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Review and evaluation of the application of anaerobic ammonium removal technology for wastewater treatment

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Executive Summary

Australian red meat processing facilities generate large volumes of wastewater that require treatment to remove organic and nutrient contaminants in order to comply with water discharge regulations. This project reviewed the application of anaerobic ammonium removal technology against a range of current and developing technologies and processes for treatment of combined slaughterhouse wastewater. A brief summary of reviews outcomes are:

- Biological nutrient removal processes, such as nitrification/denitrification are an established and low risk option and have been applied to slaughterhouse wastewater at laboratory, pilot and full scale achieving nitrogen and COD removal above 90%. While effective, BNR processes have a higher demand for aeration energy, higher production of waste sludge and in some cases may require chemical carbon addition.
- Anaerobic ammonium removal is a relatively new technology that utilises a short cut in the nitrogen cycle and results in theoretical aeration cost savings of approximately 60%. High nitrogen loading rates reduce both the footprint and the investment costs of AAR in comparison the BNR processes and creates capacity with improved sustainability. AAR is generally targeted towards streams with NH_4-N concentrations >200 mg/L and with low COD and BOD content. However there are limited examples of application to slaughterhouse wastewater and this does result in some risk.
- Ammonia stripping is not considered an appropriate option for treating slaughterhouse wastewater due to high chemical costs, high aeration energy costs and the potential to release odour that may breach EPA guidelines.
- Constructed wetlands are not a viable option for slaughterhouse wastewater due to the relatively poor ammonium/nitrogen removal, the relatively poor phosphorus removal and the high footprint.
- Struvite crystallisation is not suitable as a standard alone technology for N removal, but may provide significant benefits to processing plants where P removal is required.

If N loading rates of 0.7 kgN/m^3/d can be achieved in AAR processes in red meat processing applications (as achieved in broader industries and shown in literature), the capital costs of AAR processes could be considerably lower than the more conventional BNR processes. This is largely due to the difference in nitrogen loading rates (0.1-0.3 kgN/m^3/d for BNR) and the subsequent difference in vessel size. Operating costs also appear to be 2-4 times lower for AAR compared to conventional BNR processes; this is due to much higher aeration costs associated with nitrification/denitrification and oxidation of degradable COD. Additionally, the assessments of BNR processes required a portion of raw wastewater to bypass the CAL to provide carbon for denitrification; this resulted in a reduction in biogas produced by the CAL and a reduction in the potential revenue recovered. The combinations of these factors suggest a payback of 4-6 years for a CAL + AAR process compared to over 15 years for a CAL + BNR process. Payback relies heavily on the value of recovered energy from the CAL and does not consider interest on the capital.

AAR has emerged as a promising candidate for treating slaughterhouse wastewater. When considering the placement of AAR into the wastewater treatment train, AAR could potentially be integrated into the current mainline treatment train after primary screening (to reduce TSS and FOGs) or after anaerobic treatment (to remove organic material).

Initially, there appear to be several technical barriers to application of AAR to mainline slaughterhouse wastewater directly after primary treatment including the presence of FOG, the high degradable COD/BOD content, the high COD/N ratio and the low fraction of N as ammonium. This

application is not recommended; however many of these barriers would also impact conventional BNR technologies.

Anaerobic treatment appears to address many of the challenges/barriers to AAR of raw wastewater. Therefore, anaerobic lagoon effluent has been identified as the most likely place to implement AAR into existing wastewater treatment at Australian meat processers. However, further investigation is recommended to assess the impact of non-degradable COD on anammox activity; and to determine removal efficiencies and predict the total nitrogen and ammonium concentrations in AAR effluents compared to discharge limits for the Australian red meat processing industry.

Contents

Abbreviations

AAR	Anaerobic Ammonium Removal
AD	Anaerobic Digestion
AL	Anaerobic Lagoon
Anammox	Anaerobic Ammonium Oxidation
AnMBR	Anaerobic Membrane Bioreactor
AOB	Ammonia-oxidizing Bacteria
BNR	Biological Nitrogen Removal
CAL	Covered Anaerobic Lagoon
COD	Chemical Oxygen Demand
DAF	Dissolved Air Flotation (tank)
DO	Dissolved Oxygen
FNA	Free Nitrous Acid
FOG	Fat, Oils and Grease
HRT	Hydraulic Residence Time
MBBR	Moving Bed Biofilm Reactor
MLSS	Mixed Liquor Suspended Solids
N	Nitrogen
NH_4-N	Ammonium nitrogen
NO_2-N	Nitrite Nitrogen
NO_3-N	Nitrate Nitrogen
NOB	Nitrite-oxidizing Bacteria
OUR	oxygen uptake rate
P	Phosphorus
PO_4-P	Phosphate Phosphorus
SBR	Sequencing Batch Reactor
SRT	Sludge Retention Time
TKN	Total Kjehldahl Nitrogen
TKP	Total Kjehldahl Phosphorus
TPAD	Temperature Phased Anaerobic Digestion
TS	Total Solids
TSS	Total Suspended Solids
UASB	Upflow Anaerobic Sludge Blanket
VFA	Volatile Fatty Acids
VS	Volatile Solids

