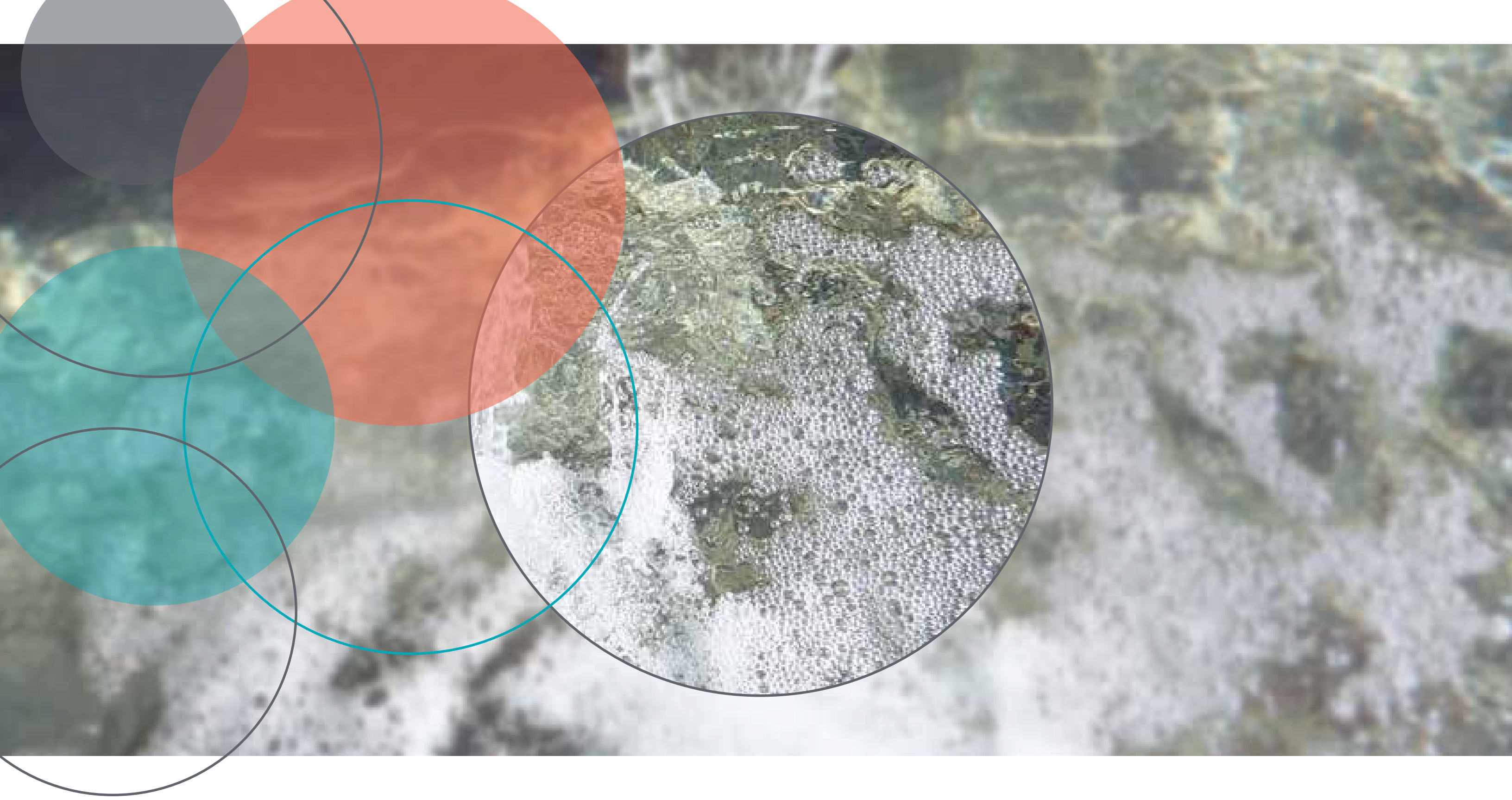


Deammonification

MARCH 2019

The *Deammonification Compendium* was prepared in 2012 and later updated in 2014 as part of the Nutrient Removal Challenge. The *Deammonification Compendium* serves as a succinct primer on the deammonification process and applications in water resource recovery facilities. This 2019 compendium updates prior versions based on reports and documents generated by the Nutrient Removal Challenge. As such, updates are limited in scope and not intended to capture all recent industry and research activity related to deammonification.

THE Water Research FOUNDATION

ACKNOWLEDGMENTS

We acknowledge the following contributors and reviewers (alphabetically).
The findings from the Nutrient Removal Challenge were incorporated into this 2019 edition by:

Bryce Figdore
HDR Inc.

JB Neethling
HDR Inc.

David Stensel
University of Washington

The following individuals contributed to earlier versions of this compendium:

Charles Bott
Hampton Roads Sanitation District

Gregory Bowden
AECOM

Ramesh Goel
University of Utah

Sudhir Murthy
DC Water

JB Neethling
HDR Inc.

Amit Pramanik
The Water Research Foundation

Julian Sandino
CH2M Hill

David Stensel
University of Washington

Ryujiro Tsuchihashi
AECOM

Please use the following citation for this document:

WRF (The Water Research Foundation). 2019. "Deammonification" from the Nutrient Removal Challenge.

1199 North Fairfax Street, Suite 900
Alexandria, VA 22314-1445

6666 West Quincy Avenue
Denver, CO 80235-3098

info@waterrf.org
www.waterrf.org

TABLE OF CONTENTS

DEAMMONIFICATION

Deammonification has emerged as a cost effective, efficient, and reliable option to treat high-strength ammonia wastewater treatment streams, in particular to treat recycle streams from dewatering of anaerobic digested sludge. These recycles typically carry 20 to 25 percent of the total ammonia load in a treatment plant. The technology has been applied to treat the higher strength side streams at more than 50 full-scale facilities. Approximately 14 full-scale deammonification processes are currently (2019) in operation in the United States. These installations operate well and require a modest level of operator attention.

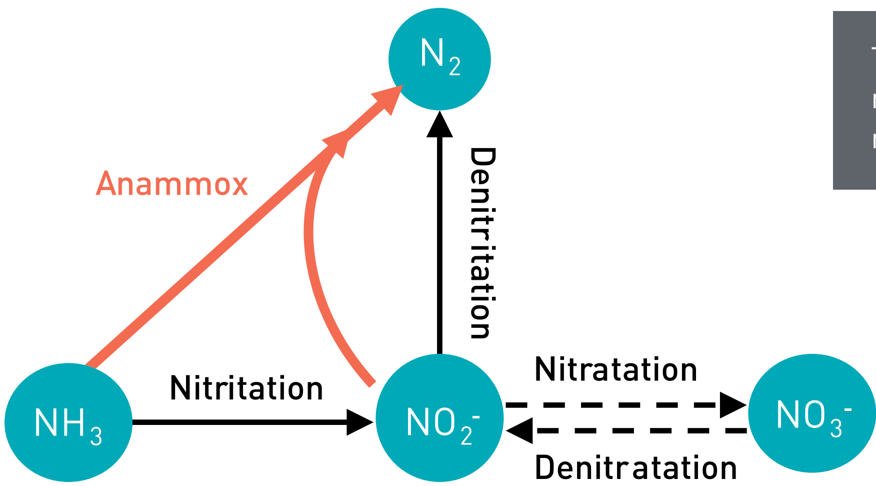
This nutrient management compendium document presents some of the key questions related to the deammonification process in a frequently asked question and answer format. It is intended to present an overview of deammonification, the technology process arrangements that are currently available, key factors that contribute to the success of the process, and potential challenges that need to be addressed in its implementation. The reference list included in this document is also intended to be a resource for those seeking further level of detail with regard to this technology.

What Is Deammonification?

Deammonification is a biological treatment process to convert ammonia to nitrogen gas. Deammonification is referred to as a shortcut nitrogen removal process because various steps of the more traditional nitrification/denitrification nitrogen removal process are bypassed or eliminated. Unlike nitrification/denitrification, deammonification does not require an organic carbon source to drive nitrogen removal.

Deammonification is accomplished by two biological process steps as shown in Figure 1. The first is termed nitritation, which is the aerobic oxidation of ammonia-N (NH_4-N) to nitrite-nitrogen (NO_2-N) by autotrophic aerobic ammonia oxidizing bacteria (AerAOB). Only about half of the ammonia needs to be converted to nitrite based on deammonification process stoichiometry; therefore, this step is more specifically referred to as partial nitritation. Nitritation is well known in wastewater treatment, as it is the initial step in biological nitrification of NH_4-N to nitrate-nitrogen (NO_3-N). The second step of deammonification is the anammox (anaerobic ammonia oxidation) reaction, in which NH_4-N is oxidized by anaerobic[1] ammonia-oxidizing bacteria (AnAOB)[2] that can use NO_2-N as the electron acceptor. AnAOB are also referred to as anammox bacteria. About 89 percent of the inorganic nitrogen (NH_4-N + NO_2-N) converted by AnAOB ends up as N_2 gas and about 11 percent as NO_3-N.

Figure 1 - Nitrogen Transformations



The deammonification nitrogen removal pathway includes partial nitritation and anammox.

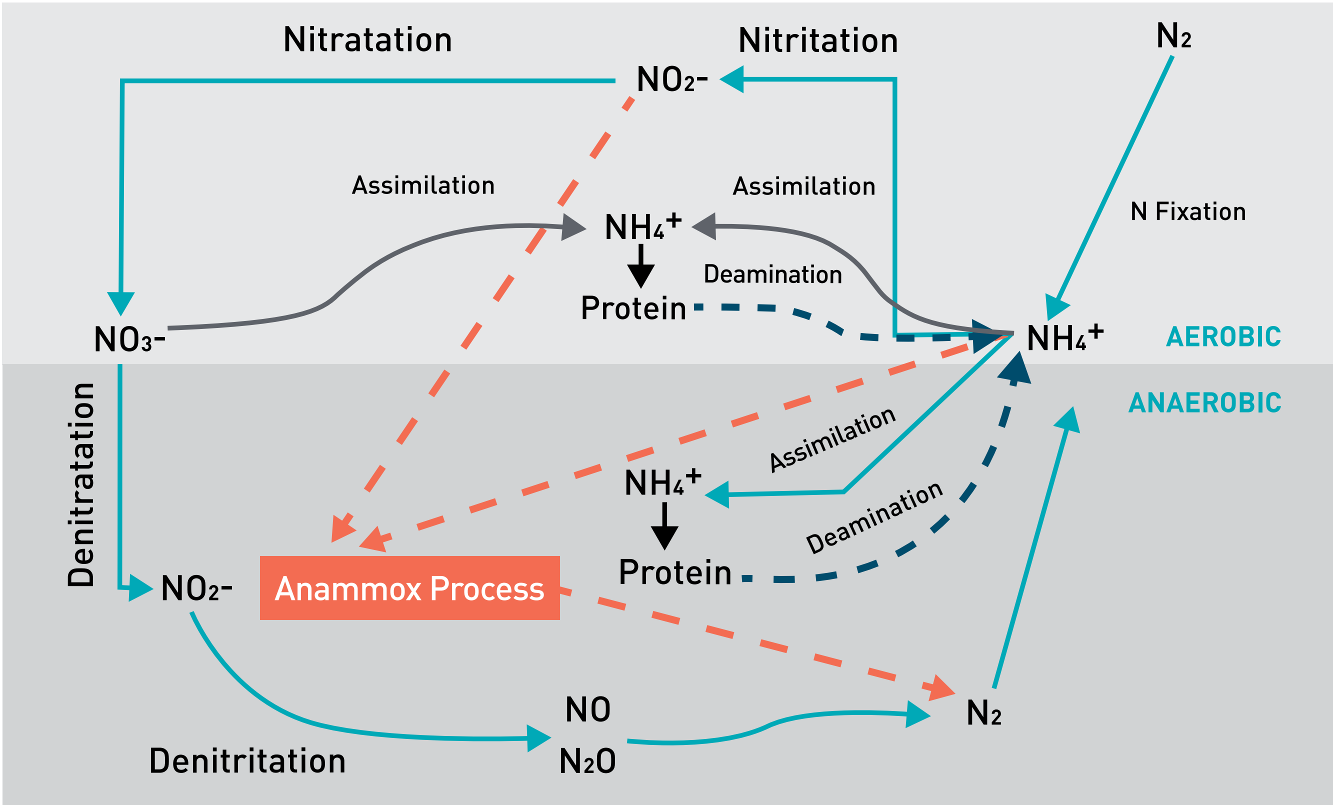
The anaerobic biological oxidation of NH_4-N was named the "Anammox" (An̲aerobic Amm̲onia Ox̲idation) process, by Mulder et al. (1995) in the first report of observations on anaerobic oxidation of ammonia. Follow-up research by Van de Graaf et al. (1995) confirmed that the NH_4-N conversion to nitrogen gas was accomplished with NO_2-N under anaerobic conditions and was biologically mediated. The bacteria responsible for the reaction was called the anammox bacteria and was first identified by Strous et al. (1999a) as being an autotrophic bacterium under the order Planctomycetales in the phylogenetic tree for bacteria. Since then, the anammox bacteria has been found to be abundant on the earth and frequently observed in biological wastewater treatment, and in marine and fresh water sediments (Kuenen 2008, Van Hulle et al. 2010). The discovery of the anammox bacteria helped to explain the cause of the "missing nitrogen" that some scientists have reported in the past when doing nitrogen mass balances on water bodies (Ward et al. 2011). The nitrogen cycle (Figure 2) now incorporates the anammox reaction.

