innovative Nitrification (BASIN) process (Parker and Wanner, 2007). The BAR and AT-3 processes have been proven at demonstration scale while the others are in the start-up phase (USEPA, 2008).

Single-Reactor High-Activity Ammonia Removal Over Nitrite Process

The Single-Reactor High-Activity Ammonia Removal Over Nitrite (SHARON) process is operated at elevated temperatures (30–35°C) and lower SRTs to favor growth of NH_3 oxidizers (such as *Nitrosomonas* spp.) over nitrite oxidizers (such as *Nitrobacter* spp.) under aerobic conditions. This is followed by anoxic conditions where denitrifiers are then encouraged to convert the nitrite to N_2. By not oxidizing the NH_3 completely to nitrate, oxygen and energy usage is reduced (Warakomski et al., 2007). However, the addition of a carbon substrate, such as methanol (CH_3OH), may be necessary during the denitrification phase. The SHARON process is used at several locations in Europe and is being installed at the New York City Wards Island Water Pollution Control Plant (WPCP) (USEPA, 2008). The SHARON process was developed by Grontmij Consulting Engineers in cooperation with the Technical University of Delft and the Water Authorities of Zuiveringschap Hollandse Eilanden en Waarden and Hoogheemraadschap De Stichtse Rijnlanden. Grontmij Consulting Engineers holds the patent on the process design (patent number EP826639).

Anaerobic Ammonia Oxidation

The Anaerobic Ammonia Oxidation (ANAMMOX) process uses a newly isolated group of autotrophic microorganisms that can oxidize NH_3 using nitrite. These microorganisms favor elevated temperatures (above 35°C) but have slow growth rates. To balance the concentrations of the reactants needed in this process, partial nitrification of the dewatered sludge supernatant to nitrite is required. One particular strategy for obtaining nitrite is to use the first step of the SHARON process to produce nitrite, then oxidize a bypassed NH_3 stream with the nitrite (USEPA, 2008). ANAMMOX systems have been implemented in Europe in many locations (Warakomski et al., 2007). Two fixed-film processes using similar strategies to SHARON and ANAMMOX—the Oxygen Limited Aerobic Nitrification-Denitrification (OLAND) process and the Completely Autotrophic Nitrogen Removal Over Nitrite (CANON) process—are under development (Stensel, 2006). The aerobic/anoxic deammonification (DEMON) process employs a suspended growth single ANAMMOX reactor and has been implemented in the Netherlands and Germany as well as pilot tested at the Blue Plains Wastewater Treatment Plant (WWTP) in Washington, DC (Murthy, 2011). Patents for the DEMON process, equipment, and specialized bacteria seed sludge are held by Dr. Berhard Wett, ARAConsult GmbH, Austria, and Dr. Geert Nyhuis, Cyklar-Stulz GmbH, Switzerland.

TECHNOLOGY REVIEW

The SHARON and ANAMMOX processes both share the benefit of lower energy costs as less oxygen is needed to oxidize NH_3-N to nitrite. Lower biodegradable carbon additions are needed in SHARON for denitrification, and the ANAMMOX process needs no carbon supply for its dissimilatory N removal process. Additionally, both technologies have been proven to significantly reduce N loads in sidestream processes both in the United States and Europe. Therefore, these two technologies will be the focus for N removal technology review for sidestream treatment at the Egan WRP.

Single-Reactor High-Activity Ammonia Removal Over Nitrite

SHARON is the process of nitritation, where only nitrite is produced aerobically by controlling the ammonia oxidizing bacteria (AOB) and nitrite oxidizing bacteria (NOB) in the reactor. Denitrifiers are then encouraged to convert the nitrite to N_2.

Nitrification. *Microbiology.* AOBs and NOBs are autotrophic bacteria; they use carbon dioxide (CO_2) for their carbon source. More specifically, they are aerobic chemoautotrophic bacteria due to the fact that, in addition to CO_2, they require dissolved oxygen (DO) to oxidize the inorganic compounds ammonium nitrogen (NH_4-N) or nitrite nitrogen (NO_2-N) to produce cell energy. A key functional enzyme possessed by all these bacteria is NH_3 monooxygenase. This enzyme oxidizes NH_3 to hydroxylamine, which is subsequently converted to nitrite by hydroxylamine oxidoreductase (USEPA, 2010).

Stoichiometry. The oxidation of NH_3 by AOB is as follows (Metcalf and Eddy, 2003):

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O \quad [1]$$

The oxygen requirement is 3.43 g O_2/g NH_4-N. The energy producing reaction by NOB is as follows and consumes 1.14 g O_2/g NO_2-N:

$$NO_2^- + 0.5O_2 \rightarrow NO_3^- \quad [2]$$

The overall reaction accounting for alkalinity consumption by the hydrogen is:

$$NH_4^+ + 2O_2 + 2HCO_3^- \rightarrow NO_3^- + 2CO_2 + 3H_2O \quad [3]$$

Reaction 3 shows 4.57 g O_2/g NH_3-N and consumption of 7.14 grams of alkalinity measured as calcium carbonate ($CaCO_3$) per g NH_3-N (USEPA, 2008). The oxygen required and alkalinity consumption per g NH_4-N removed as calculated from Equation 3 will be less in actual bioreactors, as some of the NH_4-N removed is consumed for biomass synthesis by the nitrifying

bacteria. The oxygen consumption normalized to N removal via cell synthesis is 4.33 g O_2/g NH_3-N oxidized to NO_3-N with 3.22 g O_2 used for NH_4-N oxidation to NO_2-N and 1.11 g O_2 for oxidation of NO_2-N to NO_3-N. This fits reasonably well with the stoichiometry presented by Haug and McCarty (1972) in which biomass yields of 0.15 g volatile suspended solids (VSS)/g NH_3-N and 0.02 g VSS/g NO_2-N were measured.

The balance for NH_3 consumption including cell synthesis is as follows:

$$55NH_4^+ + 76O_2 + 109HCO_3^- \rightarrow \underset{cell}{C_5H_7NO_2} + 54NO_2^- + 104H_2CO_3 + 57H_2O \quad [4]$$

For nitrite consumption and cell synthesis, the balance is:

$$400NO_2^- + NH_4^+ + 195O_2 + HCO_3^- + 4H_2CO_3 \rightarrow \underset{cell}{C_5H_7NO_2} + 400NO_3^- + 104 + 3H_2O \quad [5]$$

However, for design purposes cell synthesis can be ignored, and a total of 4.57 g O_2/g NH_3-N oxidized to NO_3-N can be used. Approximately 7.14 g alkalinity as $CaCO_3$ per g NH_3-N can be used during the first step of nitrification, i.e. nitritation (Haug and McCarty, 1972). The first stage of SHARON produces nitrite (Reactions 1 and 4).

Kinetics and Temperature. Nitrifying bacteria have slower growth rates than carbon-consuming heterotrophic bacteria and thus require a longer SRT. At 20°C, the yield (Y) and the maximum specific growth rate (μ_{max}) for AOBs are 0.15 g VSS AOB/g N oxidized and 0.90 g VSS NOB/g VSS-day, respectively. Likewise, the Y and μ_{max} for NOBs are 0.05 g VSS/g N oxidized and 1.0 g VSS/g VSS-day, respectively (USEPA, 2010). As a comparison, aerobic heterotrophs have a yield of 0.40 g VSS/g chemical oxygen demand (COD) (Metcalf and Eddy, 2003).

At lower temperatures, nitrification kinetics for both AOBs and NOBs decrease. Therefore, for a bioreactor, longer SRTs are needed at lower temperatures to achieve low effluent NH_3-N concentrations (USEPA, 2010). The nitrification rate doubles for every 8–10°C rise in temperature (USEPA, 2008). At temperatures above 15°C, and especially between 30–40°C, the growth rates of AOBs are greater than NOBs (van Kampen et al., 2010). Additionally, nitrification rates are maximized when DO concentrations are greater than 2 mg/L (USEPA, 2008).

Temperature is a key parameter in the nitrification process, but the exact influence is difficult to determine because of its interaction with mass transfer, chemical equilibria, and growth rate. A temperature rise creates two opposing effects: increased NH_3 inhibition and increased activity of the organisms. This increased activity holds up only to a certain critical temperature, above which biological activity decreases again. Experiments with pure cultures gave an optimal temperature of 35°C for AOBs and 38°C for NOBs (Grunditz and Dalhammar, 2001). Van Hulle et al. (2007) showed that temperatures between 35°C and 45°C are optimal for partial nitritation, i.e. only converting part of the NH_3 to nitrite. However, only short-term effects were investigated. Long-term exposure to temperatures above 40°C is expected to lead to deactivation for the nitrifying organisms (Hellinga et al., 1999). Hellinga et al. (1998) concluded that temperatures above

25°C lead to an increase of the specific growth rate of AOBs, which becomes higher than that of NOBs; this is the underlying principle of the SHARON process.

In this process, nitrification of ammonium to nitrite is established in a chemostat by working at high temperature (above 25°C) and maintaining an appropriate SRT of 1–1.5 days, so that AOBs are maintained in the reactor, while NOBs are washed out and further nitrification of nitrite to nitrate is prevented. Literature values for activation energies of AOBs and NOBs range from 30 to 72 kJ/mol and from 43 to 47 kJ/mol, respectively (determined in studies between 7° and 30°C) indicating that the activity of AOBs will increase faster than the activity of NOBs (Jetten et al., 1999; Helder and De Vries, 1983; Knowles et al., 1965; Stratton and McCarty, 1967).

The partial nitrification process has been reported to be maintained at lower temperatures (between 15 and 30°C) (Yamamoto et al., 2006). These results indicate that the application of the partial nitritation process might not be restricted to effluent with temperatures around 30°C such as the effluent from methanogenic reactors but might be applicable to many kinds of industrial wastewater treatments. However, the performance of AOBs dramatically decreased below 15°C.