1 Introduction

1.1 Project Purpose and Description

This project aims to review the application of anaerobic ammonium removal technology for wastewater treatment at red meat processing facilities against current (conventional) technologies and processes.

This review will address several research questions including, but not restricted to, those listed below. The review will include coverage of current (conventional) technologies and approaches, national and international publications detailing the application of anaerobic ammonium removal technologies and practices, and objective evaluation of three feasibility studies (prepared as part of the project 'A.ENV.0164 - Feasibility Study to qualify the approach for applying anaerobic ammonium removal technology for wastewater treatment at red meat processing facilities') into the application of anaerobic ammonium removal technology at three representative meat processing sites.

The review is to include (method):

1. Coverage of current (conventional) technologies in meat processing and the current practices for wastewater treatment with regards to identifying the potential for the application of anaerobic nitrogen removal technologies;
2. An explanation of how ammonium removal technology is currently used in other sectors and how it might be applied to our industry – including the challenges and suitability of application;
3. Consideration of process and activities for current/conventional treatment (for meat processing) and where anaerobic nitrogen removal technologies could be applied.
4. An overview of the R&D gaps/opportunities that exist in relation to integrating anaerobic ammonium removal technologies;
5. An overview of the next likely R&D step(s) in relation to the above;
6. A summary of some alternatives, when considering the above report, current MLA/AMPC projects and other work currently underway within the meat processing industry such as HRAT, AMBR, etc.) that might provide an alternative or complement the integration of ammonium removal technology.

The project will build on current industry research and development in this area, including:

- A.ENV.0164 Feasibility study into the application of anaerobic ammonium removal technology for wastewater treatment at red meat processing facilities
- A.ENV.0132/0150 High Rate aerobic treatment with AD and ANAMMOX
- A.ENV.0133/0149 Integrated agroindustrial wastewater treatment and nutrient recovery
- A.ENV.0154 Nutrient recovery from paunch and DAF sludge (struvite)
- A.ENV.0151 NGERS and Wastewater Management – mapping waste streams and quantifying the impacts.

1.2 Project Objectives

The project objectives as per the A.ENV.0162 project contact are:

1. Carry out a review and evaluation of the application of anaerobic ammonium removal technology for wastewater treatment at red meat processing facilities against current (conventional) options/technologies and processes. The review should take into account the research questions listed in project overview (Section 1.1). The review should also outline

the potential for the application of anaerobic ammonium removal technology for wastewater treatment within the Australian red meat processing industry.

2. Prepare a critique of the feasibility study prepared as part of the project 'A.ENV.0164 - Feasibility Study to qualify the approach for applying anaerobic ammonium removal technology for wastewater treatment at red meat processing facilities'.

1.3 Background

Australian red meat processing facilities generate large volumes of wastewater rich in organic contaminants and nutrients [1, 2]. While potentially expensive, the removal of these contaminants is necessary in order to comply with water discharge regulations. Therefore red meat processing facilities are strong candidates for advanced treatment processes aimed at removal and/or subsequent recovery of energy, nutrient, and water resources.

Waste and wastewater originates from several major process operations at a slaughterhouse including cattle preparation, cattle slaughter, recovery of by-products and reprocessing of by-products [2]. Generally, waste streams from different processing areas are transported separately within the site then combined for bulk treatment (e.g. in an anaerobic lagoon). The structure of waste and wastewater handling processes varies between sites; however a recent investigation of 6 Australian meat processing facilities (A.ENV.0151) identified common trends. A general structure of wastewater handling practices is presented in Figure 1.