Investigations of anammox bacteria using modern molecular tools has resulted in finding nine species within five genera as members of the bacteria order Planctomycetales (Ward et al. 2011). They are preceded by the name *Candidatus*, which is used when a species or genus is well characterized but not studied in pure culture. These are *Candidatus Kuenenia* spp., *Candidatus Brocadia* spp., *Candidatus Scalindua* spp., *Candidatus Jettenia* spp., and *Candidatus Anammoxogloubus* spp. The anammox bacteria found in wastewater treatment applications are species within

[1]In the field of microbiology, the term "anaerobic" broadly refers to an environment with zero dissolved oxygen. In environmental engineering and the wastewater treatment industry in particular, the term "anoxic" has been used to describe an environment in which there is no dissolved oxygen but nitrate and/or nitrite available as an electron acceptor. In spite of nitrite present in the anammox reaction, preference has been given to the term anaerobic ammonia-oxidizing bacteria.

[2]Anaerobic Ammonia-Oxidizing Bacteria (AnAOB) includes all autotrophic bacteria capable of catabolic oxidation of ammonia with nitrite to nitrogen gas. AnAOB may also be referred to as anammox bacteria.

Figure 2 - Nitrogen Cycle Including Anammox Reaction



Candidatus Kuenenia spp. and *Candidatus Brocadia* spp. (Kuenen 2008). Anammox bacteria enrichments develop a deep red color (Jetten et al. 1999) and are found in dense granular flocs in suspended sludge systems with a stable operation (Strous et al. 1999b). The floc characteristics provide an advantage for separating and maintaining them in suspended sludge treatment systems. Anammox bacteria also grow in attached-growth biofilms.

The deammonification process has been successfully demonstrated for sidestream treatment conditions at high temperature (30°C–35°C) and high ammonia concentrations in both fixed film and suspended growth biological reactors treating anaerobic digester dewatering recycle streams (Schmidt et al. 2003). Ongoing efforts seek to achieve nitrogen removal via deammonification in the mainstream treatment process at more challenging conditions with lower temperatures and more dilute NH_4-N concentrations.

DEAMMONIFICATION PROCESS

What Are the Benefits of a Deammonification Process?

Use of the deammonification process instead of the conventional nitrification/denitrification process for nitrogen removal results in remarkable savings:

- The aeration energy needed for the deammonification process is about 55 to 60 percent of that needed for the conventional nitrification/denitrification process.
- No carbon is needed for nitrogen removal by the deammonification process. If carbon is added to remove the nitrate produced from the anammox process, the total carbon demand is still about 90 percent less than that used in the conventional nitrification/denitrification process.
- The process also is a net consumer of CO_2, compared to a release of CO_2 from carbon oxidation by heterotrophic bacteria in the conventional nitrification/denitrification process.
- The alkalinity demand for ammonia oxidation is reduced by about 45 percent.
- Reduction in sludge production.

What Biochemical Reactions Are Involved in a Deammonification Process? How Do They Compare to the Conventional Biological Nitrification/Denitrification Process and to a Nitritation/Denitritation Process?

In the conventional nitrification/denitrification process, NH_4-N is first oxidized to NO_2-N and further to NO_3-N by autotrophic bacteria. Then the NO_3-N is biologically reduced to N_2 by heterotrophic bacteria with the consumption of organic substrate and absence of dissolved oxygen. In the nitritation/denitritation process the NH_4-N is oxidized to only NO_2-N, and then the NO_2-N is biologically reduced to N_2 by heterotrophic bacteria with the consumption of organic substrate and absence of dissolved oxygen.

The deammonification process requires aerobic nitritation of NH_4-N to NO_2-N, but only for about 55 percent of the feed NH_4-N. The remaining NH_4-N is anaerobically oxidized with NO_2-N to N_2 gas in the anammox process.

Aerobic biological nitrification is well known and is accomplished by Nitroso-bacteria for NH_4-N oxidation to NO_2-N and by Nitro-bacteria for NO_2-N oxidation to NO_3-N. These reactions are summarized below, accounting for ammonia used in the production of nitrifying bacteria biomass:

Equation 1: Nitritation by Aerobic Ammonia-oxidizing Bacteria

$$1.0\ NH_4^+ + 1.404\ O_2 + 0.0743\ HCO_3^- \rightarrow$$
$$\rightarrow 0.985\ NO_2^- + 0.0149\ C_5H_7O_2N + 1.911\ H^+ + 1.03\ H_2O$$

Equation 2: Nitratation by Aerobic Nitrite-oxidizing Bacteria

$$1.0\ NO_2^- + 0.473\ O_2 + 0.005\ NH_4^+ + 0.020\ CO_2 + 0.005\ HCO_3^- + 0.005\ H_2O \rightarrow$$
$$\rightarrow 1.0\ NO_3^- + 0.005\ C_5H_7O_2N$$

Equation 3: Total Oxidation Reaction of Ammonia to Nitrate for Conventional Nitrification

$$1.0\ NH_4^+ + 1.86\ O_2 + 0.02\ CO_2 + 0.079\ HCO_3^- \rightarrow$$
$$\rightarrow 0.981\ NO_3^- + 0.0197\ C_5H_7O_2N + 1.902\ H^+ + 1.02\ H_2O$$

Based on the above and considering nitrogen removed for cell synthesis, the oxygen required for complete oxidation of ammonia to NO_3-N (Equation 3) is 4.25 g O_2/g NH_4-N removed with 3.21 g O2/g NH_4-N used for nitrite production (Equation 1) and 1.08 g O2/g NO_2-N used for nitrite oxidation (Equation 2).

The second stage of the traditional method for biological nitrogen removal in wastewater treatment is biological denitrification, which is the reduction of NO_3-N to N_2. Note that denitrification is the combination of denitratation (reduction of NO_3-N to NO_2-N) and denitritation (reduction of NO_2-N to N_2) as shown in Figure 1 and Figure 2. In the absence of dissolved oxygen, heterotrophic bacteria can oxidize an organic substrate with NO_2-N or NO_3-N as the electron acceptor to reduce the oxidized nitrogen to N_2 gas. Environmental engineers have commonly used the term anoxic to describe the conditions in mixed, non-aerated reactor designs for the biological conversion of nitrite and nitrate to nitrogen gas, with oxygen absent and nitrate and/or nitrite as the main electron acceptors.

Examples of biological denitritation and denitrification stoichiometry with acetate consumption and heterotrophic biomass growth are shown below. The stoichiometric relationships are based on bioenergetics calculations to determine the synthesis cell yield of denitrifying bacteria using nitrate as the electron acceptor (Rittman and McCarty 2001). The same synthesis cell yield is used for nitrite reduction based on reports on the relative amount of substrate used for nitrite reduction compared to that for nitrate reduction (van Loosdrecht 2008).

Equation 4: Denitritation (Nitrite Reduction to N_2 by Heterotrophic Bacteria)

$$1.0\ NO_2^- + 1.0\ H^+ + 0.24\ NH_4^+ + 0.975\ CH_3COO^- \rightarrow$$
$$\rightarrow 0.5\ N_2 + 0.24\ C_5H_7O_2N + 0.015\ CO_2 + 0.735\ HCO_3^- + 1.235\ H_2O$$

Equation 5: Denitrification (Nitrate Reduction to N_2 by Heterotrophic Bacteria)

$$1.0\ NO_3^- + 1.0\ H^+ + 0.33\ NH_4^+ + 1.45\ CH_3COO^- \rightarrow$$
$$\rightarrow 0.5\ N_2 + 0.33\ C_5H_7O_2N + 0.13\ CO_2 + 1.12\ HCO_3^- + 1.62\ H_2O$$

Based on the previous equation, 6.6 g acetate COD is needed per g of NO_3-N denitrified (Equation 5). For denitritation of NO_2-N, about 30 percent less acetate is needed, at 4.5 g acetate COD per g of NO_2-N denitrified (Equation 4).

The anammox reaction is now included in the nitrogen cycle (Figure 1 and Figure 2), and, in addition to being the second key step in the deammonification process, it also occurs in the natural environment. Since the discovery of the anammox reaction in wastewater treatment, it has been frequently observed in lake and estuary sediments, in which NO_2-N is produced in an oxygen limited top aerobic sediment layer. The NO_2-N diffuses down to a contiguous anaerobic layer where anammox bacteria use the NO_2-N to oxidize NH_4-N (Ward et al. 2011). This type of condition can also be created in wastewater treatment biofilm and granular sludge suspended growth processes with dissolved oxygen control.

Anammox involves the energy yielding reaction of NH_4-N oxidation by NO_2-N and the uptake of CO_2 and nutrients by the autotrophic anammox bacteria for biomass growth. The overall reaction, accounting for cell synthesis, was put forth by Strous et al. (1998):

Equation 6

$$1.0\ NH_4^+ + 1.32\ NO_2^- + 0.066\ HCO_3^- + 0.13\ H^+ \rightarrow$$
$$\rightarrow 1.02\ N_2 + 0.26\ NO_3^- + 0.066\ CH_2O_{0.5}N_{0.15} + 2.03\ H_2O$$

As it can be seen from Equation 6, during the anaerobic oxidation of ammonia some nitrate is formed from nitrite, which may provide the reducing power for fixation of carbon dioxide (Schmidt et al. 2002). Equation 6 also indicates that the removal of 1.0 mole of NH_4-N requires 1.32 moles of NO_2-N and produces 0.26 moles of NO_3-N. The amount of NO_3-N produced accounts for 11.2 percent of the NH_4-N and NO_2-N metabolized. Research has found that some species of anammox bacteria are able to reduce NO_3-N with acetate, formate, and propionate (Gueven et al. 2005, Kartal et al. 2007, and Winkler et al. 2012).