Ammonia and Nitrous Acid. Un-ionized NH_3 and free nitrous acid (HNO_2) concentrations have a strong influence on nitrification as these uncharged N forms are the actual substrate/inhibitor for ammonium and nitrite oxidation rather than ammonium (NH_4^+) and nitrite (NO_2^-) (Suzuki et al., 1974; Anthonisen et al., 1976). This was clearly confirmed by Van Hulle et al. (2007) for AOBs active in a SHARON reactor (van Hulle et al., 2005). The ratio between the charged (NH_4^+ and NO_2^-) and the uncharged forms (NH_3 and HNO_2) is determined by the pH and temperature values in the reactor and can be calculated based on the acid-base equilibrium. The amount of NH_3 increases with increasing pH, while the amount of nitrous acid decreases, which apparently promotes ammonium oxidizers but suppresses nitrite oxidizers. Hence, ammonium oxidizers outcompete nitrite oxidizers in a weak alkaline environment (7.5–8) in order to produce a suitable effluent in the nitritation zone. However, the potential of using this engineered environment seems somewhat limited since adaptation of the NOBs has been reported (Turk and Mavinic, 1989), and at this time the literature does not suggest that this approach has been used in any ANAMMOX system. Therefore, the achievement of stable nitritation will only occur when factors other than NH_3 and HNO_2 are regulated (Peng and Zhu, 2006). Concerning inhibition, NH_3 is the main inhibitor of nitrification at high pH (>8), whereas HNO_2 is the main inhibitor at low pH (<7.5).

pH. Despite a wide divergence of the reported effects of pH on nitrification, there seems to be a consensus that the optimum pH for both AOBs and NOBs is between 7 and 8. One explanation is the influence of pH on the NH_4^+/NH_3 and HNO_2/NO_2^- equilibria. The preference of AOBs for slightly alkaline environments is due to the fact that these organisms use NH_3 as substrate (Suzuki et al., 1974) while at certain pH values NH_3 and HNO_2 can exhibit inhibitory effects. Apart from the influence of pH on chemical equilibria in which the substrate/inhibitors are involved, direct pH effects on the activity exist (van Hulle et al., 2007). Hellinga et al. (1998)

observed a decrease in the growth rate of NOBs at pH 7 compared with pH 8, whereas the variation in growth rate of AOBs at these pH values is negligible.

Alkalinity. In nitritation, two moles of protons are produced for every mole of ammonium oxidized. To neutralize acid, two moles of bicarbonate are needed, i.e. 7.14 g alkalinity as $CaCO_3$ per g NH_3-N. Guisasola et al. (2007) and Wett and Rauch (2002) reported a reduction in AOB activity due to a bicarbonate limitation.

Phosphorus. Nitrite oxidation might also be affected by phosphorus (P) deficiency (Nowak et al., 1996). In a biological pre-treatment plant treating highly nitrogenous (1,100 mg/L TKN) wastewater (T>25°C), nitrite oxidation was substantially reduced at phosphate levels below 0.2 mg/L. The phosphate half-saturation coefficient for NOBs is about one order of magnitude higher than for AOBs (0.2 mg P/L for NOBs and 0.03 mg P/L for AOBs) (Nowak et al., 1996). NOBs are especially unable to oxidize nitrite to nitrate in the absence of phosphates, which Nowak et al. (1996) named the so-called "phosphate block".

Dissolved Oxygen. During nitrification, the DO concentration is critical for both AOBs and NOBs (Philips et al., 2002). AOBs seem to be more robust during low DO concentrations compared to NOBs (Philips et al., 2002). This difference could be explained by the higher energy released per amount of oxygen consumed by AOBs compared to NOBs.

The accumulation of nitrite at low DO is usually explained by the difference in oxygen half-saturation constant (K_O) for AOBs and NOBs (Hanaki et al., 1990). According to Hunik et al. (1994), the K_O for DO is 0.16 mg O_2/L and 0.54 mg O_2/L for the ammonium oxidizer *Nitrosomonas europaea* and the nitrite oxidizer *Nitrobacter agilis*, respectively. However, values for K_O given in literature for activated sludge vary in the range of 0.25–0.5 mg O_2/L and 0.34–2.5 mg O_2/L, respectively (Barnes and Bliss, 1983). This variation is likely due to the variation of the oxygen mass transfer efficiency in the bioreactors (Ciudad et al., 2005). The DO concentration inside a sludge floc or biofilm does not necessarily equal that of the bulk water phase. The K_O is therefore dependent on the biomass density, floc size, mixing intensity, and rate of diffusion of oxygen in the floc (Munch et al., 1996). Manser et al. (2005) showed that the K_O values for sludge in a conventional activated sludge plant ($K_O = 0.18 \pm 0.04$) and sludge in a membrane bioreactor ($K_O = 0.13 \pm 0.06$) exhibited a major difference as sludge flocs in the membrane bioreactor are much smaller.

Peng et al. (2004) and Jubany et al. (2009) were able to control the DO concentration in a bioreactor by turning aeration off at the point when NH_3 oxidation was complete. NH_3 oxidation was determined from the on-line pH and DO measurements. Nitrite oxidation was then prevented by the lack of oxygen. Aeration patterns have been proposed as alternative control for nitrate formation (Hidaka et al., 2002). Hyungseok et al. (1999) reported that nitrate formation can effectively be prevented by frequently switching between oxic and anoxic phases.

Sludge Age. AOB proliferation and washout of NOBs can be selectively accomplished by the application of an appropriate SRT in suspended growth systems because of different minimum required sludge ages. The minimum doubling time for AOBs and NOBs is 7–8 h and 10–13 h, respectively (Bock et al., 1986). Selection of the AOB on the basis of different growth rates is used in the SHARON process if operated as a sequencing batch reactor (SBR). This SHARON SBR process operates at a hydraulic retention time (HRT) (equal to SRT and therefore no washout) of one day under high temperature and high oxygen concentration to favor the growth of NH_3 oxidizers and to wash out the NOBs (van Kempen et al., 2001).

Other Factors. Ahn and Hwang (2004) and Molinuevo et al (2009) have reported that the nitritation process can successfully treat wastewater with low carbon/nitrogen (C/N) ratios although other streams with high organic content and high ammonium concentration such as swine wastewater. Yang et al. (2005) reported that monosodium glutamate manufacturing streams are also used in nitritation processes. Mosquera-Corral et al. (2005) observed stimulation of NH_3 oxidation in the SHARON process when acetate was fed as a carbon source in a 0.2 g C/g N ratio leading to an effluent with nitrite-to-NH_3 molar ratios higher than the stoichiometric ones. On the other hand, an inhibitory effect of NH_3 oxidizing activity of 10 percent was observed when 0.3 g C/g N was brought into the reactor. Hanaki et al. (1990) suggested that this inhibition was caused by a decreasing affinity of AOBs for NH_3. One possible explanation is that the transport of NH_3 from the bulk water phase to the cell of the AOB could be hindered by the presence of the crowded cells of heterotrophs which assimilate the NH_3 and consume the oxygen before it reaches the nitrifiers. However, Hanaki et al. (1990) found that for the same SRT, the NH_3 oxidation efficiency decreased at higher COD concentrations. Additionally, at a constant COD concentration, NH_3 oxidation efficiency was restored by increasing the SRT. Therefore, a moderate increase of the SRT to two to three days could potentially minimize the effect of heterotrophs on the NH_3 oxidation. However, this is outside the suggested SRT range of approximately 1–1.5 days needed to wash out NOBs.

Many industrial wastewaters rich in ammonium also contain high salt concentrations which could inhibit NH_3 oxidation. However, the SHARON process has operated successfully at high sodium chloride concentrations of 100 mM in batch experiments and 427 mM in continuous operation (Mosquera-Corral et al., 2005). This unexpected success was attributed to the adaptation of biomass to the saline environments.

Nitrifying bacteria are also sensitive to a wide range of organic and inorganic compounds including but not limited to organic solvents, amines, proteins, tannins, phenolic compounds, free NH_3 and un-ionized nitrous acid, nickel, chromium, and copper (USEPA, 2010; Metcalf and Eddy, 2003).

Denitrification. Denitrification is the second step of the SHARON process. Denitrification is the biological reduction of nitrate or nitrite, and can be assimilatory and/or dissimilatory. Assimilatory denitrification involves the reduction of nitrate or nitrite to NH_4-N for use in biomass synthesis when NH_4-N is not otherwise available. Most references to biological denitrification

refer to dissimilatory denitrification to nitrogen gas, in which nitrate/nitrite is the ultimate electron acceptor by bacteria for the oxidation of various organic and inorganic substrates.

Nitrate reduction follows a series of intermediate products, nitrite (NO_2^-), nitric oxide (NO), and nitrous oxide (N_2O) to N_2, with each step using a specific reductase enzyme in the respiratory chain to transfer electrons.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad [6]$$

During the SHARON process, this reaction chain starts at NO_2^-.

Denitrification can be accomplished by heterotrophic bacteria oxidizing organic substrates, heterotrophic nitrifying bacteria, and autotrophic bacteria. Heterotrophic bacteria are mainly responsible for denitrification in biological nitrification-denitrification processes. Most heterotrophic bacteria responsible for biological denitrification that use BOD in influent wastewater are facultative aerobic bacteria that use elemental oxygen, nitrate, or nitrite as their terminal electron acceptors for the oxidation of organic material. When oxygen is present, they will use oxygen as the electron acceptor, but the bacteria produce reductase enzymes for denitrification in the absence of oxygen (USEPA, 2010).

Microbiology. Heterotrophic bacteria capable of denitrification are very common in wastewater treatment, with *Pseudomonas* being the most prevalent. In many BNR process applications, a supplemental carbon source has been needed to (1) provide sufficient carbon for nitrate/nitrite reduction for wastewaters with lower C/N ratios; and/or (2) accelerate denitrification rates to reduce tank volume requirements. CH_3OH has commonly been used as an additive as it is inexpensive, but because of its unique single-carbon compound structure, it supports growth of a less diverse, more specific bacterial population.

Denitrification has been observed for a number of autotrophic bacteria using nitrate/nitrite to oxidize a variety of electron acceptors including iron, reduced sulfur compounds, and NH_3 (USEPA, 2010).