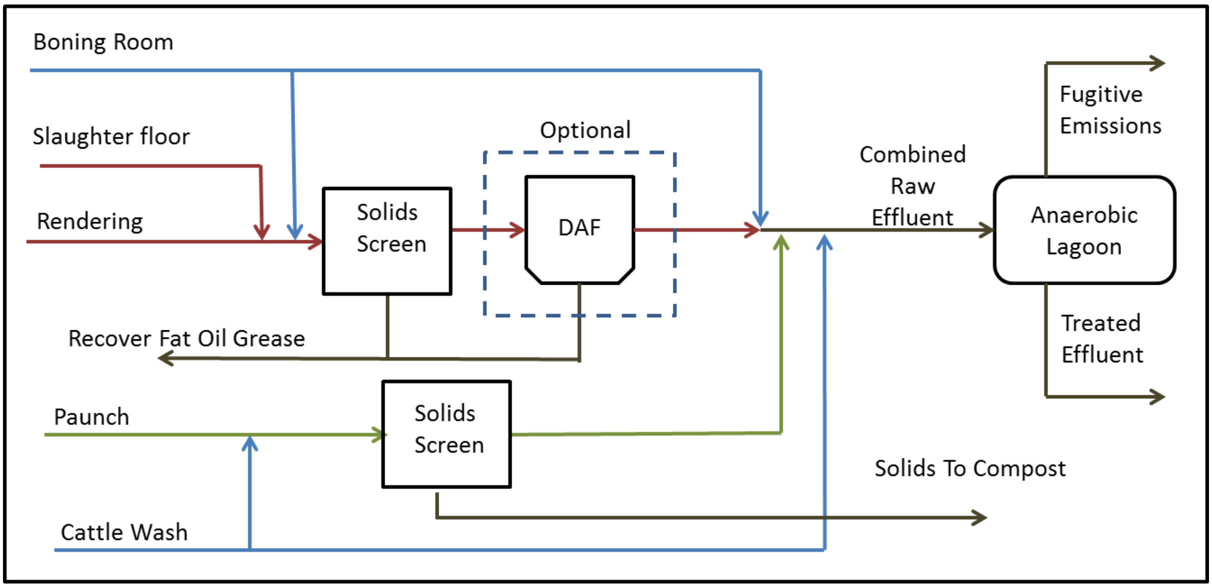


Figure 1: Major wastewater sources and generalised structure of waste and wastewater handling practices at Australian red meat processing sites

Combined slaughterhouse wastewater is composed of a mixture of grease, fat, protein, blood, intestinal content, manure and cleaning products [1]. It contains high concentrations of organic matter (represented by chemical oxygen demand, COD); oil and grease (FOG); nitrogen (N); phosphorus (P) and other trace metals. The characteristics of slaughterhouse wastewater after basic solids screening/fat recovery as reported in several international studies are shown in Table 1.

Table 1: Characteristics of slaughterhouse wastewater after primary treatment/solids removal [3].

Reference	Country	TCOD mg/L	SCOD mg/L	FOG mg/L	TKN mgN/L	NH4-N mgN/L	TP mgP/L
Borja et al. [4]	Spain	5,100	-	-	310	95	30
Caixeta et al. [5]	Brazil	2,000-6,200	-	40-600	-	20-30	15-40
Li et al. [6]	China	628-1,437	-	97-452	44-126	25-105	10-16
Manjunath et al. [7]	India	1,100-7,250	-	125-400	90-150	-	8-15
Martinez et al. [8]	Spain	6,700	2,400	1,200	268	-	17
Nunez and Martinez [9]	Spain	1,440-4,200	720-2,100	45-280	-	-	
Russell et al. [10]	NZ	1,900	-	-	115	30	15
Sachon [11]	France	5,133	-	897	248	-	22
Sayed et al. [12]	Holland	1,500-2,200	-	-	120-180	-	12-20
Sayed et al. [13]	Holland	1,925-11,118	780-10,090	-	110-240	-	13-22
Stebor et al. [14]	US	4,200-8,500	1,100-1,600	100-200	114-148	65-87	20-30
Thayalakumaran et al. [15]	NZ	490-2,050	400-1,010	250-990	105-170	26-116	25-47

The current default treatment methods for slaughterhouse wastewater vary widely. However, the principal set up of wastewater treatment processes in the red meat industry are shown in Figure 2.