Acid is decreased in the anammox reaction as shown by the 0.13 moles of H+ consumed and the removal of nitrous acid (HNO_2) with a smaller molar amount of nitric acid (HNO_3) production.

Combining the appropriate ratio of nitritation (Equation 1) and anammox (Equation 6) gives the overall deammonification reaction as:

Equation 7

$$1.0\ NH_4^+ + 0.804\ O_2 + 0.071\ HCO_3^- \rightarrow 0.436\ N_2 + 0.111\ NO_3^-$$
$$+ 0.009\ C_5H_7O_2N + 0.028\ CH_2O_{0.5}N_{0.15} + 1.038\ H^+ + 1.46\ H_2O$$

Table 1 shows the benefits of the deammonification process in terms of less biomass production, no exogenous carbon source requirement, and less aeration energy for oxygen supply in a comparison to the conventional nitrogen removal processes based on the above stoichiometric reactions with acetate as the exogenous carbon source for denitritation and denitrification.

Table 1 - Comparison of Nitrogen Removal Processes from Stoichiometric Calculations

Parameter	Deammonification	Nitritation/ 89% Denitritation	Nitrification/ 89% Denitrification
Oxygen Demand[a]	1.84 g O_2/ g NH_4-N Oxidized	3.42 g O_2/ g NH_4-N Oxidized	4.57 g O_2/ g NH_4-N Oxidized
Acetate-COD Demand	Not Required for N Removal[b]	4.5 g Acetate COD/ g NO_2-N Reduced	6.6 g Acetate COD/ g NO_3-N Reduced
Biomass Production	0.12 g Biomass VSS/ g NH_4-N Removed	1.5 g Biomass VSS/ g NH_4-N Removed	1.93 g Biomass VSS/g NH_4-N Removed

a Oxygen demand shown for ammonia oxidation only. Does not include carbonaceous oxygen demand or oxygen demand "credit" for influent BOD used for denitrification or denitritation.

b In absence of organic carbon, deammonification pathway produces 0.11 g NO_3-N per g NH_3-N removed. Consequently, the maximum N removal possible for deammonification is 89 percent in absence of organic carbon for denitrification of the residual NO_3-N. Removal of this residual NO_3-N requires 0.7 g acetate COD per g NH_3-N removed.

How Does the Growth Rate of Anammox Bacteria Compare to Aerobic Nitrifying Bacteria?

Most of the information on anammox bacteria kinetics has been obtained from operations at 30°C–35°C, which correspond to the typical sidestream temperatures from the dewatering of anaerobically digested sludge. Anammox growth at temperatures near 4°C and at 43°C has been observed with the lower temperature growth based on observations of anammox activity in Arctic environments (Ward at al. 2011). Anammox bacteria growth has been sustained in reactors at 15°C (Ward et al. 2011) and 18°C (Winkler et al. 2012).

The anammox bacteria have much slower growth rates compared to aerobic ammonia oxidizing bacteria (AerAOB) but also very slow decay rates. The growth rate of both of these bacteria is much lower than for heterotrophic bacteria. At 30°C the solids retention time (SRT) for the anammox bacteria needs to be 5-10 times longer than that for AerAOB (Jetten et al. 2001, Schmidt et al. 2003, and Van de Star et al. 2008). Nitrite is a key growth substrate for anammox but is also inhibitory at elevated concentration. On the plus side, compared to AerAOB the anammox bacteria have a much higher affinity for ammonia and nitrite as indicated by their very low half velocity coefficients of 0.07 to 0.10 mg/L (Strous et al. 1999b, Jetten et al. 2001) compared to about 0.50 mg/L for AerAOB.

The biomass yield for anammox bacteria is in the same range as that for AerAOB and much lower than for heterotrophic bacteria, as is typical for autotrophic bacteria with their energy needs for CO_2 fixation (Strous et al. 1999b, Schmid et al. 2003).

Nitrite accumulation in a deammonification reactor can lead to process inhibition or failure at high enough levels. The level of NO_2-N that can be tolerated varies between researchers, but a safe operation is with NO_2-N concentrations at 5.0 mg/L or less (Wett et al. 2010b). Strous et al. (1997) found that dissolved oxygen (DO) inhibition also occurs but is reversible, such that an intermittently aerated reactor for nitritation and deammonification is possible.

What Are the Essential Features of a Deammonification Process?

Deammonification relies on two different treatment processes: aerobic ammonia oxidation to nitrite and anaerobic ammonia oxidation to nitrogen gas with nitrite as electron acceptor. The deammonification process can be operated either as a two-stage reactor, such as SHARON-Anammox (nitritation-anammox), a single stage arrangement, also known as the CANON (completely autotrophic nitrogen removal over nitrite) process, or fixed film processes to retain the anammox bacteria.

Stable deammonification operation requires a long sludge age to sustain a large population of the slow growing anammox bacteria, control of the DO/oxidation-reduction potential (ORP)/pH, elimination of bacteria competing for nitrite (nitrite-oxidizing bacteria [NOB] primarily), and avoiding nitrite inhibition. The single-stage process requires solids retention (Vazquez-Padin et al. 2009), and both aerobic and anaerobic (oxygen-limiting) conditions in the reactor (Jetten et al. 2001). Irrespective of the mode of operation, the essential requirement for the successful operation of a deammonification process is the availability of nitrite and ammonia to sustain anammox bacteria growth. Nitrite and ammonia are consumed by anammox bacteria at the 1.3:1.0 ratio indicated in Equation 6 but do not need to be, and typically are not, present at that ratio in the reactor bulk liquid.

Competition for nitrite can inhibit the anammox bacteria by robbing anammox of an essential substrate. Competition for nitrite can come from NOB or heterotrophic denitritation bacteria. These competing bacteria appear in the reactor as a fluffy growth that can be removed by selective wasting from a clarifier or cyclone. Increased abundance of NOBs, which compete with anammox bacteria for NO_2–N, has been cited as a cause of instability in a single-stage deammonification system (Joss et al. 2011). Based on a comparison of parallel full-scale reactors, it was suggested, that an infrequent and short-term increase in O_2 supply (e.g., for maintenance of diffused aerators) that exceeded AerAOB oxygen demand and associated oxygen uptake rate may have caused increased NOB abundance.

Long start-up periods and lack of operational reliability have been a concern of deammonification technology. Transfer of sufficient seed sludge has proven to accelerate the start-up period down to about 50 days. Seeding with anammox bacteria greatly accelerates the start-up of a new deammonification process application. Strous et al. (1998) found that a critical bacteria concentration is necessary to realize good anammox activity. A possible reason is that for small anammox flocs the ammonia oxidation intermediate hydrazine diffuses quickly to the bulk liquid. The addition of hydroxylamine or hydrazine was found to accelerate anammox bacteria activity (Van Hulle et al. 2010). The slow decay rate of anammox allows the seed to remain active during storage and transportation.

DEAMMONIFICATION PROCESS TECHNOLOGIES

What Deammonification Process Technologies Are Used?

Several technical solutions have been developed to grow and support anammox bacteria. Table 2 shows a list of more than 50 installations from various technical solutions. Many different deammonification processes are commercially available. These processes differ in terms of the method to grow and retain the anammox bacteria, number of stages, the configuration of the process, and control strategies implemented. Configurations include granular sludge reactors, suspended growth sequencing batch reactors (SBRs), moving bed biofilm reactors, and rotating biological contactors.

Deammonification has demonstrated ammonia reduction of 90 to 95 percent and total nitrogen reduction of 80 to 85 percent.

The first full-scale facility was constructed in 2001. Since 2007, the number of installations rapidly increased with a large number in the design and construction phase; some operating on side-stream dewatering reject water and some industrial applications. The DEMON process has the most installations. The largest facilities include two 24,000 lb N/d ANAMMOX® facilities in China and a 27,000 lb N/d DEMON® facility at the DC Water Blue Plains Facility in Washington, DC. See Bowden et al. (2014) for a more detailed listing of deammonification facilities.

Table 2 - Installations and Performance of Deammonification Technologies (As of 2014)

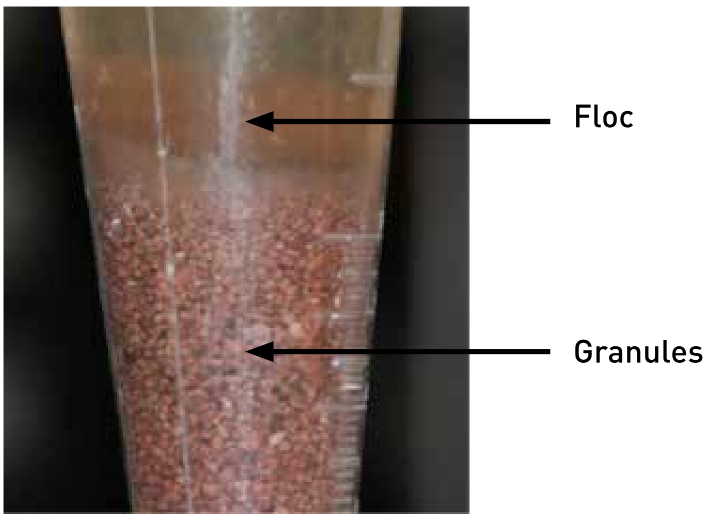
Technology Trade Name	Number[a]	Smallest lb N/d	Largest lb N/d	First Installation
ANAMMOX®	22	100	24,300	2002
ANITA™Mox	6	220	700	2010
DeAmmon®	3	260	4,900	2001
DEMON®	37	100	27,000	2004
Terra-N®	5	180	1,500	2008

[a] Facilities in operation and under design/construction

ANAMMOX®/AnammoPaq™ Granular Sludge Process

The anammox bacteria can be grown in granules typically greater than 1 mm in diameter that settle rapidly. Granular sludge reactors take advantage of the rapid settling rates of the anammox granules by using a proprietary compact upflow liquid-solids separator. The separator captures the anammox granules and retains them in the reactor to sustain the required long sludge age while flushing out the competing bacteria flocs (Figure 3). Two arrangements have been used: a two-step and a single-step arrangement. The ANAMMOX® process was developed by Paques in the Netherlands and is licensed by Ovivo and marketed as AnammoPaq™ for the U.S. municipal market.

Figure 3 - Anammox Granules Captured in Basin



Anammox granules must be retained in the basin and competing bacteria wasted to maintain a healthy operation.

Two-Step ANAMMOX® Process

The two-step ANAMMOX® process by Paques, uses the SHARON process (Hellinga et al. 1998) for partial nitritation in the first stage followed by a separate reactor where the anammox reaction occurs in a second stage (Figure 4). The first reactor is optimized to convert ammonia to nitrite by maintaining the proper pH, dissolved oxygen, and taking advantage of the higher growth rate of ammonia oxidizing bacteria at the higher temperatures found in dewatering return water (Van Loosdrecht and Jetten 1998). Anammox bacteria grow in granules and are retained in the second reactor using upflow clarification. Once the population is established, the nitrogen conversion capacity in the second stage anammox reactor is high (4.8 kg TN/m^3-d) because the second stage is not aerated and thus no longer practically limited by oxygen transfer equipment capabilities.