Stoichiometry. Denitrification reactions for wastewater and CH_3OH are shown below in Reactions 7 and 8, respectively:

$$C_{10}H_{19}O_3N + 10NO_3^- \rightarrow 5N_2 + 10CO_2 + 3H_2O + NH_3 + 10OH^- \quad [7]$$

$$5CH_3OH + 6NO_3^- \rightarrow 3N_2 + 5CO_2 + 7H_2O + 6OH^- \quad [8]$$

In these reactions 50 g alkalinity as $CaCO_3$ per 14 g N reduced or 3.57 g alkalinity as $CaCO_3$ per g NO_3-N is produced. The half reactions for the electron transfer for oxygen, nitrate, and nitrite, respectively, are shown below:

$$0.25O_2 + H^+ + e^- \rightarrow 0.5H_2O \quad [9]$$

$$0.20NO_3^- + 1.2H^+ + e^- \rightarrow 0.1N_2 + 0.6H_2O \quad [10]$$

$$0.33NO_2^- + 1.33H^+ + e^- \rightarrow 0.17N_2 + 0.67H_2O \quad [11]$$

These redox reactions suggest that less substrate oxidation is needed per unit of oxidized N removed for NO_2-N reduction compared to NO_3-N reduction so that processes that stop nitrification at NO_2-N need less carbon for denitrification (USEPA, 2010). The denitritation reaction can be expressed as (van Hulle et al., 2010):

$$3C + NO_2^- + 2H_2O + CO_2 \rightarrow 4HCO_3^- + 2N_2 \quad [12]$$

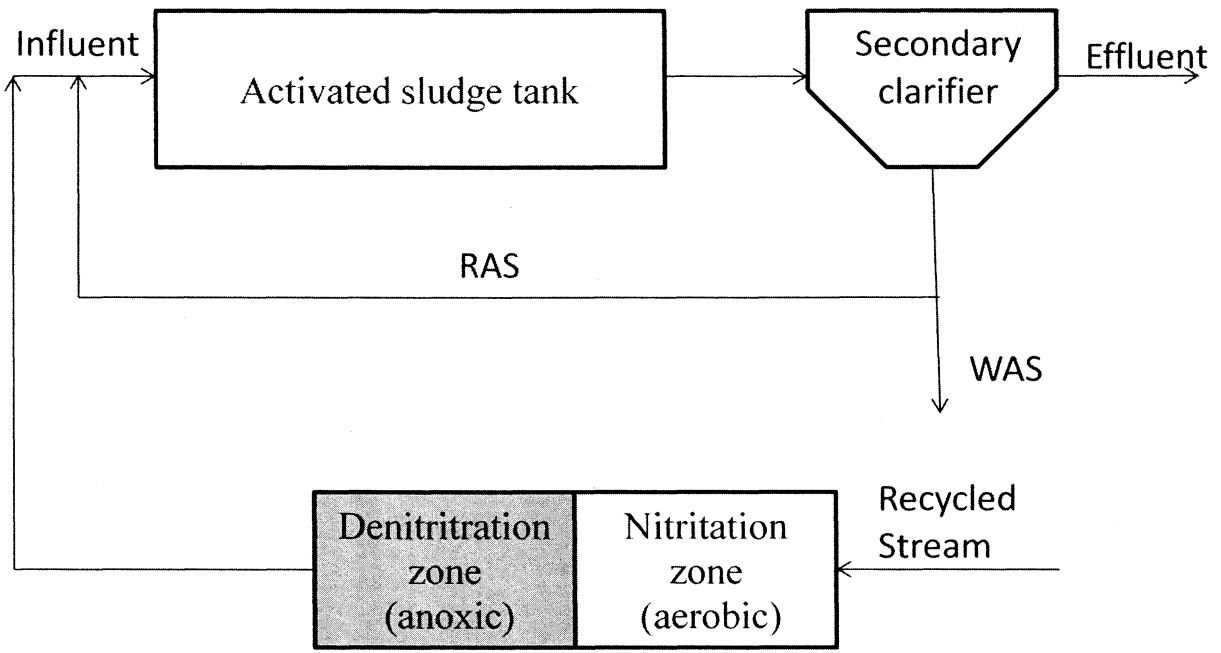
Kinetics and Temperature. Denitrification rates in BNR processes depend on many factors. A semi-empirical approach has been used to assess the rate in terms of a specific denitrification rate (SDNR) in terms of g NO_3-N reduced/g mixed liquor VSS (MLVSS)-day. Depending on the wastewater characteristics, temperature, and design loading to an anoxic zone, the SDNR may range from 0.03 to 0.20 g NO_3-N/g MLVSS-day. The Y and μ_{max} for denitrifying heterotrophs are 0.47 g VSS/g COD oxidized and 3.2 g VSS/g VSS-day, respectively. For CH_3OH, Y ranges from 0.20 to 0.30 g VSS/g COD, and SDNRs range from 0.10–0.25 NO_3-N/g MLVSS-day (Metcalf and Eddy, 2003; USEPA, 2010). However, for CH_3OH additions, a considerable acclimation period is needed to fully utilize the CH_3OH added to achieve the maximum denitrification rate. Acclimation time is not a significant issue with some other exogenous substrates, such as ethanol (C_2H_5OH) and acetate. Turk and Mavinic (1989) reported nitrite denitrification rates are 1.5 to 2 times higher than with nitrate.

Dissolved Oxygen. DO inhibition on denitrification has been shown at DO concentrations of 0.20 mg/L by Dawson and Murphy (1972). Oxygen inhibition is greater on nitrite reduction than on nitrate reduction.

pH. There is less concern about pH effects on denitrification than for autotrophic bacteria, though Dawson and Murphy (1972) showed a decrease in denitrification rates as the pH was decreased from 7.0 to 6.0 in batch tests. Generally, 4 g of wastewater influent BOD is needed per g of NO_3-N removed (Metcalf and Eddy, 2003). Additionally, the higher the biomass yield, the higher the consumption of carbon either through the wastewater or exogenous sources in order to remove the NO_3-N in the system (USEPA, 2010).

Single-Reactor High-Activity Ammonia Removal Over Nitrite Design and Process Control. The SHARON process is a two-stage process that can be operated (1) in space via aerobic and anoxic zones in a single reactor or pair of reactors as illustrated in Figure 1 or (2) in time through cycles in a completely mixed tank reactor under alternating aerobic and anoxic

FIGURE 1: SINGLE-REACTOR HIGH-ACTIVITY AMMONIA REMOVAL OVER NITRITE REACTOR DESIGN AND FLOW DIAGRAM



conditions. A generalized single tank suspended growth reactor including the suggested process controls is illustrated in Figure 2. Based on the description of SHARON above, the fundamental process control parameters are feed and draw, aeration and anoxic HRTs and SRTs (aeration time and anoxic time), temperature, pH, and DO, NH_3, and NO_2^- concentrations (Johnson et al.).

There is no solids retention in the process, so that HRT and SRT are the same and controlled by the feed and effluent pumps (Figure 2). Generally, it is expected that the HRT and SRT could be set through a programmable logical controller (PLC) computer system and pumps.

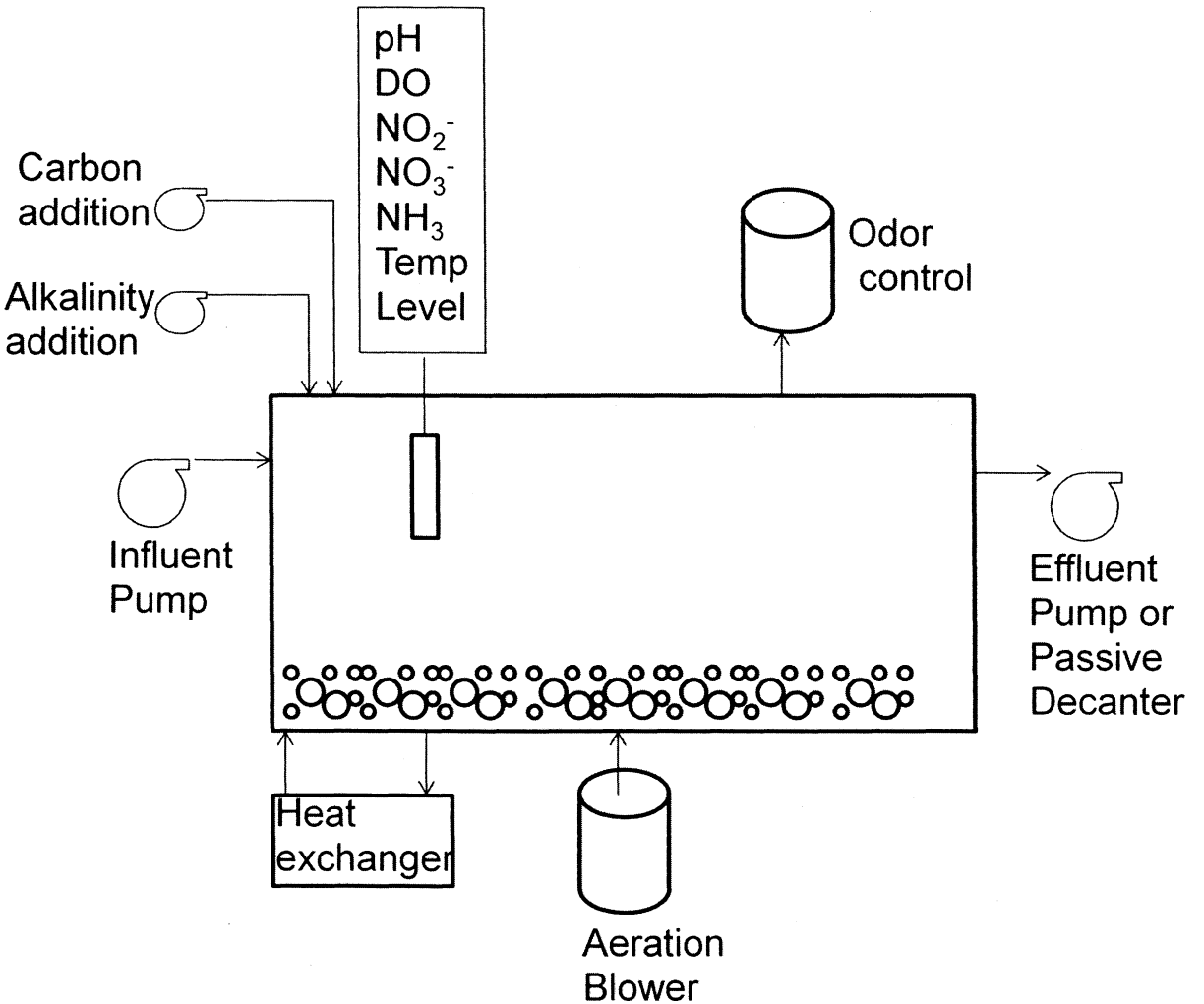
The temperature during the nitrification stage is usually maintained above 25°C, but optimally between 35–40°C. The design temperature is 35°C, but at a full-scale installation, maximum temperatures of 42°C were reached. It is expected that the dewatered sludge supernatant will enter the SHARON reactor(s) at elevated temperatures and that heat will be generated through the biochemical reactions, but a heat exchanger for the reactor may be needed to maintain the design temperature (Grontmij). Generally, most plants use an in-situ temperature probe and PLC computerized feedback system to automatically control the temperature adjustment (Figure 2). The Rotterdam plant uses a spiral flow in line heat exchanger.