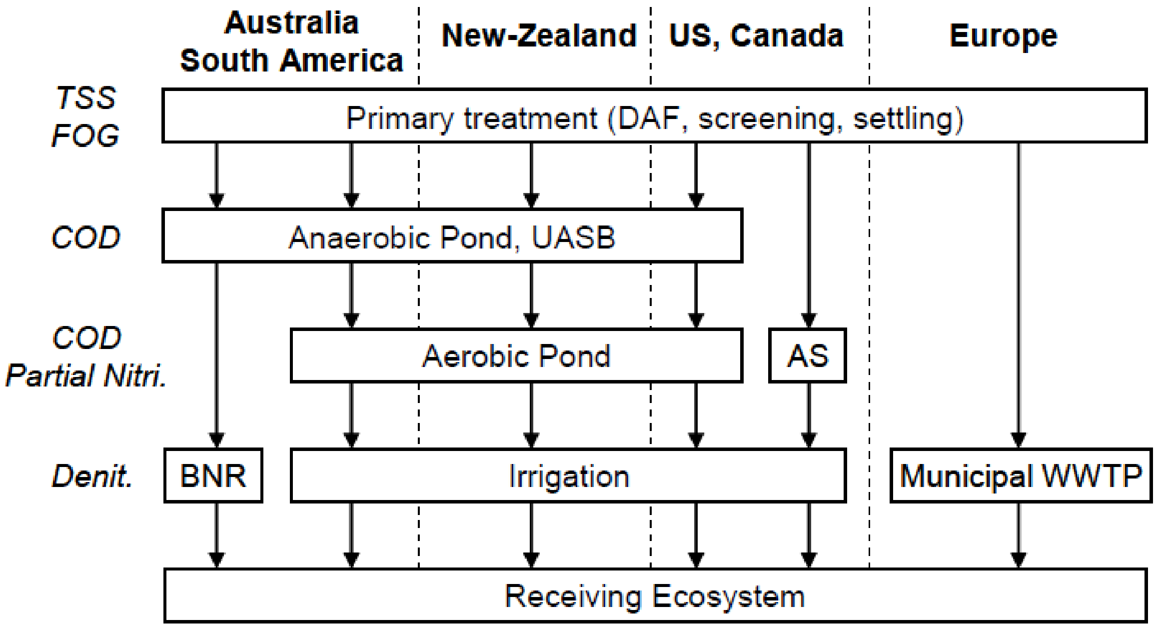


Figure 2: Principal wastewater treatment set-up of the meat industry [3]. Note: At some smaller Australian plants, primary treatment may be bypassed and/or raw effluent may be used for irrigation or land application.

The general processes in Australia include dissolved air flotation (DAF) as a pre-treatment to remove fat, oil and grease (FOG) and total suspended solids (TSS).

The DAF effluent is fed to an anaerobic treatment step. Anaerobic lagoons with hydraulic retention times (HRT) ranging between 7 and 14 days [16] are commonly used in tropical and equatorial temperate zones and engineered reactor systems (including activated sludge and UASB reactors) are commonly used in polar equatorial temperate zones. Anaerobic lagoons are effective at removing organic material (COD); however lagoon based processes also have major disadvantages including large footprints, poor gas capture, poor odour control, limited ability to capture nutrients and expensive de-sludging operations. Even in warmer climates, there is an emerging and strong case for reactor based technologies.

In the anaerobic step, proteins will be converted to biogas and the organic bound nitrogen will be realised as ammonium. Reliable biological COD and nitrogen removal systems have been successfully developed and applied for abattoir wastewater treatment using continuous activated sludge systems [17-19]. However, existing technologies can require energy intensive aeration steps and carbon chemical addition. Anaerobic ammonium removal technology is an emerging option to replace these existing (conventional) technologies for nitrogen/nutrient removal, with reductions in cost, energy consumption, footprint and elimination of chemical addition.

2 Anaerobic Nitrogen Removal

2.1 Introduction to Anaerobic Ammonium Oxidation (Anammox)

Anaerobic Ammonium Oxidation (anammox) was first discovered in a wastewater treatment plant in The Netherlands in 1995 [20]. Since its initial discovery, Anammox has been extensively researched as a promising method for nitrogen removal from ammonium rich wastewater of various sources.

In cooperation with the Dutch Universities Radboud Nijmwegen and TU Delft and the Dutch company Paques bv, an industrial process evolved in the next few years. The first full-scale anammox reactor (70 m^3) was built and commissioned in 2002 at a Dutch water utility in Rotterdam (WSHD). The reactor was inoculated with nitrifying sludge from the wastewater treatment plant. In September 2006, the reactor was in full operation and was converting between 8–10 kgN/m^3/day, a performance level that was twice its design capacity [21].

This reactor treated digester centrate in a two-step configuration. A single reactor high activity ammonium removal over nitrite (SHARON) reactor was used to oxidise half of the ammonium to nitrite and the effluent of this reactor was fed into the anammox reactor in a high rate internal recirculation (IC) set-up. In the last decade, other process configurations and several suppliers emerged and the anammox technology can be considered as fully grown.