The two-step process was introduced in the Netherlands and implemented at a number of treatment plants (WWTP Dokhaven in Rotterdam). This process achieved 80 percent ammonia conversion to nitrogen gas at a loading rate of 1.2 kg N/m^3-d (van Dongen et al. 2001). Abma et al. (2007) report the performance of system receiving 1000-1300 mg NH_4-N/L to produce an effluent of 5-10 mg/L mg NO_2-N/L, 60-130 mg NH_4-N/L, and about 130 mg NO_3-N/L. The advantage of this arrangement is that the two main biological reactions (nitritation and deammonification) occur in separate reactors allowing better control of each process. The disadvantage of the arrangement is that the overall reactor volumes are significantly larger due to the large nitritation reactor.

Single-Step ANAMMOX® Process

In the single-step ANAMMOX® process nitritation and deammonification (anammox) occur in one basin. Process control becomes essential in this arrangement. It is essential to maintain the long SRT for anammox bacteria. Process control (pH, DO, and ORP) is used to prevent other bacteria competing for nitrite (heterotrophic denitritation and in particular autotrophic NOBs). AerAOB grow on the outer aerobic layer of the granules and anammox grow in the inner anoxic/anaerobic layer of the granules.

Figure 4 - Two-Step Nitritation/Anammox Process

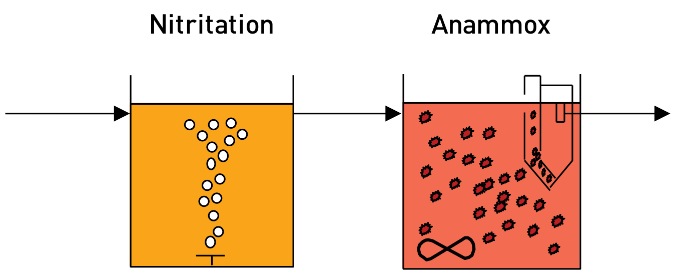
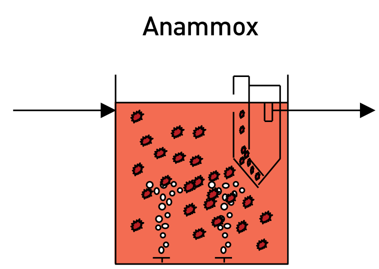


Figure 5 - Single-Step Anammox in Sludge Granular Reactor



The single-step process at Ohlburgen, The Netherlands, treats a high strength potato waste with about 300 mg/L NH4-N in the influent to the anammox process. It achieves 95 percent ammonia reduction and over 80 percent total nitrogen reduction. Effluent nitrate is 20-100 mg N/L (Abma et al. 2010). This process operated in a stable state by only controlling the aeration intensity.

The advantage of this technology is that the reactors are very compact due to the high concentrations of the granules and ability to maintain the entire process in a single reactor. The granular growth also protects the anammox if high nitrite concentrations should develop. The disadvantage is that the reactor must be operated and controlled to achieve stable nitritation as well as stable deammonification in one reactor. Operational experience with the system has been good. The single-step process is the most commonly used arrangement for the ANAMMOX® process. Fond du Lac, WI, has the only U.S. installation of a single-stage ANAMMOX®/AnammoPaq™ system.

DEMON® Hybrid Granular and Flocculent Activated Sludge Process

The DEMON® process (World Water Works, Inc.) is an acronym for DE-amMONnification and uses a suspended sludge process with mixture of flocculent and granular activated sludge. Anammox granules in the DEMON® process are smaller than those in the ANAMMOX® process where the upflow separator provides greater hydraulic selection pressure for larger granules. Nevertheless, anammox bacteria have a tendency to naturally form granules and are clearly present as red granules in the DEMON® process with AerAOB and other bacteria growing as flocculent activated sludge.

Prior to 2018, most DEMON® systems were designed as sequencing batch reactors (SBRs). The control of the DEMON® SBR process has been refined to address three elements: (1) Time control to allow the fill and draw operation of the SBR, (2) DO, and (3) pH control. DO is controlled to a low value (around 0.3 mg/L) to limit exposure of anammox bacteria to high DO concentrations, prevent rapid nitrite production by AOB, and repress NOBs. On-off aeration is used during the reaction period. Controlling the aeration frequency and duration regulates the pH: nitritation depresses the pH during the aeration period, while alkalinity-rich sidestream feeding increases the pH during the unaerated period. Aeration is initiated at the upper pH and stopped at the lower pH set point to maintain the pH within 0.01 units (Wett el al. 2007). Flat aeration panels are typically used in full floor coverage designs to minimize energy demand, though alternative designs are possible.

Figure 6 - Hydrocyclone

Hydrocyclone retains anammox and washes out competing bacteria.

Most DEMON® SBR systems included a hydrocyclone (Figure 6) to separate the granular anammox bacteria from the flocculent bacteria. The heavier anammox granules are returned to the reactor while the flocs are separated and wasted. Selective anammox retention and wasting of flocculent bacteria stabilizes the process and helps to limit NOBs growth.

As of 2018, seven DEMON® systems were in operation in the North America (Table 3). All of these systems were designed as SBRs with hydrocyclones.

Table 3 - DEMON® Installations in North America (As of 2019)

Location	Year	Nitrogen Load (lb N/d)
York River, VA	2012	480
Reedy Creek, FL	2013	1,550
Alexandria, VA	2015	3,970
Greeley, CO	2015	780
Guelph, Canada	2015	920
Pierce County, WA	2017	3,260
Washington, DC	2017	26,000

More recently after circa 2018, modifications to the DEMON® process design have been proposed by the vendor. In some instances, these modifications have been implemented at full-scale. The modifications involve approaches to liquid-solids separation and anammox retention as described below. The newer-generation DEMON® systems still use the core pH-based process control logic with on-off aeration.

The primary modification is moving from an SBR process to a flow-through process, which is achieved by adding a dedicated liquid-solids separation device. A lamella settler has been used in flow-through DEMON® systems in Europe. The vendor also has proposed designs with a quiescent settling zone built inside the reactor, but such designs have yet to be implemented at full scale. The continuous-flow process eliminates settling and decanting steps of SBR operation and allows higher design volumetric loading rates and smaller reactor volumes.

Another recent modification to the DEMON® process is replacing the hydrocyclone with rotary drum micro-screening for anammox retention. A 50 um mesh size is used at Strass, Austria. The micro-screen has a lower energy demand than the hydrocyclones and has been equally effective at retaining anammox.

The Strass, Austria DEMON® facility has operated in both SBR mode with cyclone and flow-through mode with screens. Nitrogen loadings rates in SBR mode were in the range of 0.7 to 1.0 kg TKN/m^3-d. Loading rates in flow-through mode with micro-screen have been up to 1.4 kg TKN/m^3-d with no loss of nitrogen removal efficiency. Common design loading rates are approximately 0.8 and 1.2 kg TKN/m^3-d for SBR and flow-through modes, respectively. Wett (2006) reported the specific energy demand for a DEMON® SBR without hydrocyclone was approximately 0.8 kWh/kg N removed. Specific energy demand in the range of 0.8 to 1.1 kWh/kg N removed is expected depending on system design, operating conditions, and sidestream characteristics.

The stability of the DEMON process depends on controlling the feed rate, the pH and the DO in the reactor to balance the nitritation and anammox reactions (O'Shaughnessy et al. 2008). The nitritation step of the DEMON process can be associated with inhibition symptoms, leading to process instability. Nitritation inhibition was observed at the Alexandria Sanitation Authority (ASA) system but not at a similar system in Strass, Austria. The vendor claims that this inhibition can be overcome purely by process control through adjustment of DO. Operating at higher DO concentrations, however, can have the effect of lowering the nitrogen removal efficiency of the system. The brief demonstration study at ASA verified the resilience of the anammox organisms when controlled by the DEMON process control system.

Jaroszynski and Oleszkiewicz (2011) compared the removal of total nitrogen from anaerobically digested sludge reject water by a fully autotrophic process in either one- or two-reactor systems.

In their study the two-reactor (SHARON) systems had a similar nitrogen removal rate to the one-reactor (DEMON) systems with evidence that the partial nitrification was a limiting step. They conclude that the partial nitrification of the two-reactor system may be improved by adjusting key parameters: the un-ionized form of the substrates, DO, or the SRT.

Weissenbacher et al. (2010) gave a quantitative description of the gaseous nitrogen and carbon emissions of a full-scale Strass wastewater treatment plant. Deammonification accounted for the net carbon sequestration of 0.16 g CO_2/g NO_2-N. Both nitrogen dioxide (NO_2) and nitric oxide (NO) were minor trace gases (<0.1 percent nitrogen output). However, in comparison, the nitrous oxide (N_2O) emission (1.3 percent nitrogen output) was significant. The global warming potential of the N_2O emissions from the DEMON were similar to those found in conventional simultaneous nitrification/denitrification systems. However, CO_2 emissions, therefore overall environmental effect, in the investigated system were significantly lower. This was the first time such an analysis has been performed on a DEMON system.

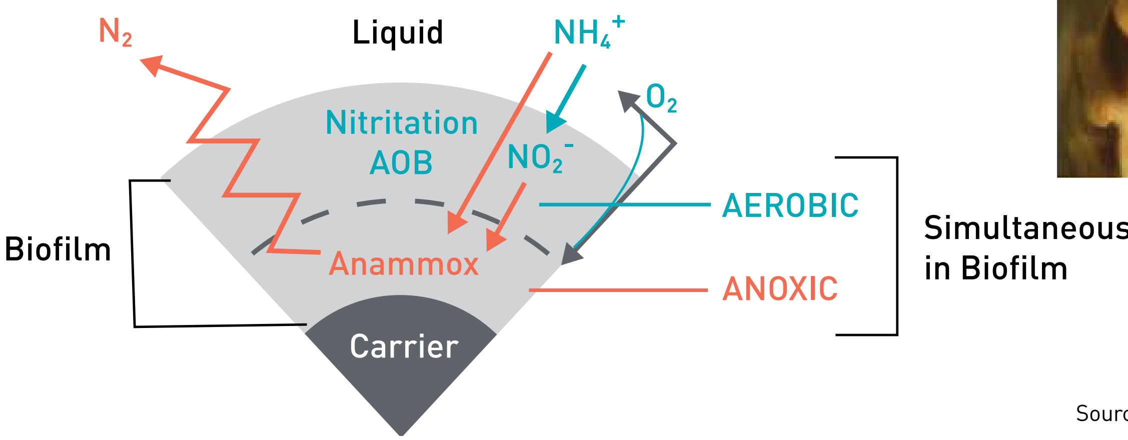
In a case study, Jardin and Hennerkes (2012) achieved start-up of a SBR (Plettenberg WWTP in 2007) within only 1 day by using seeding sludge from Strass WWTP. After stable operation for several months, increasing nitrate concentrations were observed in the effluent of the system indicating high activity of NOB. Several approaches were used to suppress NOB organisms in full-scale without success, e.g., low DO levels and high free ammonia concentrations. Finally, the reduction of the aerobic cycle length during intermittent aeration down to 8 min, followed by an anoxic mixing period of only 18 min was successful in inhibiting the activity of NOB organisms, most probably due to their elevated lag-phase compared with ammonium oxidizing bacteria. Key to a sustained inhibition on NOB is an appropriate selection of DO level, pH-value and cycle times for aerobic and anoxic phases. Nitrogen elimination was stabilized at more than 80 percent at a daily volumetric loading rate of 0.5 kg N/m^3/d with a total cost amounting to $3.02/kg N eliminated.