The SHARON design aeration times should be between 1–1.5 days. At the design temperature of 35°C and HRT of 1-1.5 days, AOBs can have higher growth rates than the NOBs, and the NOBs are washed out from the aeration stage. Upon review of six Netherlands SHARON plants, aeration is controlled by blower and fine bubble diffuser plate systems or jet aerators (Mulder et al., 2006). The Rotterdam plant uses a Korting high efficiency jet aerator (van Kempen et al., 2010). DO levels should be maintained above 0.5 mg/L and optimally at 1.5 mg/L to ensure nitritation. However, AOBs are less affected by lower oxygen levels than NOBs. As such the system can be optimized so that AOBs outcompete NOBs (Metcalf and Eddy, 2003; Borger et al., 2008). Target DO levels are generally met through the use of an in-situ DO probe and PLC system (Figure 2).

For denitrification, DO levels should be set below 0.2 mg/L to produce the optimal anoxic environment (Metcalf and Eddy, 2003). The mixing and anoxic conditions are generally met through the same DO probe, air distribution system, and PLC computer (Figure 2).

The SHARON environment should be slightly alkaline with pHs between 7.5–8.0, but operations between 6-8 have been employed at Utrecht and Rotterdam (van Kempen et al., 2010). Under these conditions AOBs can outcompete NOBs. At higher pH, free NH_3 is more prevalent and inhibitory to NOBs; above a pH of 8, free NH_3 is inhibitory to AOBs. Below 7.5, HNO_2 is prevalent and inhibitory to both AOBs and NOBs. The pH reduction that occurs through nitrification can be reduced by (1) CO_2 stripping in the reactor which occurs during aeration cycles and can neutralize 50 percent of the pH decrease; however, CO_2 stripping cannot be controlled; (2) denitrification which can neutralize a maximum of 50 percent of the pH reduction if the SHARON process occurs in a single tank; (3) caustic chemical addition; and (4) the existing alkalinity and buffering capacity of the dewatered sludge supernatant (Mulder et al., 2006). However, it was noted that Utrecht uses sodium hydroxide and Rotterdam uses caustic soda as their alkalinity source.

FIGURE 2: SINGLE-REACTOR HIGH-ACTIVITY AMMONIA REMOVAL OVER NITRITE REACTOR AND CONTROL SCHEMATIC



Addition of exogenous carbon such as CH_3OH may be needed for the denitrification stage. The ratio of COD:N for denitrification has a stoichiometric value of 2.86 in case of denitrification of nitrate and 1.71 in case of nitrite. Including sludge production, the COD consumption is expected to be about 4 g COD/g NO_3-N and 2.4 g COD/g NO_2-N. The actual COD consumption is the best indicator for nitrification/denitrification by the nitrite route. However, online NH_3, NO_3^-, and NO_2^- analyzers can provide continuous data with respect to the process efficiency and NH_3 toxicity (Figure 2). A metering pump can automatically deliver the necessary carbon substrate based on these feedback data and PLC computer control (Mulder et al., 2006).

An off-gas odor control system may also be necessary in the SHARON process (Figure 2). The specific type of odor control equipment and operation was not discussed in the reviewed literature.

Current Single-Reactor High-Activity Ammonia Removal Over Nitrite Practices. The SHARON process has been employed at a number of WWTPs in Europe, more specifically the Netherlands, for treatment of rejection water. According to Grontmij, the SHARON process is being employed at 12 plants including one in New York. There are six Netherland plants that treat dewatered sludge supernatant. The process configurations including tank volumes for the aerobic and anoxic stages are included in Table 2. The anoxic tanks for the two tank systems are half the size of the aerobic tanks. The influent N loading, aeration times for the aerobic cycle of the SHARON process, and the N removal efficiencies for these plants are also summarized in Table 2. All plants operated between one and two days aeration time except for Utrecht and could efficiently remove up to 85–98 percent of the incoming NH_3. The N load per primary aerobic tank size ratio ranged from 330 to 580 kg N/gal for these similarly operated plants (Mulder et al., 2006).

The plants all used different exogenous carbon sources during the denitrification cycle, including CH_3OH and condensate from the sludge drying process. COD:N ratios were below 2.4 g COD per gram NO_2^- removed at all plants. The Utrecht SHARON process had a ratio of 3.0 g COD/g NO_2-N, but they were operated at longer aeration times, and a portion of the N was converted to nitrate. A ratio of 4 g COD/g NO_3-N is needed for denitrification of nitrate (Mulder et al., 2006). The anoxic residence time is 1.25 day for Utrecht and 0.5–1.4 days for Rotterdam (van Kempen, 2001); anoxic retention times for the other plants were not available.

The New York Wards Island WPCP recently commenced operation of a SHARON plant with a capacity of 5.0 kg N/day and has achieved ≥95 percent removal. SHARON plants have also recently been commissioned in Chile, Great Britain, Switzerland, and France.

Energy and Economics. The SHARON process has become very popular in the last decade due to its low operational costs. Because the process does not completely nitrify the influent NH_3, less oxygen is needed, which can translate into 25 percent lower energy costs than nitrification versus nitritation. Additionally, less carbon substrate is needed to denitrify the nitrite to N_2 relative to nitrate which can translate to a 40 percent reduction in chemical addition costs (van Kempen et al., 2001). Van Kempen et al. (2001) compared the SHARON process to a

TABLE 2: NETHERLANDS WASTEWATER TREATMENT PLANT SINGLE-REACTOR HIGH-ACTIVITY AMMONIA REMOVAL OVER NITRITE PROCESS CONFIGURATIONS AND OPERATIONS DATA

Plant	Operation Start	Wastewater Treated	Tanks	Tank 1 Volume (gal)	Tank 2 Volume (gal)	Inlet Concentration (mg NH_3-N/L)	Load (kg N/day)	Aeration Time (days)	NH_4 Removal Efficiency (%)
Utrecht	1997	Sludge dewatering	2	792,600	396,300	600–900	900	3–6	90–95
Rotterdam–Dokhaven	1999	Sludge dewatering	1	475,560	N/A	1,000–1,500	850	1.3–1.8	85–98
Zwolle	2003	Sludge dewatering	2	237,780	118,890	400–600	410	1.3–1.8	85–95
Bervwijk	2003	Sludge dewatering	2	396,300	198,150	700–900	1,200	1.3–1.8	85–95
The Hague–Houtrust	2005	Sludge dewatering	1	528,400	N/A	900–1,200	1,300	1.5–1.8	85–98
Groningen-Garmerwolde	2005	Sludge dewatering	2	1,294,580	647290	700–800	2400	1.4–1.5	≥95

N/A = Not applicable.

number of other approaches for N removal and determined that the SHARON process showed a significant cost savings of 30 percent relative to an average cost of €5.0/kg N removed for the other technologies (Table 3).

Anaerobic Ammonia Oxidation

The ANAMMOX process promotes partial nitritation, where only nitrite is produced aerobically by controlling the AOBs and NOBs in a bioreactor; this is very similar to SHARON. However, direct NH_3 conversion to N_2 using nitrite as an electron acceptor is accomplished under anaerobic conditions via the ANAMMOX bacteria. The principles of SHARON's aeration cycle where nitritation occurs is most often paired with the ANAMMOX NH_3 removal cycle. The microbiology, stoichiometry, and environmental factors controlling the conversion of NH_3 to nitrite will not be discussed in detail as it was above; rather the ANAMMOX process will be the focus.

Microbiology. Since the initial discovery, ANAMMOX bacterial activity has been reported in different wastewater treatment facilities (Schmid et al., 2005), ranging from installations treating wastewater with high N load at low DO concentrations (Siegrist et al., 1998) to municipal wastewater treatment plants (Chouari et al., 2003). Further, ANAMMOX bacteria are present in different natural environments and contribute significantly to the world's N cycle as it is found in several of the world's seas and rivers such as the Black Sea (Kuypers et al., 2003) and the Thames estuary (Trimmer et al., 2003). Depending on the organic load, up to 70 percent of the N_2 production in marine sediments can be attributed to ANAMMOX bacteria (Dalsgaard and Thamdrup, 2002). Strous et al. (1999a) showed that the bacteria responsible for the ANAMMOX process are members of the order of the *Planctomycetes*. ANAMMOX biomass has a brown-reddish color, which is probably due to the high cytochrome content (Sinnighe et al., 2002).

Stoichiometry. By estimating the mass balances over different ANAMMOX bacteria enrichment cultures, the overall stoichiometry of the ANAMMOX reaction was determined as follows (Hulle et al., 2010):

$$NH_4^+ + 1.32NO_2^- + 0.066HCO_3^- + 0.13H^+ \rightarrow 1.02N_2 + 0.26NO_3^- + 0.066\underset{cell}{CH_2O_{0.5}N_{0.15}} + 2.03H_2O \quad [13]$$

The ANAMMOX process involves the oxidation of NH_3 into N_2 in the absence of oxygen (Strous et al., 1998). Nitrite is the electron acceptor for the oxidation of ammonium and is also oxidized to nitrate which provides the reducing equivalents necessary for carbon fixation (van de Graaf et al., 1997; van de Graff et al., 1996). Since ANAMMOX bacteria are autotrophic, the conversion of NH_3 into dinitrogen gas can take place without the addition of organic matter (Jetten et al., 2001). Finally, because NH_3 is converted directly to N_2, it is important that full nitritation is not accomplished in the aeration cycle of the process or a portion of the untreated wastewater is bypassed to the anoxic zone in either a one or two-reactor system.

TABLE 3: GENERAL COMPARISON OF DIFFERENT TECHNIQUES FOR NITROGEN REMOVAL FROM REJECTION WATER

Process	Production Chemical Sludge	Production Biological Sludge	Dosage Chemicals	Energy Requirements	Ease of Operation	Cost Estimate (Euro/kg N)
Air Stripping	Yes	No	Yes	Average	Average	6
Steam Stripping	Yes	No	Yes	High	Complex	8
MAP/CAFR Process	Yes	No	Yes	Low	Complex	6
Membrane Bioreactor	No	Yes	Yes	High	Average	2.8
Biofilm Airlift Reactor	No	Low	Yes	Average	Average	5.7
SHARON	No	Low	Yes	Average	Simple	1.5

MAP/CAFR = Magnesium-ammonia-phosphate precipitation.

Kinetics and Temperature. ANAMMOX bacteria are slow-growing organisms with doubling times of about 11 days at 30°C (Jetten et al., 1999). Capuno et al. (2008) determined the biokinetic parameters of the bacteria for a biofilm reactor and observed maximum specific growth rates of 0.08 g VSS/g VSS-d and yields of 0.11 g VSS/g N oxidized at 30°C. However, van der Star et al. (2008) concluded that the doubling time of ANAMMOX bacteria is at most 5.5–7.5 days calculated on the basis of maximum conversion capacity, but possibly as low as three days. Researchers recently have claimed they optimized a bioreactor's conditions to such an extent that a doubling time of 1.8 days was achieved (Isaka et al., 2006). In the earlier stages of this research, this low growth rate and the difficulty in obtaining pure cultures strongly hindered the ANAMMOX process research (Strous et al., 1998; Strous et al., 1999a).