2.1.1 The Nitrogen Cycle

Anaerobic Ammonium Oxidation (anammox) refers to a short-cut in the nitrogen cycle, where ammonium is oxidized directly to nitrogen gas using nitrite as the electron donor. As an anaerobic process, the aeration and energy demands of anammox are greatly reduced.

The nitrogen cycle involves the conversion of ammonium (NH_4^+) to nitrite (NO_2^-) and then nitrate (NO_3^-) through a (NO_3^-) through a process called nitrification, followed by the conversion of nitrate to nitrogen gas (N_2) in a process in a process known as denitrification. Nitrification is an aerobic process and requires substantial energy input through energy input through aeration. Denitrification is an anoxic process, aeration is not applied to this step, however the step, however the addition of an external carbon source (such as methanol) may be required. A summary of the summary of the nitrogen cycle and the anammox short-cut is shown in

Figure 3.

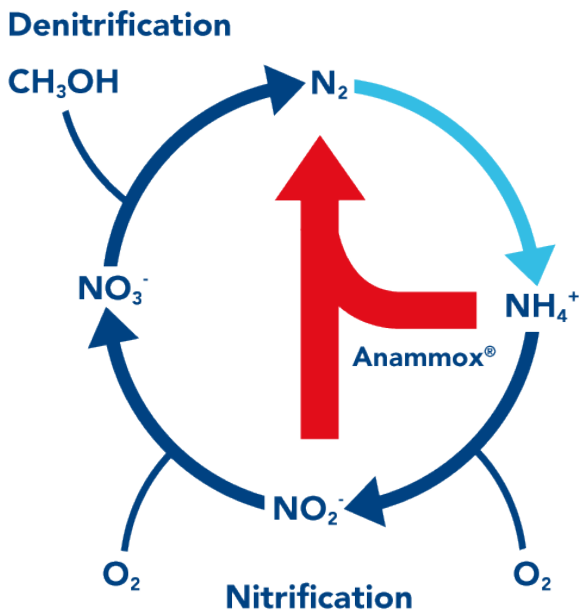


Figure 3: Nitrogen cycle with anammox pathway[1]

2.1.2 Biochemical Reactions of Anammox

Two different bacterial groups are involved in the anammox process which may or may not (depending on the process set-up) coexist in one reactor.

In the first step, ammonium oxidizing bacteria (AOB) oxidize approximately 50% of ammonium in the wastewater to nitrite (Eq.1). This is critical to provide nitrite for subsequent reactions. Anammox bacteria are then able to convert the remaining ammonium and nitrite into nitrogen gas (Eq.2 and 3). During the anammox reactions, approximately 10 - 12% of the nitrogen is converted to nitrate. The overall reaction of the anammox process is shown in Eq.4. The anammox bacteria are autotrophic using CO_2 as a carbon source; therefore there is no requirement to add an external carbon source (such as methanol). In most of the cases sufficient alkalinity is present in the wastewater to attain the needed NO_2-N/NH_4-N ratio of 1.32 [22].

Step 1. Partial nitrification (nitritation)

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + H_2^+ + H_2O \quad \text{(Eq.1)}$$

Step 2. Anammox

$$NH_3 + 1.32NO_2^- + H^+ \rightarrow 1.02N_2 + 0.26NO_3^- + 2H_2O \quad \text{(Eq.2)}$$

$$NH_4^+ + 1.3NO_2^- + 0.066HCO_3^- + 0.13\,H^+ \rightarrow 1.N_2 + 0.26NO_3^- + 0.066\,CH_2O_{0.6}N_{0.15}S_{0.05} + 2H_2O \quad \text{(Eq.3)}$$

Overall Reaction (partial nitrification + anammox)

$$NH_3 + 0.85O_2 \rightarrow 0.44N_2 + 0.1NO_3^- + 0.14H^+ + 1.43H_2O \quad \text{(Eq.4)}$$

The oxidation of ammonium to nitrate (Eq.5) by nitrite-oxidizing bacteria (NOB) is a competing biochemical reaction that must be inhibited or eliminated for stable and efficient anammox to occur. NOB inhibiting or eliminating reactor conditions include low dissolved oxygen (DO), high temperature, high concentrations of free ammonia and free nitrous acid (FNA) [23]. Depending on

[1] from http://en.paques.nl/pageid=66/ANAMMOX%AE.html (accessed 5/11/2013)

the process configuration, a selective up-flow velocity in the sludge retention system is also suitable for selective washout of NOB.