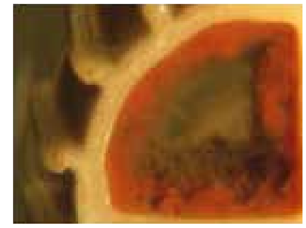
Moving-Bed Biofilm Reactor (MBBR) Processes

Shortly after the discovery of anammox bacteria, MBBR configurations were proposed for deammonification of high-strength, ammonium-rich plant recycle streams (Seyfried et al. 2001) using different types of support media.

As illustrated in Figure 7, AerAOB and anammox bacteria are established within the biofilm with AerAOB in the outer oxygen-penetrated layer of the biofilm and anammox in the inner anoxic/anaerobic portion of the biofilm closest to the carrier media. As shown in the photograph to the right, the biofilm tends to locate in protected regions of the support media.

Figure 7 - Deammonification in a Biofilm





Source: AnoxKaldnes/Veolia

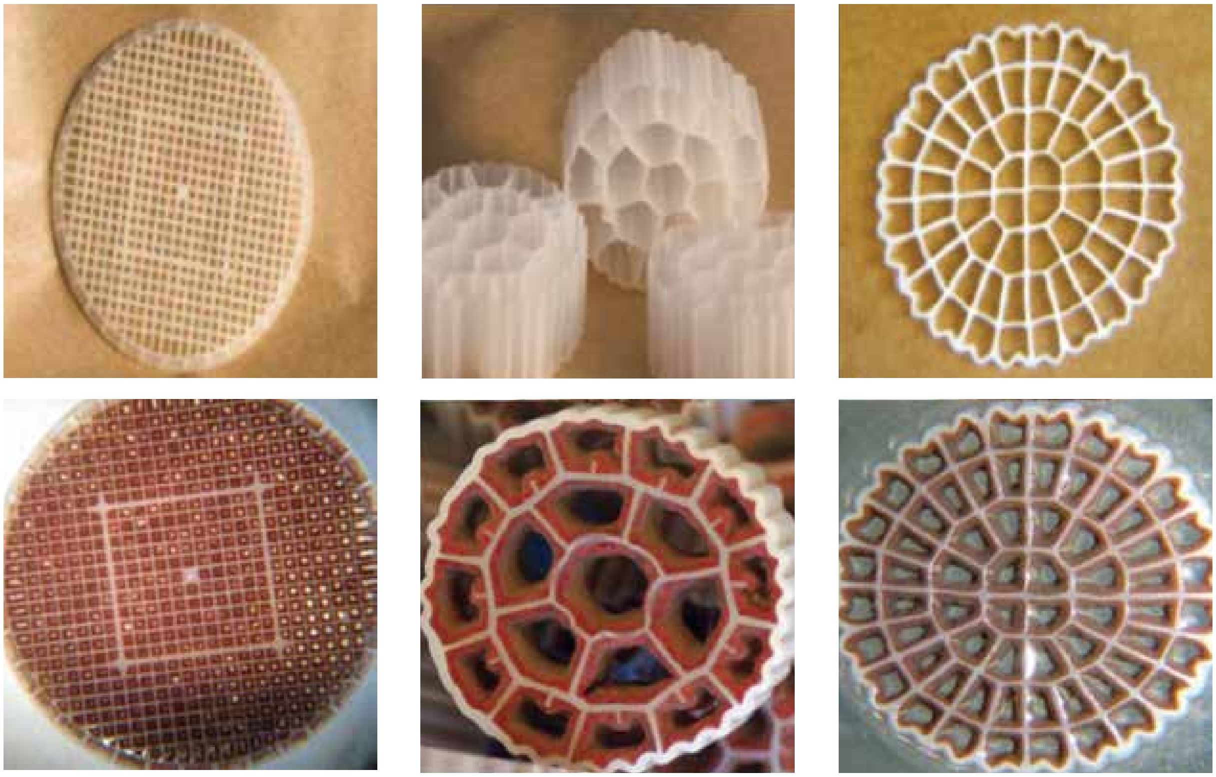
ANITA™Mox

A single-stage deammonification MBBR system under the trade name ANITA™Mox was developed by AnoxKaldnes/Veolia. Studies conducted with different types of polyethylene media of various geometries and specific surface areas (Figure 8) showed that the MBBR volumetric removal rate could be increased through the use of media with a higher specific surface area (Lemaire et. al. 2011). Using an AnoxKaldnes media under the trade name, BiofilmChip™M, which has a protected surface area for biofilm growth of 1,200 m^2/m^3, a volumetric ammonium-N removal rate up to 1.2 kg-N/m^3-day was demonstrated with a 40 percent media fill volume and a MBBR temperature range of 27°C to 30°C (Christensson et al. 2011). The corresponding specific surface area ammonium removal rate was approximately 3 g-N/m^2-d. Under these conditions, ammonium-N and total inorganic nitrogen removal efficiencies were approximately 90 percent and 80 percent, respectively. No pH control was applied resulting in a pH in the range of 6.7 to 7.5.

The ANITA™Mox MBBR is continuously aerated and the dissolved oxygen concentration controlled in the range of 0.5 to 1.5 mg/L. Medium to coarse bubble stainless steel air diffusers are used for aeration. The DO set point is adjusted based on online reactor ammonium and nitrate concentration data to ensure NOB growth is being restricted. Due to the mixing energy provided by continuous aeration, mechanical mixing may not be required, providing additional energy savings. An ANITA™Mox MBBR in Malmo, Sweden demonstrated an ammonia removal rate of 1.2 kg NH_3-N/m^3-d and specific energy demand of 1.5 kWh/kg N removed (Christensson et al. 2013). The same study reported that N_2O emissions were less than 1 percent of the nitrogen load removed.

Seeding the ANITA™Mox MBBR with a small fraction of media (2 to 3 percent of the media volume) with well-established anammox activity was found to significantly reduce the startup

Figure 8 - AnoxKaldnes Media Evaluated for ANITA™Mox



BiofilmChip™M (1,200 m²/m³) is on the left; K3 (500 m²/m³) in the middle; Anox™ K5 (800 m²/m³) on the right

Source: Christensson et. al. 2011

time from eight to ten months to one or two months (Thesing 2013). Feeding a new MBBR effluent from a fully active anammox MBBR was also found to reduce the startup time to four to five months.

As of 2018, six ANITA™Mox systems were in operation in North America (Table 4).

Table 4 - ANITA™Mox Installations in North America (As of 2019)

Location	Year
James River, Hampton Roads Sanitation District, Newport News VA	2014
Durham, NC	2015
Egan, IL	2016
Denver Metro, CO	2017
Howard County, MD	2018

DeAmmon®

The DeAmmon® MBBR process was developed by Purac/Läckeby AB (Sweden) in collaboration with the University of Hannover and the Ruhr River Association (Ruhrverband). The process consists of a single or dual train reactor system with three stages per reactor. The stages are operated in series, but piping flexibility is provided to allow parallel operation. Kaldnes (AnoxKaldnes/Veolia) K1 media (500 m^2 of active area per m^3) has been typically used to support biofilm growth and is added to each stage at a fill volume up to 40 percent. To retain the media within each stage, screens are provided between stages. Internal recirculation from the third to the first stage may be required for very high strength streams to limit the ammonia concentration in the first stage.

Aeration is provided through medium to coarse bubble stainless steel diffusers and each zone is intermittently aerated to support the partial nitritation and anammox reactions. Aeration and anoxic times are adjusted to limit nitrite accumulation in the bulk liquid and restrict the growth of aerobic nitrite oxidizing bacteria. Aeration and anoxic times of 20-50 minutes and 10-20 minutes, respectively, have been reported (Plaza et al. 2011, Thöle 2007). A DO concentration of 3 mg/L during the aeration periods is considered appropriate for design and operation, but higher concentrations are avoided to prevent the potential for NOB growth and to limit anammox inhibition. Mechanical mixers are provided to ensure well mixed conditions during non-aerated periods. Reactor pH is typically in the range of 7.3 to 7.7 and is not controlled. On-line conductivity measurements were found to be a low-cost effective tool for monitoring performance and making adjustments to the process operating conditions, e.g. duration of the aeration periods. An optimized energy consumption rate of 2.3 kWh/kg-N-removed was reported (Christensson et al. 2011).

An inorganic nitrogen removal efficiency in the range of 70 to 85 percent has been reported over a temperature range of 25°C to 30°C (Jardin et al. 2006, Plaza et al. 2011). The minimum operating temperature is 20°C where substantial deterioration in performance was observed (Thöle 2007). A design loading of approximately 0.6 kg-N/m^3-total-reactor-volume per day can be assumed with the K1 media.

Terra-N® Process

The Terra-N® MBBR process was developed by Clariant/SÜD-Chemie AG (Munich, Germany). Instead of plastic media, bentonite is used as the support media for biofilm growth. The process is designed as a single-stage SBR or as two-stages in series with gravity clarification. The bentonite product contains a wide range of particle sizes, with a mean in the range of 25 to 45 microns, and a mean surface area of 60 m^2/g, although the active surface area for biofilm growth is lower.

In the nitritation stage of the two-stage design or in the SBR, bentonite is added to a concentration of 10-12 g/L, resulting in a total suspended solids concentration of 15-20 g/L at full loading. Granulation of the anammox bacteria in the second stage of the two-stage system eliminates the need for a support media, although the addition of the media does not negatively impact performance. Second-stage biomass concentrations of 5-7 g/L have been reported (no media). Bentonite induces a very high solids settling rate and compact settled solids, resulting in insignificant loss of bentonite in the effluent. Biomass lost with the effluent via biofilm sloughing is sufficient to control the reactor solids concentration and no additional solids wastage should be necessary.

Fine bubble diffusers are typically used for aeration. In the SBR design, intermittent aeration is applied, and the aerobic/anoxic pattern is adjusted based on the ammonium loading to the system and the reactor performance. Mechanical mixing is required to maintain the solids in suspension during the anoxic phases of the SBR aeration cycle and in the anammox stage of a two-stage system.

Information on sidestream pretreatment as currently practiced in the existing full-scale systems is not available. Since the media may serve to capture and retain sidestream suspended solids, a sidestream total suspended solids (TSS) limit is likely, requiring a solids removal pretreatment step.

Many of the Terra-N® systems are retrofits of existing process tankage and operate at loading rates from 0.4 to 1.0 kg-N/m³-day (Clariant/SÜD Chemie 2012). With the SBR design, a loading rate up to 1.5 kg-N/m³-day is possible. Inorganic nitrogen removal efficiencies in the range of 80 to 90 percent have been reported (Clariant/SÜD Chemie 2012). Startup time for the SBR or the anammox stage of the two-stage process can be reduced to 60-90 days by seeding the reactors with anammox-enriched sludge from an existing full-scale system.

Integrated Fixed-film Activated Sludge (IFAS) Processes

Modifications to the Veolia ANITA™Mox MBBR process including liquid-solids separation and activated sludge return to allow IFAS operation have been studied by the vendor. Such IFAS ANITA™Mox systems are currently being designed for several full-scale facilities with thermal hydrolysis solids pretreatment as discussed below.