Several authors found that the optimum temperature for the growth of ANAMMOX bacteria was around 30–40°C (Strous et al., 1999b; Egli et al., 2001). Dosta et al. (2008) used batch tests to observe the short-term effect of temperature on ANAMMOX bacterial activity, finding that the maximum activity of non-adapted ANAMMOX biomass ranged between 35° and 40°C, while a temperature of 45°C caused an irreversible decrease of the ANAMMOX bacterial activity due to biomass lysis. Small differences in optimal temperature were found for "*Candidatus* Kuenenia stuttgartiensis" and "*Candidatus* Brocadia anammoxidans." "*Ca.* B. anammoxidans" has showed highest activity at 40°C (Strous et al., 1999b) while the highest activity of "*Ca.* K. stuttgartiensis" was observed at 37°C (Egli et al., 2001) and a pH of 8.

However, Cema et al. (2007) and Isaka et al. (2006) demonstrated that the ANAMMOX process in rotating biological contactors (RBCs) and anaerobic biological filter reactors, respectively, could be successfully operated at a low temperature of 20°C. The slow adaptation of the ANAMMOX mixed liquor seems to be a key factor in order to operate an ANAMMOX reactor at low temperatures, since a change in the operational conditions, such as wastewater temperatures dropping from the optimal range of 30-35°C to below 15°C, could lead to destabilization of the biological system (Szatkowska and Plaza, 2006).

The Egan WRP digesters operate at an optimum temperature of 95°F (35°C). Little heat loss is expected prior to the digester draw entering the centrifuges due to the high heating capacity of water. The 2011 Egan raw wastewater temperature ranged from 10.0 to 21.1 °C with an average of 16.3°C.

Dissolved Oxygen. ANAMMOX bacteria are strictly anaerobic and are inhibited by DO. Inhibition caused by low concentrations of DO was demonstrated, however, to be reversible. Egli et al. (2001) stated that DO inhibits ANAMMOX bacteria metabolism reversibly at low oxygen levels (air saturation of 0.25–2 percent) but probably irreversibly at high levels (>18 percent air saturation). Strous et al. (1997a) concluded from experiments with intermittent oxygen supply that the ANAMMOX process was reversibly inhibited by oxygen, making partial nitritation and the ANAMMOX process possible in one reactor (Strous et al., 1997a).

pH. Musabyimana et al. (2008) reported that a preferred pH operating range is 7.0 to 7.7. However, Strous et al. (199b) reported pH ranges between 6.7 and 8.3, with an optimum of 8.0.

Alkalinity. ANAMMOX bacteria, being chemolithoautotrophs, mainly utilize inorganic carbon as a carbon source. Therefore, the influent bicarbonate concentration is an important factor affecting ANAMMOX bacteria enrichment. Dexiang et al. (2007) observed low ANAMMOX bacteria activity at a low bicarbonate:ammonium ratio of 2.3. At these conditions, a limitation of the activity could occur since not enough CO_2 is present. On the other hand, high bicarbonate concentrations (bicarbonate:ammonium ratio of 4.7) also lead to inhibition. A possible explanation could be the formation of a high amount of NH_3 since the pH in the reactor reached 8.1 (van Hulle et al., 2010).

Inhibition of Substrates and Products. Nitrite concentration is an important parameter to control due to ANAMMOX bacteria inhibition. However, no agreement has been established with respect to threshold values of nitrite inhibition. Dapena-Mora et al. (2007) conducted activity tests and found that 350 mg N/L nitrite corresponded to 50 percent inhibition of the ANAMMOX process. In the presence of more than 100 mg N/L nitrite, Strous et al. (1999b) found that the ANAMMOX process was completely inhibited. Fux (2003) showed in a long-term experiment that maintaining a nitrite concentration of 40 mg N/L over several days led to the irreversible inactivation of the ANAMMOX organisms.

Different ANAMMOX genera have higher tolerances to nitrite concentrations. The inhibition experiments conducted by Strous et al. (1999b) were performed with "*Ca.* B. anammoxidans." Experiments of Egli et al. (2001) with "*Ca.* K. stuttgartiensis" showed that the ANAMMOX process was only inhibited at nitrite concentrations higher than 182 mg N/L.

The ANAMMOX process is not inhibited by ammonium or by nitrate at concentrations as high as 1 g N/L (Straus et al., 1999b). Dapena-Mora et al. (2007) observed a 50 percent activity loss with high concentrations of ammonium and nitrate (770 and 630 mg N/L, respectively).

Phosphate and Sulfide. Similar to nitrite inhibition, a difference in tolerance for phosphate exists between different ANAMMOX species. Van de Graaf et al. (1996) experienced a loss of activity for "*Ca.* B. anammoxidans" at phosphate concentrations above 155 mg P/L, while Egli et al. (2001) did not see any inhibitory effect of phosphate when a culture of "*Ca.* K. stuttgartiensis" was supplied with up to 620 mg P/L. Dapena-Mora et al. (2007) observed 50 percent inhibition of ANAMMOX bacteria activity at the same phosphate level of 620 mg P/L.

Dapena-Mora et al. (2007) showed an ANAMMOX bacteria inhibition of 50 percent at low sulfide concentrations of 9.6 mg S/L while van de Graaf et al. (1996) showed a resistance of ANAMMOX bacteria to at least 64 mg S/L in continuous and batch experiments. This large difference in sulfide inhibition could be explained by the addition of nitrate as an electron donor for the ANAMMOX biomass in van de Graaf et al. (1996); here, sulfide could reduce nitrate to nitrite, which is the preferable electron donor of the process. Recently, simultaneous removal of ammonium and sulfate by ANAMMOX bacteria has been reported by Yang et al. (2009).

Organic Carbon. During anaerobic digestion readily biodegradable organic matter is converted to biogas. Generally, the less-readily biodegradable organic matter remains in the dewatered sludge supernatant. Ruscalleda et al. (2008) found that ANAMMOX bacteria and denitrifiers could co-exist and play an important role in treating streams with high quantities of less readily biodegradable organic carbon such as digested liquor and landfill leachate. In such streams, heterotrophic denitrifying growth is limited by the absence of easily biodegradable organic carbon. As a consequence, denitrifiers are not able to dominate in these systems and cannot outcompete ANAMMOX organisms.

Several other studies reported that the presence of organic matter has a negative impact on ANAMMOX bacterial growth (Jetten et al., 1999; Molinuevo et al., 2009; Sabumon, 2007; Tang et al., 2010; Chamchoi et al., 2008; Guven et al., 2005; Jianlong and Jing, 2005). In presence of certain concentrations of organic carbon, ANAMMOX organisms are no longer able to compete with denitrifiers for nitrite. This could be due to the fact that the growth rate of denitrifiers is higher than ANAMMOX bacteria (Strous et al., 1999a). Moreover, the denitrification reaction is thermodynamically favored over ammonium oxidation; the Gibbs free energy of ANAMMOX bacteria is -355 kJ mol^{-1} (Jetten et al., 1999), while the Gibbs free energy of denitrifying bacteria is -427 kJ mol^{-1} (Rittmann and McCarty, 2001). The reported threshold concentration for organic carbon in which denitrifiers outcompete ANAMMOX bacteria differ. Guven et al. (2005) reported that ANAMMOX bacteria are no longer able to compete with heterotrophic denitrifying bacteria at a C:N ratio above 1.0, while Chamchoi et al. (2008) stated that an organic matter concentration above 300 mg COD/L or COD:N ratio of over 2.0 inactivates ANAMMOX organisms.

The ANAMMOX process removes only 90 percent of the incoming N as NH_3/nitrite and leaves 10 percent of N as nitrate in the effluent. A co-existence of ANAMMOX bacteria and denitrification in one reactor would aid to reduce the nitrate concentration in the reactor. Under anoxic conditions nitrate can be reduced by denitrifiers to nitrite as an intermediate which can be utilized by ANAMMOX bacteria for the oxidation of ammonium (Kumar and Lin, 2010). ANAMMOX bacterial activity is completely and irreversibly inhibited by low concentrations of CH_3OH (15 mg/L) and C_2H_5OH (Guven et al., 2005). This aspect must be taken into account since CH_3OH is often used in heterotrophic denitrification. A possible explanation for the CH_3OH inhibition is the formation of formaldehyde by the ANAMMOX bacteria's enzyme hydroxylamine oxidoreductase (Paredes et al., 2007).

Recent studies observed that some organic carbon sources do not have an inhibition effect on ANAMMOX bacteria activity. Kartal et al. (2007) reported that "*Ca.* B. fulgida" and "*Ca.* Anammoxoglobus propionicus" are able to oxidize acetate and propionate, respectively. Experiments by Guven et al. (2005) with propionate as the carbon source showed that ANAMMOX organisms oxidized propionate with nitrate and/or nitrite as the electron acceptor and simultaneously oxidized NH_3.

Salinity. In natural saline ecosystems, only the ANAMMOX species belonging to the genus *Scalindua* have been detected (Dalsgaard and Thamdrup, 2002). The other genera are known to inhabit freshwater ecosystems (Kartal et al., 2006). Dapena-Mora et al. (2007) found

that sodium chloride concentrations below 150 mM did not affect the ANAMMOX bacteria activity while potassium chloride and sodium sulfate had a deleterious affect at concentrations higher than 100 and 50 mM, respectively.

Biomass Concentration. Biomass concentration plays a crucial role for ANAMMOX bacteria activity. Strous et al. (1999a) found that ANAMMOX bacteria is only active when cell concentrations are higher than 10^{10} to 10^{11}/L, even in purified cultures.