Nitrification

$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2^+ + H_2O$ (Eq.5)

2.1.3 Anammox Microbiology

Anammox biomass is recognized for its deep red colour, caused by specific enzymes (Figure 4). While anammox bacteria have not been isolated and grown as a pure culture, as of 2008, 8 species of anammox bacteria had been identified within mixed culture communities (Table 2). While the diversity of identified anammox bacteria is relatively limited, these microbes are known to exist in both engineered systems and natural environments with oxygen depleted zones and available nitrogen sources. These environments include wastewater treatment systems, marine ecosystems, freshwater ecosystems, terrestrial ecosystems, and even sea ice [24]. While obtaining a culture containing anammox bacteria is relatively easy, the anammox bacteria are often in very low abundance within these populations and must be enriched before they will be effective in anaerobic ammonium removal technologies. Anammox bacteria are slow growing microorganisms with a doubling time of approximately 11 days at 32 - 33°C [25]. Therefore, enrichment from natural occurring populations can be a slow and difficult process.

The anammox bacteria are very sensitive to oxygen exposure and higher nitrite concentrations [22]. The toxic nitrite level depends on size of biomass aggregates and acclimation periods [26]. In addition, methanol inactivates the anammox bacteria [27].

Table 2: List of Anammox Bacteria (adapted from [28])

Genus	Species	Source
Brocadia	Candidatus Brocadia anammoxidans	Wastewater
	Candidatus Brocadia fulgida	Wastewater
Kuenenia	Candidatus Kuenenia stuttgartiensis	Wastewater
Scalindua	Candidatus Scalindua brodae	Wastewater
	Candidatus Scalindua wagneri	Wastewater
	Candidatus Scalindua sorokinii	Seawater
Others	Candidatus Jettenia asiatica	Not Reported
	Candidatus Anammoxoglobus propionicus	Synthetic water

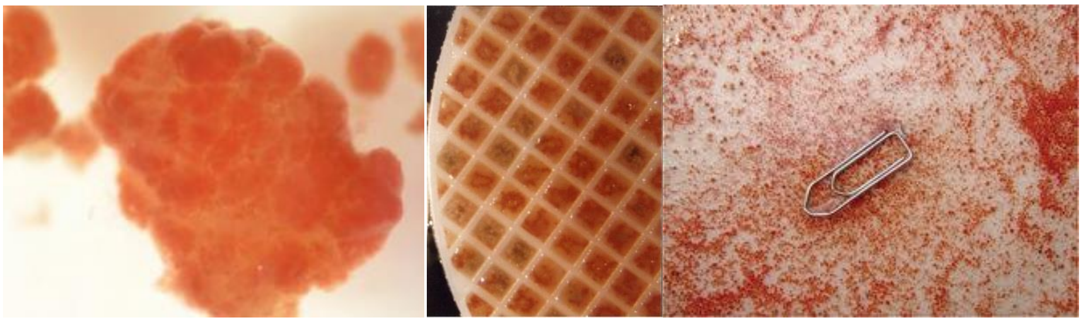


Figure 4: Picture of an anammox granule [29] (left), anammox on carrier (middle) and anammox sludge from an SBR (right).

2.1.4 Process Operation and Performance

Anammox has been successfully applied at laboratory scale, pilot scale and full scale for the treatment of ammonium-rich wastewater. The anammox process is generally targeted towards streams with NH_4-N concentrations >200 mg/L and with low COD and BOD content. Theoretically the NH_4-N removal can be as high as 100% with typical operating efficiencies of ~95%. Ammonium removal will be higher than total nitrogen (TN) removal (88 – 90%), due to the small concentrations of nitrate produced in the anammox reactions (see Eq.4).

Anammox effluent will typically contain some NH_4^+ (10 - 50 mg/L), NO_2^- (1 - 20mg/L) and NO_3^- (10 – 12% of the NH_4^+ removed). The NH_4^+ background concentration is due to operational/control decisions to provide protection against over-oxidation. The flowrate and NH_4^+ concentration in wastewater fluctuates over time, therefore the amount of oxygen needed for partial nitrification also fluctuates. When supplying excess oxygen, ammonium may be converted to nitrite above the desired stoichiometric ratio of the anammox reaction. This results in excessive aeration costs and energy consumption. Excess nitrite will also allow growth of NOBs and oxidation of NO_2-N to NO_3-N. The nitrate cannot be used by the anammox bacteria and thus total nitrogen removal efficiency is reduced. Where the total nitrogen concentration in the effluent is too high to meet discharge requirements COD can be dosed to denitrify the remaining nitrate. However, this adds cost to the process.