In contrast to the MBBR configuration where AerAOB and anammox grow together in the biofilm, Zhao et al. (2013) showed that AerAOB primarily grow as suspended sludge flocs while anammox grow as a biofilm on the attached-growth media in IFAS configuration. Loading rates approximately two times that of the MBBR configuration are also possible with IFAS configuration for the same influent sidestream wastewater composition (Zhao et al. 2014).

IMPACT OF THERMAL HYDROLYSIS SOLIDS PRETREATMENT

Is Sidestream Deammonification Impacted by Thermal Hydrolysis Solids Pretreatment?

Yes. Sidestreams from thermal hydrolysis process (THP) and anaerobic digestion solids treatment trains have different wastewater characteristics than typical sidestreams from conventional mesophilic anaerobic digestion. Compared to typical sidestreams from conventional mesophilic anaerobic digestion, sidestreams from THP digestion processes are characterized by higher NH_3-N and TKN concentration (approximately twice that of conventional sidestreams) and higher soluble COD concentration. Both biodegradable and non-biodegradable soluble COD concentrations are higher for THP sidestreams.

The THP sidestream characteristics impact deammonification in a number of ways. Due to the higher biodegradable COD concentration, THP sidestreams have higher potential for heterotrophic growth. This heterotrophic growth potential is problematic for certain sidestream deammonification configurations. For example, an MBBR biofilm would be adversely impacted by excess heterotrophic growth because heterotrophs may out-compete AerAOB for oxygen and space in the aerobic biofilm layer, thus reducing allowable design loading rates. In the case of THP sidestream treatment, an IFAS configuration allows for growth of AerAOB and heterotrophs in the flocculent sludge and anammox in the biofilm, thereby allowing the highest design loading rate possible for THP sidestream treatment with the attached-growth anammox process. Selective wasting of flocculent growth in the DEMON® hybrid flocculent and granular activated sludge process addresses this issue in a similar manner.

The THP process produce some compounds and/or particles that reduce the activity of AerAOB and anammox. This activity reduction may also be referred to as "inhibition" associated with THP sidestreams. A pilot study by Figdore et al. (2011) showed that AerAOB and anammox activities were lower in batch assays fed THP centrate than those fed conventional centrate. AerAOB were impacted to a greater extent than anammox. The pilot DEMON® reactor also sustained a lower loading rate near 0.6 kg N/m^3-d compared to the full-scale DEMON® reactor, which operated stably at loading rates as high as 1.2 kg N/m^3-d at the time of the pilot study. Zhang et al. (2018) provided deeper insight into the nature of THP-associated inhibition of AerAOB and anammox by evaluating impacts of different COD size fractions as well as biodegradable versus non-biodegradable COD. Their study used THP sidestreams generated by DC Water pilot and full-scale systems. Their results indicated the following:

- AerAOB inhibition is primarily attributed to indirect inhibition by large colloidal COD, which limits diffusion of substrates. This indirect inhibition can be minimized but not eliminated by optimizing dewatering polymer addition to reduce the filtrate colloidal COD concentration;

- To a lesser extent, AerAOB inhibition is due to direct inhibition from unspecified dissolved biodegradable COD compounds in the THP sidestream, with the exclusion of acetate; and
- AnAOB are directly and strongly inhibited by unspecified dissolved non-biodegradable COD compounds in the THP sidestream.

The results of Zhang et al. (2018) show that the fundamental causes of AerAOB and AnAOB inhibition from THP-AD sidestreams are complex and cannot be attributed to a single component of the sidestream COD.

Experience at pilot and full scale has shown that stable deammonification of THP sidestreams can be achieved with reductions in design loading rate and dilution water addition to reduce the extent of THP-associated inhibition. Treatment of THP filtrate in a DEMON® process at DC Water (Washington, DC) has been successful at loading rates near 0.6 kg NH_3-N/m^3-d with approximately 1:1 dilution of the filtrate, similar to the pilot operation described by Figdore et al. (2011). Lemaire et al. (2015) studied IFAS ANITA™Mox reactors treating THP sidestream and showed that specific removal rates near 4 g NH_3-N/m^2-d were sustained at lab and pilot scale with 1:3 dilution. A lower removal rate near 2.5 g NH_3-N/m^2-d was achieved at 1:2 dilution indicating a greater degree of partial inhibition with lower dilution. The corresponding volumetric nitrogen removal rate in the IFAS ANITA™Mox was approximately 1.3 kg NH_3-N/m^3-d considering the fill ratio (42 percent), media specific surface area (800 m^2/m^3) and specific nitrogen removal rate at 1:3 dilution. A THP sidestream in Vaxjo, Sweden recently converted from MBBR to IFAS ANITA™Mox configuration, and several other IFAS ANITA™Mox systems treating THP sidestream will be commissioned in 2019.

DEAMMONIFICATION COSTS AND SAVINGS

What Are the Costs and Cost Savings for Deammonification?

Deammonification reduces the cost for nitrogen removal. Construction and operating costs are very site and technology specific. Lux and Siegrist (2004) found that the construction costs may be similar to conventional solutions.

There are significant opportunities to reduce operational cost when comparing a conventional nitrification/denitrification and anammox process. It should be noted that the proprietary process suppliers usually charge a license fee, and the fee is often determined based on the estimated operational cost savings by installing the anammox-based process.

- Aeration energy cost is reduced significantly. Nitritation is required for less than a half of the influent ammonia. Compared to conventional nitrification/denitrification process, the stoichiometric oxygen requirement is approximately 60 percent less. Savings in aeration energy cost depends on aeration equipment and process configuration.
- Carbon addition is eliminated, even though external carbon may be used in a smaller amount to trim the effluent nitrogen by eliminating nitrate produced during the deammonification process. Approximately 11 percent of ammonia entering the deammonification process leaves as nitrate (see Equation 7).
- Alkalinity supplements may be reduced since only partial nitritation is required (approximately 50 percent of the ammonia). The need for supplemental alkalinity in alternative configurations depends on the extent of alkalinity recovered through denitrification and extent that recovered alkalinity is recycled to the nitrification step.
- Solids management costs are reduced because of significantly lower biomass production.

FULL-SCALE DEAMMONIFICATION PILOT STUDIES

Are Pilot Studies Required to Successfully Implement Full-scale Deammonification?

The need for pilot and other studies depends on several considerations. Since the composition of sidestream process return water at municipal wastewater treatment facilities with conventional mesophilic anaerobic digestion is relatively consistent and water quality parameters can be easily measured with common laboratory techniques, pilot studies are not essential to prove treatability. Leading commercial processes have been demonstrated at numerous full-scale facilities worldwide and have sufficient installations to establish suitable design criteria and test the reliability of control systems for implementing the process at new facilities. Other emerging technologies would benefit from pilot testing to demonstrate the control strategy and ability to provide stable, consistent operation.

Full-scale deammonification installations treating thermophilic or thermal hydrolysis sidestreams are limited, but pilot and full-scale experience thus far indicates that piloting is not necessary if appropriate design modifications are made. The number of deammonification facilities treating such sidestreams is increasing with additional facilities expected to come online in 2019 and beyond.

MAINSTREAM DEAMMONIFICATION

Applying deammonification for mainstream nitrogen removal has received significant industry attention over the last decade. Conceptually, mainstream deammonification allows influent carbon to be diverted to anaerobic digestion for energy generation because it is no longer needed as a carbon source for denitrification in conventional nitrification-denitrification processes. Together, the increased energy generation from carbon diversion and decreased aeration demand from mainstream deammonification in lieu of nitrification-denitrification can increase WRRF energy self-sufficiency.

The reader is referred to two publications (WEF and WERF 2015, O'Shaughnessy 2015) that address strategies, process configurations, and data from facilities operating for mainstream deammonification at various scales from bench to full scale.

A key challenge in achieving mainstream short-cut nitrogen removal by mainstream deammonification and/or nitrite shut is repressing or preventing NOB growth. NOB compete with anammox for NO_2-N consumption. Thus, limiting or ideally preventing NOB growth through NOB out-selection is critical and required for successful mainstream deammonification or short-cut nitrogen removal. Strategies for NOB out-selection have arisen from a broad body of research. A comprehensive set of NOB out-selection strategies were summarized by O'Shaughnessy (2015) and include the following:

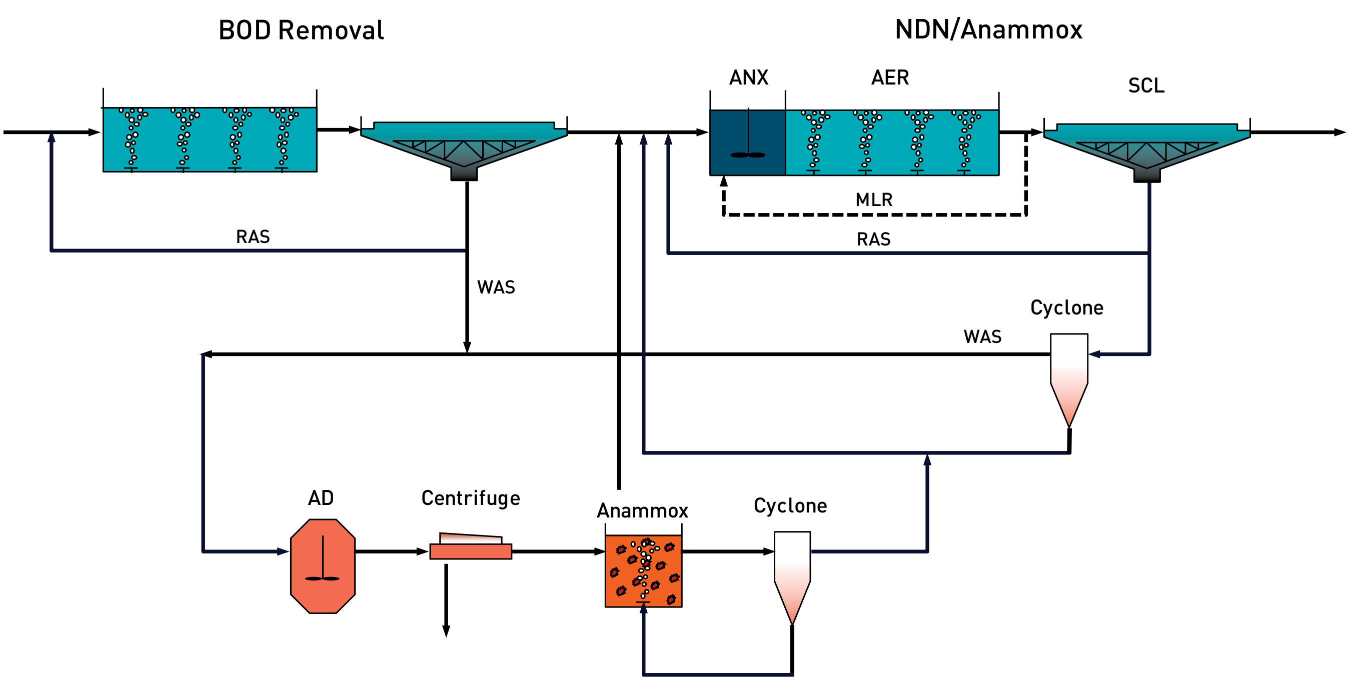
- **Residual NH_3-N (>2 mg/L):** Maintaining elevated NH_3-N concentrations in aerobic zones sustains high AerAOB activity rates and pressure NOB to compete for dissolved oxygen (DO). Allowing for residual NH_3-N also enables lower operating SRTs, a related strategy for NOB out-selection. A polishing step may be required to reduce NH_3-N concentrations at plants with lower NH_3-N limits.
- **High DO Concentrations (>1.5 mg/L):** Compared to low DO concentrations less than 0.5 mg/L, conventional DO concentrations near 2 mg/L maintain high AerAOB activity and are more favorable for NOB out-selection based on oxygen affinities of AerAOB and NOB.
- **Sufficient Alkalinity:** Low HCO_3^- limits AerAOB activity, and sufficient alkalinity must be available to maintain high AerAOB rates.
- **Transient Anoxia with COD Pressure:** Rapid transition from aerobic to anoxic conditions at the end of ammonia oxidation deprives NOB of DO when NO_2-N is available, thereby limiting NOB growth. The availability of COD under transient anoxic conditions enhances this phenomenon by driving DO consumption and NO_2-N reduction by heterotrophs. NOB have also shown a lag in activity compared to AOB in response to transient anoxia, which has been claimed to be caused by inactivation and delayed reactivation of enzymes for aerobic NO_2-N oxidation.
- **Limiting Aerobic SRT:** The objective of limiting aerobic SRT is to allow AerAOB growth while limiting NOB growth or preferably washing-out NOB. Strategies 1 to 4 above promote high

AerAOB growth rates and pressure NOB to lower their growth rate. Operating close to the AerAOB wash-out SRT and with other pressures on NOB are required to eliminate NOB but allow AerAOB growth.