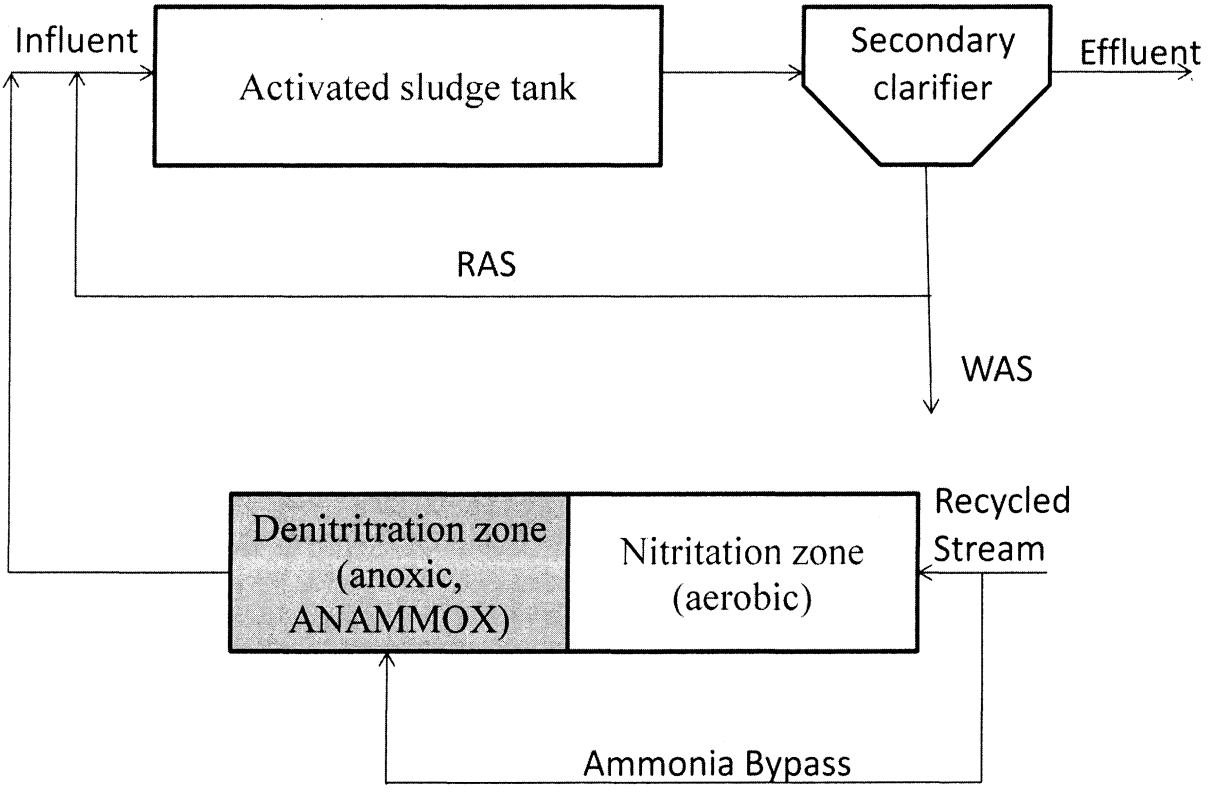
Suspended Solids. Flocculants are often used to remove colloidal organic and inorganic substances from wastewater prior to the ANAMMOX process. The effect of these flocculants on the ANAMMOX process was tested in batch tests by Dapena-Mora et al. (2007). Concentrations up to 1 g/L of a polymeric positively charged compound used as a flocculent did not cause a detrimental effect on ANAMMOX bacteria activity. In the study by Yamamoto et al. (2008), a large amount of influent suspended solids (SS) present in a partially nitrified digested liquor attached to the nonwoven materials covering the ANAMMOX biomass growing on the carriers, causing a decrease in ANAMMOX bacteria activity. The use of a flocculent improved the settleability of the influent SS and reduced their accumulation inside the reactor. However, the flocculent itself attached to the surface of the nonwoven carriers, thereby reducing ANAMMOX bacteria activity (van Hulle et al., 2010).

Other Factors. ANAMMOX bacteria activity was also found to be sensitive to visible light. A decrease in activity of 30 to 50 percent was observed by van de Graaf et al. (1996). Arrojo et al. (2006) investigated the effect of shear stress on the ANAMMOX process in a sequencing batch reactor (SBR), finding that stirring speeds up to 180 rpm had no negative effect on the performance of the ANAMMOX process.

Anaerobic Ammonia Oxidation Reactor Design and Process Control. The ANAMMOX process is a two-stage process that can be operated (1) in space via aerobic and anoxic zones in a single reactor or pair of reactors with influent bypass to the anoxic zone, as illustrated in Figure 3; and (2) in time in a single tank reactor under alternating aerobic and anoxic conditions. Based on the description of the ANAMMOX process above, the fundamental process control parameters are aeration and anoxic mixing times, SRTs, temperature, pH, DO, NH_3 and NO_2^- concentrations, phosphate, sulfide, seed concentration, and inoculation time.

With a two-reactor system, nitritation and the ANAMMOX process are physically separated in space, allowing flexibility and a more stable process performance since the steps can be controlled separately (Wyffels et al., 2004; Veys et al., 2010). In the first reactor, half of the ammonium is converted to nitrite, while in a second reactor ANAMMOX bacteria are active. One process control strategy is to maintain the influent of the ANAMMOX reactor at a constant composition in view of the nitrite toxicity. The application of the two-unit configuration would be appropriate when toxic or organic biodegradable compounds are present, since these compounds will be degraded in the proceeding nitritation step thus avoiding their entrance to the

FIGURE 3: ANAEROBIC AMMONIA OXIDATION REACTOR DESIGN AND FLOW DIAGRAM



ANAMMOX reactor (Vazquez-Padin, 2009a; Lackner et al., 2008). The two-stage systems are operated under slightly different control parameter ranges. However, the majority of research and full-scale applications are single-stage systems.

The use of a single reactor has some advantages with respect to the partial nitritation-ANAMMOX process configuration. Single-stage processes generally have higher volumetric N removal rates and lower capital costs than two-stage systems since no additional nitritation reactor volume is required (Wyffels et al., 2004). In a single-reactor system, a co-culture of aerobic and anaerobic ammonium-oxidizing bacteria is established under microaerobic conditions to avoid inhibition of ANAMMOX bacteria by oxygen and to achieve appropriate conditions to obtain partial nitritation (Strous et al., 1997a). Various names are used to describe the single-reactor systems (Fux, 2003): (1) the OLAND process (Kuai and Verstraete, 1998); (2) the CANON process (Third et al., 2001); (3) the DEMON process (Hippen et al., 1997; Wett, 2007); and (4) the Single-Stage Nitrogen Removal Using ANAMMOX and Partial Nitritation process (SNAP) (Furukawa et al., 2006). The difference lies in the organisms that were originally assumed to be responsible for anaerobic ammonium oxidation. In both the OLAND process and the DEMON process nitrifiers were assumed to perform this ammonium oxidation under microaerobic conditions (Kuai and Verstraete, 1998; Helmer et al., 1999).

Different kind of systems such as SBR, gas-lift, rotating biological contactors (RBCs), suspended growth, and moving bed reactors have been used to obtain the microaerobic conditions for the one-step process. In biofilm or granule reactors the AOB are active in the outer layers of the biofilm or granules, producing a suitable amount of nitrite for the ANAMMOX organisms that are active in the inner layers. This protects the ANAMMOX organisms from oxygen, which is consumed in the outer layers (Wyffels et al., 2004). In these single-reactor systems, the growth of NOB (and subsequent nitrate production) is prevented due to their lower affinity for oxygen compared to AOB and for nitrite compared to ANAMMOX bacteria (Hanaki et al., 1990). Possible inhibition of NOBs by free ammonium has also been suggested (Abeling and Seyfried, 1993).

When these biofilms and granular systems are used to perform the process, mass transfer resistance is considered to be the limiting step. As long as the ammonium concentration outside of the biofilm is much higher than the oxygen or nitrite concentration, ammonium diffusion into the biofilm will not limit the process rate. If the nitrite produced in the outer layer is mainly consumed in the inner layer, oxygen is the limiting factor controlling the overall rate. Sliekers et al. (2003) and Szatkowska et al. (2007) reported that oxygen transfer was indicated as the limiting factor for a laboratory-scale air-lift and a pilot-scale moving bed reactor, respectively. This oxygen limitation can be attributed to the slow diffusion into the biofilm/granule or from poor gas-liquid transfer. Through computer model simulations, Hao et al. (2001) showed that the optimal bulk oxygen concentration for a CANON biofilm reactor is approximately 1 mg O_2/L. However, van Hulle et al. (2010) summarized laboratory-scale experimental studies of one-reactor systems and DO levels between <0.1 to 1.8 mg/L were used. Limiting oxygen transfer into the biofilms which would inhibit the ANAMMOX process, encourage AOB growth, and discourage NOB growth is fundamental in optimizing the DO levels of the system. As stated above, DO levels should be maintained above 0.5 mg/L to ensure nitritation.

A generalized process control diagram of the most common ANAMMOX process, the DEMON reactor (a SBR with suspended growth), is illustrated in Figure 4 (Johnson et al.). Time control defines the operation cycle involving a fill/react phase, a settling period, and a decant period. The react period includes both partial nitritation and anaerobic NH_3 oxidation. The time for each of these periods/phases is optimized through reactor performance evaluations (Wett et al., 2007). However, de Mooij and Thomas (2010) cited a fill and aerate time of 4.5 hours, a sedimentation time of 45 minutes, and a discharge time of 45 minutes for the Apeldoorn plant. This DEMON reactor is operated at a sludge age of approximately 20 days (de Mooij and Thomas, 2010).

Feed pumps are used to fill the reactor, and draw pumps are used during the decant phase. Solids inventory and SRT are maintained through both the sludge and recycle pumps. The recycle pump feeds a hydrocyclone to enrich the ANAMMOX biomass (Figure 4).