The short cut of the anammox reaction in the nitrogen cycle and the oxidation of only 50% of ammonium to nitrite, compared to 100% of ammonium to nitrite then nitrate, results in theoretical aeration cost savings of approximately 60%. Savings will depend on the residual COD in the wastewater, which consumes some of the supplied oxygen. There are additional benefits from the elimination of carbon chemical addition required as part of the conventional denitrification step.

Due to the slow growth rates of anammox bacteria, waste sludge production is reduced to approximately 20% that of conventional nitrification-denitrification processes. However, due to the benefits of seed sludge during process start-up and the lack of available anammox sludge in Australia, the waste anammox sludge would be considered as a valuable product in the short to medium term.

The conversion rates of the anammox bacteria are around 1.4 kgN/kgVSS and volumetric loading rates up to 0.5 [30]-2 [31] kgN/m^3/day can be achieved. This reduces both, the footprint and the investment costs of the installation and creates capacity with improved sustainability. Due to the reduced aeration and the potential saving of extra COD (e.g. methanol) the CO_2 footprint can be reduced by up to 90% However, savings at this level are not typical.

2.1.5 Inhibition and Deactivation of Anammox Process

Anammox process control optionally includes online nitrite, ammonium, DO, pH and temperature measurement but the process can also operate with less control. Environmental conditions that inhibit anammox bacteria and/or reduce activity are an area of constant investigation in literature. Currently, 37°C is suggested as the optimum temperature for AAR activity with a significant decline in activity at 45°C. Some examples of inhibitory compounds and concentrations are shown in Table 3.

Inhibitory concentrations can vary significantly for individual microbial communities and are presented as a guide only. For example, NH_4-N resulted in a 50% reduction in activity at 770 mg/L in one study [32], but resulted in no inhibition at 980 mg/L in another study [22]. Similarly, sulfide resulted in a 50% reduction in activity at 9.6 mg/L in one study [32], but improved activity at 32-160 mg/L in another study [22]. It is possible that the form of the inhibitory compound is also a critical factor influencing inhibition (e.g. sulfide/sulfate) and would therefore be impacted by temperature and pH.

The concentrations of ammonia, phosphate and chloride present in combined meat processing wastewater are expected to be much lower than the values in Table 3 and are not expected to cause inhibition. The concentrations of DO and nitrite will be subject to process control, but are also not expected to cause inhibition. However, in addition to specific inhibitors, anammox bacteria are sensitive to high COD with COD/NH_4-N ratios above 2 reported to suppress anammox activity [33], this will be a challenge for the meat processing industry.

Table 3: Compounds and concentrations reported to inhibition anammox bacteria and process activity

	unit	Dapena-Mora et al. [32]	Jung et al. [34]	Strous et al. [22]	Van de Graaf et al. [35]
Dissolved Oxygen	mg/L	-	0.2	-	-
Ammonia - N	mg/L	770	-	980 (no effect)	-
Nitrite - N	mg/L	350	35	98	-
Acetate	mg/L	2301	-	-	59 (improved activity)
Sulfide - S	mg/L	9.6	-	-	32 (improved activity)
Phosphate - P	mg/L	651	-	-	155
Chloride	mg/L	7080	-	-	1770 (no effect)

2.2 Anaerobic Ammonium Removal Technology Configurations and Providers

2.2.1 ANAMMOX®

ANAMMOX is a commercial technology developed by Paques BV. The first installation was built as a two-step configuration with a retrofitted clarifier as SHARON reactor (where ammonium is oxidised to nitrite) followed by an Internal Circulation (IC) reactor (where the subsequent anammox reactions occur). The process diagram for the two reactor ANNAMOX set-up is shown in Figure 5. An example full scale installation of the two reactor ANAMMOX process is the Sewage Treatment System (STW) Rotterdam (plant capacity of 620,000 equivalent persons). Paques reports that at this installation the ANAMMOX® technology saves over 250 tonnes of methanol per year and 275,000 kWh electricity. ANAMMOX® resulted in annual savings of A$178,000 in operational costs and a reduction of STW's carbon footprint by 500 tonnes per year[2].

[2] From http://en.paques.nl/pageid=101/articleid=239/Waterboard_Hollandse_Delta_%28STW_Rotterdam%29.html.