- **NH_3-N Oxidation Control:** As can be inferred from Strategy 1, the extent of NH_3-N oxidation must be controlled for mainstream deammonification or short-cut nitrogen removal. Two control concepts have been developed in this regard. The first is ammonia-based aeration control (ABAC), which uses online NH_3-N measurements to control aeration setpoints including DO concentration and/or aeration timing. The second is ammonia versus NOx (AVN) control, which uses online NH_3-N and NO_2-N measurements to control aeration in a similar manner to produce a 1:1 blend of NH_3-N and NO_2-N well-suited for removal by anammox bacteria downstream of AVN control.
- **Bioaugmentation:** Adding anammox from sidestream treatment provides a seed source for mainstream anammox growth and competition for NO_2-N during anoxia. Selectively retaining anammox granules over flocculent sludge in mainstream treatment separates the SRTs and anammox and flocculent sludge, allowing for long SRTs necessary to sustain anammox growth at mainstream conditions. Addition of flocculent or possibly granular AerAOB from sidestream treatment further enables lower mainstream operating aerobic SRT to put pressure on NOB.

The NOB out-selection strategies can be used in various combinations and reactor configurations to operate for mainstream shortcut nitrogen removal including mainstream deammonification. Refer to WEF and WERF (2015) and O'Shaughnessy (2015) for example configurations. One frequently cited example is Strauss, Austria (Figure 9) where side stream anammox bacteria from a DEMON®

Figure 9 - Strauss, Austria Full-plant Deammonification Process Flow Sheet



Source: Adapted from Wett et al. 2010a

process are seeded to the second stage of a mainstream A-B process. The A stage is a high-rate BOD removal stage, and the B stage is operated for mainstream short-cut nitrogen removal. Once seeded to the mainstream, the anammox bacteria are selectively retained. Hydrocyclones were initially used for anammox retention but have been replaced with micro-screens. Operations for mainstream deammonification at Strauss have shown that some NOB out-selection can be achieved at cold winter process temperatures near 11°C (O'Shaughnessy 2015). Data suggest that a mixture of nitrification-denitrification, nitritation-denitrification, and deammonification occur in the mainstream.

Incorporating anammox in the mainstream process is still an emerging process. Results from pilot and full-scale studies show promise. The ability to achieve low effluent nitrogen concentrations at temperatures less than 20°C in a process where deammonification is the primary nitrogen removal pathway has yet to be demonstrated for municipal wastewater treatment. Design strategies providing flexibility to operate for mainstream deammonification as well as conventional mainstream nitrification-denitrification are recommended in absence of long-term pilot and/or full-scale demonstrating testing.

SUMMARY AND CONCLUSION

Deammonification of high-strength dewatering return flows has been shown to be technically and economically feasible. Several technology options have been tested and commercialized to provide the appropriate environment and control to provide stable operation. More than 50 full-size deammonification units have been constructed since the first full-scale facility was constructed in 2001. Leading commercial technologies have been in stable operation for many years; others are still emerging with few full-scale installations.

Commercial technologies use different approaches to accumulate sufficient slow-growing anammox bacteria needed in the process. These processes differ in terms of the method to grow and retain the anammox bacteria, number of stages, process configuration, and control strategies implemented. Configurations include granular sludge reactors, suspended growth SBRs, intermittently-aerated flow-through reactors, moving bed biofilm reactors, integrated fixed-film activated sludge reactors, and rotating biological contactors. Process control strategies vary, with most technologies relying on aeration control and online pH measurements to sustain stable nitritation (conversion of ammonia to nitrite) and subsequent deammonification (ammonia oxidation using nitrite as electron acceptor).

Regardless of the technology, anammox seed is essential to start up the deammonification process. The first-generation SBR, MBBR, and granulation processes required up to two years to achieve full loading. The knowledge that has been gained through the startup of these initial full-scale processes and the use of anammox-enriched biomass from existing facilities has greatly reduced the startup time. In addition, as the cumulative experience with these technologies within the wastewater community continues to grow, these processes are becoming increasingly recognized as reliable processes that can provide a consistent treated effluent quality and deliver the promised energy and chemical savings.

REFERENCES

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

Abma, W.R., W. Driessen, R. Haarhuis, and M.C.M. van Loosdrecht. 2010. Upgrading of Sewage Treatment Plant by Sustainable and Cost-Effective Separate Treatment of Industrial Wastewater. *Water Sci. Technol.,* 61(7), 1715-1722.

Bowdin, G., H.D. Stensel, and R. Tsuchihashi. 2015. *Technologies for Sidestream Nitrogen Removal*. Project NUTR1R06w/1524. Alexandria, VA: Water Environment Research Foundation, and London: IWA Publishing.

Cema, G., J. Wiszniowski, S. Żabczyński, E. Zabłocka-Godlewska, A. Raszka, and J. Surmacz-Górska. 2007. Biological Nitrogen Removal from Landfill Leachate by Deammonification Assisted by Heterotrophic Denitrification in a Rotating Biological Contactor (RBC). *Water Sci. Technol.*, 55(8-9), 35-42.

Christensson, M., S. Ekström, R. Lemaire, E. Le Vaillant, E. Bundggaard, J. Chauzy, L. Stålhandske, Z. Hong, and M. Ekenberg. 2011. ANITA™Mox—A Biofarm Solution for Fast Start-up of Deammonifying MBBRs. In *Proceedings of the Water Environment Federation 84th Annual Conference and Exposition*, Los Angeles, CA.

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

Clariant/SÜD Chemie. 2012. Terra-N performance data, facilities list, and operating conditions provided to author by personal correspondence.

Egli, K., U. Fanger, P.J.J. Alvarez, H. Siegrist, J.R. van der Meer, and A.J.B. Zehnder. 2001. Enrichment and Characterization of an Anammox Bacterium from a Rotating Biological Contactor Treating Ammonium-Rich Leachate. *Archives of Microbiology*, 175(3), 198-207.

Figdore, B., G. Bowden, B. Stinson, B. Wett, M. Hell, W. Bailey, J. Carr, R. Derminassian, and S. Murthy. 2011. Treatment of Dewatering Sidestream from a Thermal Hydrolysis-Mesophilic Anaerobic Digestion Process with a Single-Sludge Deammonification Process. In *Proceedings of the Water Environment Federation Technical Exposition and Conference* (WEFTEC), 249-264.

Fux, C., and H. Siegrist. 2004. Nitrogen Removal from Sludge Digester Liquids by Nitrification/Denitrification or Partial Nitritation/Anammox: Environmental and Economical Considerations. *Water Sci. Technol.*, 50(10), 19-27.

Gao, D.W., Y.Z. Peng, and W.M. Wu. 2010. Kinetic Model for Biological Nitrogen Removal Using Shortcut Nitrification–Denitrification Process in Sequencing Batch Reactor. *Environ. Sci. Technol.*, 44, 5015–5021.

Gueven, D., A. Dapena, B. Kartal, M.C. Schmid, B. Maas, K. van de Pas-Schoonen, S. Sozen, R. Mendez, H.J.M. Op den Camp, M.S.M. Jetten, M. Strous, and I. Schmidt. 2005. Propionate Oxidation by and Methanol Inhibition of Anaerobic Ammonium-Oxidizing Bacteria. *Appl. Environ. Microbiol.*, 71(2), 1066-1071.

Hellinga, C., A.A.J.C. Schellen, J.W. Mulder, M.C.M. van Loosdrecht, and J.J. Heijnen. 2001. The SHARON Process: An Innovative Method for Nitrogen Removal from Ammonium-Rich Waste Water. *Water Sci. Technol.*, 43(11), 127–134.

Jardin, N., D. Thöle, and B. Wett. 2006. Treatment of Sludge Return Liquors: Experiences from the Operation of Full-Scale Plants. In *Proceedings of the 2006 Water Environment Federation Technical Exposition and Conference* (WEFTEC), pp. 5237–5255.

Jardin, N., and J. Hennerkes. 2012. Full-Scale Experience with the Deammonification Process to Treat High Strength Sludge Water—A Case Study. *Water Sci Technol.*, 65(3), 447-55.

Jaroszynski, L.W., and J.A. Oleszkiewicz. 2011. Autotrophic Ammonium Removal from Reject Water: Partial Nitrification and Anammox in One-Reactor versus Two-Reactor System. *Environ. Technol.*, 32(3-4), 289-94.

Jetten, M.S.M, M. Wagner, J. Fuerst, M. van Loosdrecht, G. Kuenen, and M. Strous. 2001. Microbiology and Application of the Anaerobic Ammonium Oxidation ('Anammox') Process. *Current Opinion In Biotech.*, 12(3), 283-288.

Jetten, M.S.M., M. Strous, K.T. van de Pas-Schooen, J. Schalk, U.G.J.M. van Dongen, A.A. van de Graaf, S. Logemann, G. Muyzer, M.C.M. van Loosdrecht, and J.G. Kuenen. 1999. The Anaerobic Oxidation of Ammonium. *FEMS Microbiology Reviews*, 22(5), 421–437.

Jin, R.-C., G.-F. Yang, J. Yu, and P. Zheng. 2012. The Inhibition of the Anammox Process: A Review. *The Chemical Engineering Journal*, 197, 67-79.

Joss, A., N. Derlon, C. Cyprien, S. Burger, I. Szivák, J. Traber, H. Siegrist, and E. Morgenroth. 2011. Combined Nitritation–Anammox: Advances in Understanding Process Stability. *Environ. Sci. Technol.*, 45(22), 9735-9742.

Kartal, B., M.M.M. Kuypers, G. Lavik, E.J. Schalk, H.J.M. Op den Camp, M.S.M. Jetten, and M. Strous. 2007. Anammox Bacteria Disguised as Denitrifiers: Nitrate Reduction to Dinitrogen Gas via Nitrite and Ammonium. *Environ. Microbiol*, 9(3), 635-642.