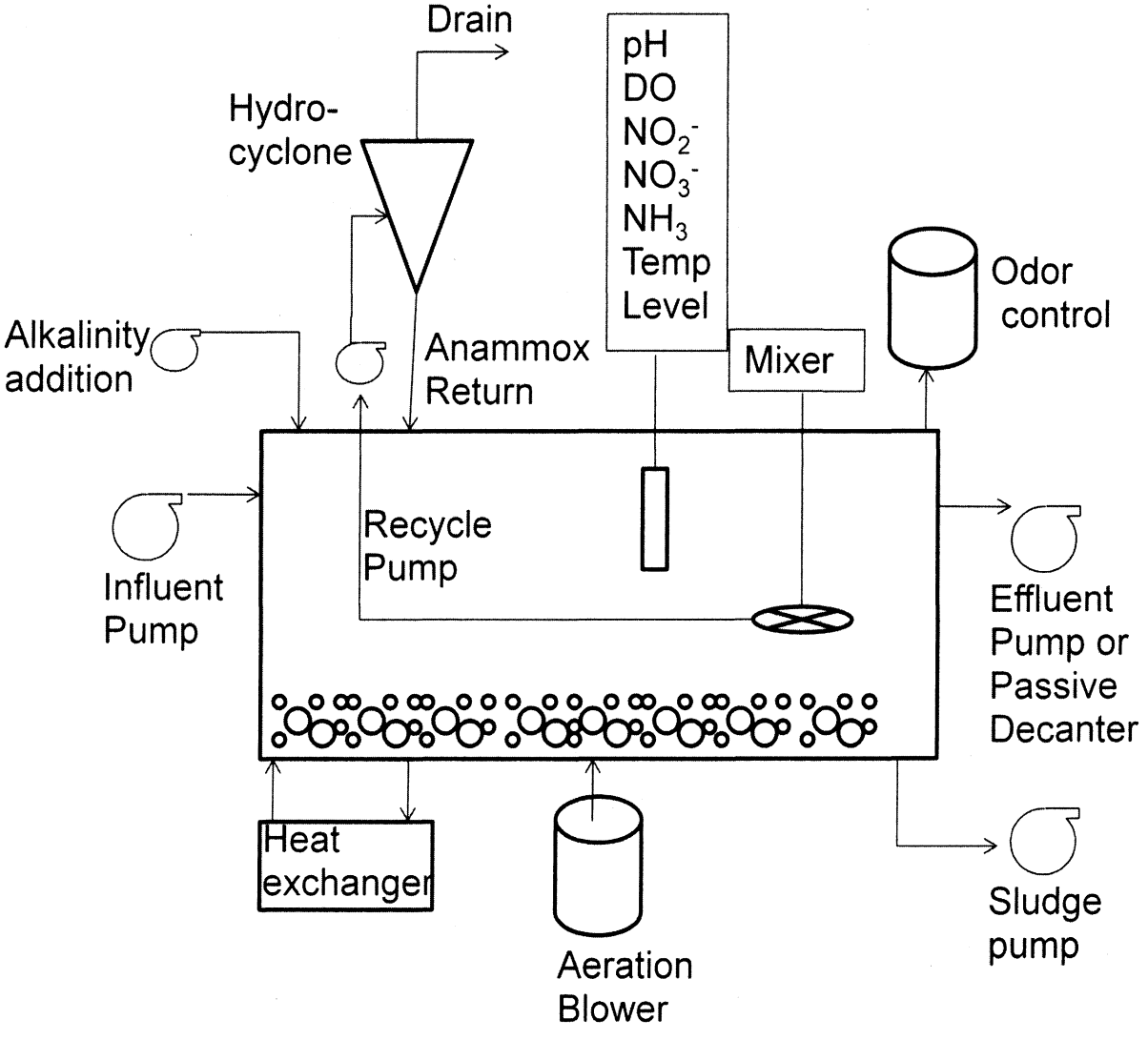
A major disadvantage of these autotrophic N removal processes is the low growth rate of AOB and ANAMMOX bacteria. The performance of reactors involving slow-growing bacteria can be enhanced by applying high SRT, such as developing biofilms (SBRs, MBRs, and RBCs) (Vazquez-Padin et al., 2009b). Since ANAMMOX bacteria are predominantly aggregated in a heavy granular fraction, the cyclone shown in Figure 4 produces centrifugal force to select the ANAMMOX bacteria populations while wasting the AOBs and NOBs. This also decouples the SRT from the system's operation. The substantially higher mass of ANAMMOX bacteria in the system compensates for the slower kinetics of these organisms compared to the AOBs. This surplus in retention of the compact red granules formed by ANAMMOX bacteria enhances the robustness and treatment capacity. By doubling the mass ratio of ANAMMOX bacteria compared to the AOBs, the robustness of the process against disturbances like over-aeration, temperature drop, or a flush of excess organic carbon is drastically improved.

The temperature during the nitritation stage is maintained above 25°C but is optimally between 35°–40°C to select AOB over NOB growth. Several authors found that the optimum temperature for ANAMMOX bacteria is in a similar range of 30°–40°C. Much like the SHARON reactor, the temperature is generally automatically controlled via a temperature probe and PLC computer feedback system and heat exchanger (Figure 4). For the Apeldoorn plant, a design temperature was set at 30°C, but the acceptable variability around this target temperature is unknown (de Mooij and Thomas, 2010).

Optimum pH levels for AOBs are 7.5–8.0, while ANAMMOX bacteria can operate between 6.7 to 8.3 with an optimum of 8.0. The pH reduction that occurs through nitrification can be adjusted by CO_2 stripping, existing alkalinity of the dewatered sludge supernatant, and alkalinity addition (Mulder et al., 2006). The actual duration of the aeration and anaerobic cycles necessary for the DEMON process is generally governed by pH control.

To operate the SBR's aeration and anaerobic cycles based on pH control, the aeration system is activated within a very tight pH-control bandwidth of 0.01. Aeration is initiated at the upper pH set point that is to be determined based on the wastewater to be treated. The nitritation reaction leads to hydrogen (H^+) production and drives down the pH value to the lower set point where aeration stops. In the subsequent anaerobic step, all the accumulated nitrite is used to

FIGURE 4: ANAEROBIC AMMONIA OXIDATION (AEROBIC/ANOXIC DEAMMONIFICATION) DEMON REACTOR AND CONTROL SCHEMATIC



oxidize NH_3. In the course of this biochemical process the recovered alkalinity, as well as the continuous feed of alkaline wastewater, leads to an increase in pH to the upper set point where aeration is again activated (Wett et al., 2007). For example, a plant can operate in a pH range of 7.00 to 7.01; at 7.01, the aeration step would begin thereby decreasing the pH to 7.00, the anaerobic step would then begin increasing the pH to 7.01. The Apeldoorn plant used a pH bandwith of 0.02 for this process control (de Mooij and Thomas, 2010). Aeration and mixing is generally provided through air distribution systems such as blowers and diffuser plates (Figure 4). Mechanical mixers can also be used during the anaerobic cycle (Wett et al., 2007).

Alkalinity adjustments can be made through the alkalinity metered feed pump. However, de Mooij and Thomas (2010) suggest that alkalinity-to-NH_3 ratios are usually near unity. At this ratio, sufficient alkalinity is available during the nitrification cycle to maintain a stable pH, and alkalinity addition through caustic soda is only used as a backup control.

The DO control for the DEMON process is specified at a low range, closer to 0.3 mg/L in order to prevent rapid nitrite accumulation and to maintain a continuous repression of the second oxidation step of nitrite to nitrate (Wett et al., 2007). This target DO is generally monitored and set through a DO probe, PLC system, and air distribution system such as blowers and diffuser plates (Figure 4). However, as stated above, the aeration and anaerobic cycles are managed through pH control, and meeting the target DO level during the aeration phase.

Nitrite concentration is a strong inhibitor of ANAMMOX bacteria. If nitrite is consumed at about the same rate as it is produced, this inhibition effect is not of significance. No negative effect of nitrite was observed by Vazquez-Padin et al. (2009a) even though a mean nitrite concentration of 25 mg N/L was registered in the bulk liquid, which may mean that low nitrite concentration occurred inside the bioreactor granules. The ANAMMOX process is not inhibited by ammonium or nitrate up to concentrations of at least 1 g N/L (Strous et al., 1999b).. Dapena-Mora et al. (2007) observed a 50 percent activity loss with high concentrations of ammonium and nitrate (770 and 630 mg N/L, respectively). The NO_2^- concentration can be monitored via an in-situ probe.

An off-gas odor control system may also be necessary. Additionally, wastewater levels, NH_3 concentrations, and nitrate concentrations can be measured via in-situ probes to monitor and optimize the DEMON process.

Two strategies are possible to start up a single-reactor autotrophic N removal system. The first method is the inoculation of nitrifying biomass into an ANAMMOX reactor and supplying air into the reactor to maintain microaerobic conditions. Otherwise, a partial nitritation reactor can be operated under oxygen-limited conditions, obtaining an ammonium:nitrite ratio of 1:1 before ANAMMOX biomass is inoculated into the reactor (Paynaert et al., 2004; Gong et al., 2007). The second strategy seems to be more appropriate since a decrease of ANAMMOX bacteria activity has been observed when the first method is applied (Sliekers et al., 2003; Sliekers et al., 2002; Liu et al., 2008). This high nitrifying activity can protect the ANAMMOX bacteria from oxygen and provides them enough nitrite. The inoculation of ANAMMOX-enriched biomass in a partial nitritation reactor accelerates the start-up and increases N removal after one or two months instead of the several months or even years without inoculation (Vazquez-Padin et

al., 2009a; Vazquez-Padin et al., 2009c). Moreover, only a limited amount of ANAMMOX biomass is necessary to start up the CANON process with this second strategy.

Various reactors were developed and optimized to enrich ANAMMOX bacteria and start up the ANAMMOX process, such as fluidized bed reactors, SBRs, RBCs, and gas-lift reactors (Srous et al., 1998, van de Graaf et al., 1996 Egli et al., 2001; Sliekers et al., 2003). SBRs offer several advantages over other reactors, including an efficient biomass retention, homogeneous mixture, and reliability for a long period of operation. Therefore, SBRs have been proven to be a very suitable system for the ANAMMOX process start-up (Jetten et al., 1999; Strous et al., 1998; van Dongen et al., 2001). Recently, membrane bioreactors have also shown to be an ideal system for start-up due to their complete biomass retention (Wang et al., 2009). Start-up still took up to four months or more using SBRs, and membrane fouling has been observed in MBRs (Wang et al., 2009; Third et al., 2001; Dapena-Mora et al., 2007; Nutchanat and Suwanchai, 2007). For a quicker start-up, appropriate seed sludge should be selected.

Current Anaerobic Ammonia Oxidation Practices. The ANAMMOX process has been employed at a number of WWTPs in Europe. Most of these plants use the ANAMMOX process to treat dewatered sludge supernatant. The operation conditions and performance of several pilot and full-scale single-reactor ANAMMOX systems are summarized by van Hulle et al. (2010) (Table 4); plant names were not provided. For the processes treating dewatered sludge supernatants, N removal ranged from 60 to 90 percent at temperatures between 23° to 35°C, pHs between 7.05 and 8.05, and DO levels between 0.3 and 3.0 mg/L. No information on HRTs, SRTs, or N loads was provided.

Three full-scale DEMON plants are in operation, in Austria, Switzerland, and Germany. Up to 20 DEMON facilities in Europe, including the Netherlands, Croatia, and Hungary, are expected to be on line by the end of 2012 (Murthy, 2011). The design criteria and performance for a number of United States and European plants are provided in Table 5 (Murthy, 2011). Typical volumetric design criterion is 0.7 kg/m^3-day with removal over 80 percent total N (TN) per day. The NO_3-N effluent concentrations are <10 percent, especially if biodegradable COD is available for conventional denitrifiers.