Knox, K. 1991. Full-Scale Nitrification of a Methanogenic Leachate Using Rotating Biological Contactor and Vegetated Ditch. Presented at 14th Annual Madison Waste Conference, University of Wisconsin at Madison, USA.

Kuenen, J.G. 2008. Timeline: Anammox Bacteria: From Discovery to Application. *Nature Rev. Microb*., 6(4), 320-326.

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

Lemaire, R., S. Ekström, M. Christensson, E. LeVaillant, and J.C. Schrotter. 2012. Experience from Start-up and Operation of ANITA Mox MBBR Plants, IWA Leading Edge Technology, June 2012, Brisbane, Australia.

Lemaire, R., F. Veuillet, A. Bauerson, S. Chastrusse, R. Monnier, M. Christensson, H. Zhao, C. Thomson, and J. Ochoa. 2015. ANITA™ Mox Deammonification Process for COD-rich and THP Reject Water. In *Proceedings of the Water Environment Federation Technical Exposition and Conference* (WEFTEC), 3266-3279.

Mulder, A., A.A. van de Graaf, L.A. Robertson, and J.G. Kuenen.1995. Anaerobic Ammonium Oxidation Discovered in a Denitrifying Fluidized Bed Reactor. *FEMS Microbiol. Ecol.*, 16(3), 177–184.

O'Shaughnessy, M., J. Sizemore, M. Musabyimana, P. Sanjines, S. Murthy, B. Wett, I. Takács, D. Houweling, N.G. Love, and K. Pallansch. 2008. Operations and Process Control of the Deammonification (DEMON) Process as a Sidestream Option for Nutrient Removal. In *Proceedings of the 2008 Water Environment Federation Technical Exposition and Conference* (WEFTEC): Session 81-90, pp. 6333-6348(16).

O'Shaughnessy, M. 2015. *Mainstream Deammonification*. Project INFR6R11/1469. Alexandria, VA: Water Environment Research Foundation, and London: IWA Publishing.

Plaza, E., S. Stridh, J. Örnmark, L. Kanders, and J. Trela, Swedish Experience of the Deammonification Process in a Biofilm System. In *Proceedings of the Nutrient Recovery and Management 2011 Conference*, Miami, FL.

Pynaert, K., S. Wyffels, R. Sprengers, P. Boeckx, O. Van Cleemput, and W. Verstraete. 2002. Oxygen Limited Nitrogen Removal in a Lab Scale Rotating Biological Contactor Treating an Ammonium-Rich Wastewater. *Water Sci. Technol.*, 45(10), 357-363.

Rittman, B.E., and P.L. McCarty. 2001. *Environmental Biotechnology: Principles and Applications*. New York: McGraw-Hill.

Schmidt, M., K. Walsh, R. Webb, W.I.C. Rijpstra, K. van de Pas-Schoonen, M.J. Verbruggen, T. Hill, B. Moffett, J. Fuerst, S. Schouten, J.S. Sinninghe Damsté, J. Harris, P. Shaw, M. Jetten, and M. Strous. 2003. *Candidatus* 'Scalindua brodae', sp. nov., *Candidatus* 'Scalindua wagneri', sp. nov., Two New Species of Anaerobic Ammonium Oxidizing Bacteria. *Systematic and Appl. Microbiol.*, 26 4), 529–538.

Schmidt, I., C. Hermelink, K. van de Pas-Schoonen, M. Strous, H.J. Op den Camp, J.G. Kuenen, and M.S.M. Jetten. 2002. Anaerobic Ammonia Oxidation in the Presence of Nitrogen Oxides (NOx) by Two Different Lithotrophs. *Appl. Environ. Microbiol.*, 68(11), 5351-5357.

Schmidt, I., O. Sliekers, M. Schmid, E. Bock, J. Fuerst, J.G. Kuenen, M.S.M. Jetten, and M. Strous. 2003. New Concepts Of Microbial Treatment Processes for the Nitrogen Removal in Wastewater. *FEMS Microbiology Review.*, 27(4), 481-492.

Seyfried, C., A. Hippen, C. Helmer, S. Kunst, and K.H. Rosenwinkel. 2001. One-Stage Deammonification: Nitrogen Elimination at Low Costs. *Water Sci. Technol.: Water Supply,* 1(1), 71-80.

Shannon, M.A., P.W. Bohn, M. Elimelech, J.G. Georgiadis, B.J. Marinas, and A.M. Mayes. 2008. Science and Technology for Water Purification in the Coming Decades. *Nature*, 452, 301–310.

Siegrist, H., S. Reithaar, and P. Lais. 1998. Nitrogen Loss in a Nitrifying Rotating Contactor Treating Ammonium Rich Leachate without Organic Carbon. *Water Sci. Technol.*, 37(4-5), 589-591.

Sliekers, A., K.A. Third, W. Abma, J.G. Kuenen, and M.S.M. Jetten. 2003. CANON and Anammox in a Gas-Lift Reactor. *FEMS Microbiol. Lett.*, 218, 339–344.

Strous M., J.J. Heijnen, J.G. Kuenen, and M.S.M. Jetten. 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., J.A. Fuerst, E.H.M. Kramer, S. Logemann, G. Muyzert, K.T. van de Pas-Schoonen, R. Webb, J.G. Kuenen, and M.S.M. Jetten. 1999a. Missing Lithotroph Identified as New Planctomycete. *Nature*, 400, 6743.

Strous, M., J.G. Kuenen, and M.S.M. Jetten. 1999b. Key Physiology of Anaerobic Ammonium Oxidation. *Applied and Environmental Microbiology*, 65(7), 3248–3250.

Strous, M., E. Van Gerven, J.G. Kuenen, and M.S.M. Jetten. 1997. Effects of Aerobic and Microaerobic Conditions on Anaerobic Ammonium-oxidising (Anammox) Sludge. *Appl. Environ. Microbiol.*, 63(6), 2446–2448.

Tao, Y., D.-W. Gao, Y. Fu, W.-M. Wu, and N.-Q. Ren. 2012. Impact of Reactor Configuration on Anammox Process Start-up: MBR versus SBR. *Bioresource Technology*, 104, 73-80.

Thöle, D. 2007. Presentation on MBBR System Startup and Performance. Dieter Thöle, Department of Technical Management and Waste Management, Ruhrverband; Hattingen WWTP, December 2007.

Thesing, G. 2013. Personal communication, Veolia.

van de Graf, A.A., A. Mulder, P. De Bruijn, M.S.M. Jetten, L.A. Robertson, and J.G. Kuenen. 1995. Anaerobic Oxidation of Ammonium Is a Biologically Mediated Process. *Appl. and Environ. Microbiol.*, 61(4), 1246–1251.

van der Star, W.R.L., A.I. Miclea, U.G.J.M. van Dongen, G. Muyzer, C. Picioreanu, and M.C.M. van Loosdrecht. 2008. The Membrane Bioreactor: A Novel Tool to Grow Anammox Bacteria as Free Cells. *Biotechnol. Bioeng.*, 101(2), 286–294.

van Dongen, U., M.S.M. Jetten, and M.C.M. van Loosdrecht. 2001. The SHARON®-Anammox® Process for Treatment of Ammonium Rich Wastewater. *Water Sci. Technol.*, 44(44), 153-160.

Van Hulle, S.W.H., H.J.P. Vandeweyer, B.D. Meesschaert, P.A. Vanrolleghem, P. Dejans, and A. Dumoulin. 2010. Engineering Aspects and Practical Application of Autotrophic Nitrogen Removal from Nitrogen Rich Streams. *Chemical Engr. Jour.*, 162(1), 1-20.

Van Loosdrecht, M.C.M. and M.S.M. Jetten. 1998. Microbiological Conversions in Nitrogen Removal. *Water Sci. Technol.*, 38(1), 1-7.

van Loosdrecht, M.C.M. 2008. "Innovative Nitrogen Removal." In *Biological Wastewater Treatment: Principle, Modeling, and Design*. London: IWA Publishing.

Vazquez-Padin, J.R., I. Fernadez, M. Figueroa, A. Mosquera-Corral, J.L. Campos, and R. Mendez. 2009. Applications of Anammox Based Processes to Treat Anaerobic Digester Supernatant at Room Temperature. *Bioresource Technology*, 100, 2988-2994.

Ward, B.B., D.J. Arp, and M.G. Klotz. 2011. *Nitrification*. Washington DC: ASM Press.

WEF and WERF (Water Environment Federation and Water Environment Research Foundation). 2015. S*hortcut Nitrogen Removal —Nitrite Shunt and Deammonification*. Alexandria, VA: Water Environment Federation.

Wett, B. 2006. Solved Upscaling Problems for Implementing Deammonification of Reject Water. *Water Sci. Technol.*, 53(12), 121-128.

Wett, B. 2007. Development and Implementation of a Robust Deammonification Process. *Water Science & Technology*, 56(7), 81-88.

Wett, B., G. Nyhuis, I. Takács, and S. Murthy. 2010a. Development of Enhanced Deammonification Selector. 2010 Water Environment Federation Technical Exposition and Conference (WEFTEC).

Wett, B., M. Hell, G. Nyhuis, T. Peumpel, I. Takács, and S. Murthy. 2010b. Syntrophy of Aerobic and Anaerobic Ammonia Oxidisers. *Wat. Sci. Technol.*, 61(8), 1915-1922.

Wett, B., S. Murthy, I. Takács, M. Hell, G. Bowden, A. Deur, and M. O'Shaughnessy. 2007. Key Parameters for Control of DEMON Deammonification Process. *Water Practice*, 1(5).

Windey, K., I. De Bo, and W. Verstraete. 2005. Oxygen-Limited Autotrophic Nitrification–Denitrification (OLAND) in a Rotating Biological Contactor Treating High-Salinity Wastewater. *Water Res.*, 39, 4512-4520.

Winkler, M.K.H., R. Kleerebezem, and M.C.M. van Loosdrecht. 2012. Integration of Anammox into the Aerobic Granular Sludge Process for Main Stream Wastewater Treatment at Ambient Temperatures. *Water Res.*, 46(1), 136-144.

Weissenbacher, N., I. Takács, S. Murthy, M. Fuerhacker, and B. Wett. 2010. Gaseous Nitrogen and Carbon Emissions from a Full-Scale Deammonification Plant. *Water Environ. Res.*, 82(2), 169-175.

Zhang, Q., S.E. Vlaeminck, C. DeBarbadillo, C. Su, A. Al-Omari, B. Wett, T. Pumpel, A. Shaw, K. Chandran, S. Murthy, and H. De Clippeleir. 2018. Supernatant Organics from Anaerobic Digestion after Thermal Hydrolysis Cause Direct and/or Diffusional Activity Loss for Nitritation and Anammox. *Water Res.*, 143, 270-281.

Zhao, H., R. Lemaire, M. Christensson, G. Thesing, F. Veuillet, J. Ochoa, D. Lamarre, and A. Gadbois. 2013. Single-Stage Deammonification Process Performance—MBBR Versus IFAS Configurations. In *Proceedings of the Water Environment Federation Nutrient Removal and Recovery Conference*.

Zhao, H., J. Bell, G. Thesing, S. Piveteay, M. Christensson, F. Veuillet, J. Ochoa, and R. Lemaire. 2014. One-State Deammonification Process—MBBR versus IFAS Configurations—Development Updates and Application Considerations. In *Proceedings of the Water Environment Federation Residuals and Biosolids Conference*.