Energy and Economics. The ANAMMOX process has become very popular in the last decade due to the low operational costs. Because the process does not completely nitrify the influent NH_3, less oxygen is needed, which can translate into over 60 percent lower energy costs than complete nitrification. Additionally, unlike SHARON, no carbon substrate is needed by the ANAMMOX bacteria, because NH_3 is an electron donor (van Hulle et al., 2011). Strass, Austria, recently switched from a single reactor SHARON process to a single reactor ANAMMOX (DEMON) process; energy use decreased from ~2.9 kWh/kg N removed to ~1.2 kWh/kg N removed. For comparison, an attached growth ANAMMOX process employed in Hattingen, Germany, uses 5.7 kWh/kg N removed (Murthy, 2011).

TABLE 4: WASTEWATER TREATMENT PLANT ANAEROBIC AMMONIA OXIDATION PROCESS CONFIGURATIONS

Reactor Type	Reject Water	Tank Volume (gal)	pH	Temperature (°C)	DO (mg/L)	Removal Efficiency (%)
SBR	Dewatered sludge supernatant	132,086	7.05–7.10	25–30	0.6	84
SBR	Dewatered sludge supernatant	105,669	7.05–7.10	25–30	0.4	90
SBR	Dewatered sludge supernatant	1,083	7.4–7.5	25	0.65	90
Upflow Reactor	Dewatered sludge supernatant	158,503	8.0	30–35	1.3	75–80
MBR	Landfill leachate	Full scale	N/P	N/P	0.33	73
MBR	Landfill leachate	Full scale	6.9	N/P	0.33	84
Moving Bed	Dewatered sludge supernatant	11	8.0–8.5	27	0.5	60–70
Moving Bed	Dewatered sludge supernatant	11	8–8.1	28–29	0.12–0.22	71–75
Moving Bed	Dewatered sludge supernatant	5,548	7.6–8.0	23–27	0.38	62
Moving Bed	Dewatered sludge supernatant	Full scale	7.8	30	0.35	64
Moving Bed	Dewatered sludge supernatant	Full scale	8	27	0.21	72
RBC	Landfill leachate	70,006	8.3	28	0.15–0.26	40–70
RBC	Landfill leachate	8,718	7.3	16	0.25–0.57	30–70
RBC	Landfill leachate	63,401	8.1	14	1.7	30–70

N/P=Not provided.

TABLE 5: OVERVIEW OF AEROBIC/ANOXIC DEAMMONIFICATION PLANT DESIGN CRITERIA AND PERFORMANCE

Plant	Average Load (kg/d)	Tank Volume (m^3)	Design Loading Rate (kg/m^3-d)	TN Removal (%)	NH_3-N Removal (%)
Apeldoorn, Netherlands	1,896	2,915	0.66	>80	>90
Thun, Switzerland	399	606	0.67	>90	>90
Glarnerland, Switzerland	249	379	0.69	>90	>90
Strass, Austria	599	492	1.2	>80	>90
Blue Plains, DC	9,072	21,955	0.58	ND	>80
Alexandria, VA	1,284	3,028	0.8	>90	ND

ND = No data.

SUMMARY

The conveyance of NH_3 rich centrate from Egan to North Side has historically caused odor problems in the service area. Currently, the centrate cannot be recycled as the nitrification capacity of Egan's aeration basins is limited. In order to mitigate the odor problem in the sewer and recycle the centrate, sidestream treatment technologies were reviewed. The SHARON and ANAMMOX processes have been determined to be the most suitable applications for the Egan WRP.

SHARON is the process of nitritation, where only nitrite is produced aerobically by controlling the AOBs and NOBs in the reactor. Generally, AOBs outcompete NOBs through temperature and SRT control in a second step. Denitrifiers convert the nitrite to N_2. The fundamental process control parameters, whether employing a single- or two-tank system, include aeration and anoxic aeration time, anoxic time, temperature, pH, DO, NH_3, and NO_2^- concentrations.

ANAMMOX is the process of partial nitritation, where only nitrite is produced aerobically by controlling the AOBs and NOBs in a bioreactor. However, unlike SHARON, direct NH_3 conversion to N_2 using nitrite as an electron acceptor is accomplished under anaerobic conditions via the ANAMMOX bacteria. Whether a single-tank or two-tank system is employed, the fundamental process control parameters are aeration and anoxic aeration times, SRTs, temperature, pH, DO, NH_3 and NO_2^- concentrations, phosphate, sulfide, seed concentration, and inoculation time.

Because the SHARON process does not completely nitrify the influent NH_3, less oxygen is needed, which can translate into 25 percent lower energy costs than complete nitrification. Additionally, less carbon substrate is needed to denitrify the nitrite to N_2 relative to nitrate, which can translate to a 40 percent reduction in chemical addition costs. However, the ANAMMOX process can translate into over 60 percent lower energy costs than complete nitrification, and, unlike SHARON, no carbon substrate is needed by the ANAMMOX bacteria, because NH_3 is an electron donor.

The SHARON process has been successfully employed at eleven WWTPs in Europe and has indicated NH_3 removal rates of 85–98 percent. More recently New York's Wards Island WTTP began operation of a 1.85 MGD and 5000 kg NH_3-N/day facility in 2009. The ANAMMOX process has been employed successfully via DEMON reactors at eleven WWTPs, and ten additional DEMON plants are under construction including one at the District of Columbia Water and Sewer Authority Blue Plains WWTP in Washington, D.C., and one at the Hampton Roads WWTP in Alexandria, Virginia (please note that Hampton Road constructed a universal reactor that can apply both the SHARON and DEMON processes). Different ANAMMOX technologies have shown the ability to reduce NH_3 by 60–90 percent, while DEMON plants have consistently shown removals of >80 percent.

FUTURE INVESTIGATION

An ANAMMOX process has greater energy and chemical savings compared to SHARON while achieving similar NH_3 removal and should be pursued at the Egan WRP for N removal. As mentioned above, the ANAMMOX process can be achieved in (1) multiple or single reactors; (2) attached growth systems; (3) suspended growth systems (DEMON); or (4) granular systems. For the Egan WRP, the ANAMMOX system design should be selected based on the space needs, volumetric loading, performance reliability, operability, energy demand, and length of start-up.

Through personal communication with Beverley Stinson of Architectural Engineering, Consulting, Operations and Maintenance (AECOM), an expert in the ANAMMOX process, in January 2012, the current promising technologies of ANAMMOX are (1) DEMON as mentioned above, (2) an attached growth moving bed bioreactor process patented as ANITAMOX by Kruger, and (3) an ANAMMOX upflow granular process (AUGP) patented by Paques.

Briefly, the ANITAMOX process employs polyethylene biofilm carriers operating in mixed motion within an aerated wastewater treatment basin. For ANITAMOX, there is a two-layer biofilm system with ANAMMOX bacteria forming an interior layer where anaerobic conditions exist around the carrier and an outer layer of nitrifiers where aerobic conditions exist. Nitritation occurs in the outer layer whereby the NO_2-N produced diffuses into the inner ANAMMOX bacterial layer for subsequent deammonification. The AUGP process is a multi-reactor system whereby nitritation occurs in a suspended growth reactor (e.g. SHARON), followed by a settling tank to remove the NOBs, followed by upflow of the NO_2-N and NH_3-N rich effluent through a granular ANAMMOX reactor for deammonification.

A comparative summary of the advantages and disadvantages of the three technologies is provided in Table 6. DEMON and ANITAMOX have similar footprint needs (0.7–1.2 kg N/m^3-day), but AUGP has a much smaller footprint relative to N loading rates (1.7–2.0 kg N/m^3-day). However, additional space is needed for the nitritation reactor and settling basin for AUGP. All three technologies have similar NH_3-N removal rates (90 percent), but DEMON has shown to have slightly higher total nitrogen removals (85 percent). Based on the current operating plants, DEMON is much more energy efficient (1.0–1.3 kWh/kg NH_3-N removed) relative to ANITAMOX (1.45–1.75 kWh/kg NH_3-N removed); AUGP is considered to have higher energy demands than either of the other two technologies, but actual data is not available according to Stinson. DEMON has also been documented to have slightly shorter start-up periods with seeding (two to five months) relative to ANITAMOX (5 months) and AUGP (<6 months).

Dissolved oxygen and pH control are crucial for efficient DEMON operation as mentioned above. DO levels are usually maintained at <0.3 mg/L with a very narrow but achievable pH bandwidth of 0.01 to 0.02 during the aerobic/anaerobic cycles through automated controls; however, nonreversible toxic conditions can occur at NO_2-N concentrations of greater than 5 mg/L. Inhibition caused by elevated DO concentrations is reversible.

TABLE 6: COMPARATIVE SUMMARY OF ANAEROBIC AMMONIA OXIDATION TECHNOLOGIES

		DEMON	ANITAMOX	AUGP
Volumetric Loading Rates	kg N/m3/day	0.7–1.2	0.7–1.2	1.7–2.0
Performance TN Removal	%	90% NH_3-N 85% TN	90% NH_3-N 80% TN	90% NH_3-N 75% TN Higher residual NO_3-N
Energy Demand	kWh/kg NH_3-N removed	1.0–1.3	1.45–1.75	No data, but literature suggests higher
Start-Up	Months	2–5 months w/ seed and cyclone	5 months w/ 3% seeding	<6 months w/ seeding
Sensitivity/Flexibility		pH & DO control <0.3 mg/L DO <5 mg/L NO_2-N	DO control 3 mg/L DO 50 mg/L NO_2-N	DO control Elevated NO_2-N tolerated

Operational controls for ANITAMOX and AUGP are much more robust. ANITAMOX biofilms are very self-regulating with respect to diffusion of DO and NO_2-N. Bulk liquid concentrations can be on the order of 3 mg/L DO and 50 mg/L NO_2-N. AUGP DO controls for the nitritation cycle in the aerobic process are the same as the aerobic cycle for SHARON. The separate upflow granular process is strictly anaerobic and has been observed to be extremely tolerant of elevated NO_2-N concentrations according to Stinson.

Overall, the DEMON process is much more mature, energy efficient, and has been employed at many more full-scale operations than ANITAMOX or AUGP, the latter of which has only been used for industrial applications. Although there are some operational concerns with DEMON, there have been no documented failures due to elevated NO_2-N or DO concentrations, the process design is relatively simple as it is operated like a SBR, and automated control is done primarily through on-line pH and DO probes. Additionally, most systems have PLC safety logic controls to prevent overaeration and NO_2-N accumulation.

Given the DEMON technology's viability and cost effectiveness, it is the Environmental Monitoring and Research Division's recommendation that it be investigated for nitrogen removal of the centrate sidestream at the Egan WRP. Because the DEMON process is a patented technology, the United States vendor World Water Works will be approached to help in this investigation.

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