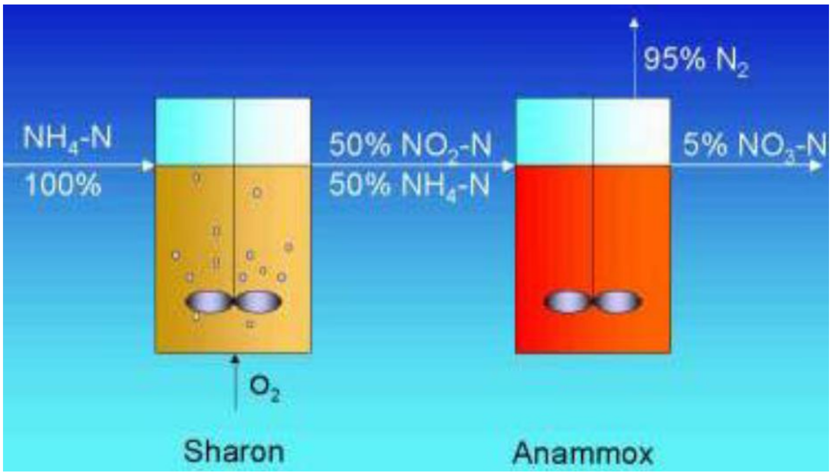


Figure 5: Schematic set-up of a two-step system, including a SHARON® and ANAMMOX® reactor[3]

Paques bv now supplies a single reactor process whereby ammonium oxidation to nitrite and the anammox reactions occur in the same reactor. ANAMMOX® processes contain granular biomass, rather than biomass growth on artificial carriers. The biomass retention in this system is achieved via lamellar settlers and relies on the high settling velocity of the granular biomass. The mixing and oxygen supply for the AOBs is achieved with fine bubble aerators. Figure 6 shows the principal set-up of the reactor and the anammox granular biomass of the Paques bv system; closed and open reactors are available.

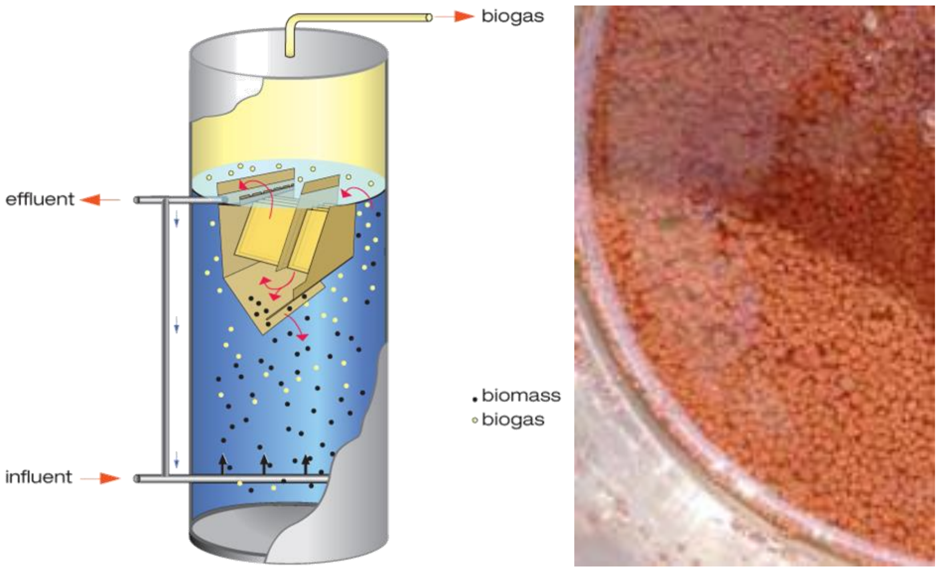


Figure 6: Schematic set-up of the ANAMMOX® process (left)[4] and the biomass content of this reactor set-up (right)[5].

ANAMMOX has now been applied at full scale for a range of wastewater types; a reference list of known full scale installations is shown in Table 4. From Table 4 it is clear that ANAMMOX® processes are flexible in scale varying from 60 kgN/d to over 10,000 kgN/d.

[3] From http://www.stowa-selectedtechnologies.nl/Sheets/Sheets/sharon-anammox%201904_files/image002.jpg.
[4] From http://en.paques.nl/pageid=206/BIOPAQ%AEUASB+.html
[5] From http://www.waterwastewaterasia.com/ebook/WWA_JulAug2012/files/assets/seo/page48_images/0002.jpg

Table 4: ANAMMOX® reference list (provided by Paques bv 05-2013).

Installation	Wastewater	kgN/d	Year
Severn Trent-Stoke Bardolph (UK)	centrate	1600	2012
Energiefabriek RWZI Tilburg (NL)	centrate	2100	2012
Zheljiang Guyuelongshan Shaoxing Wine (CN)	yellow wine	1045	2012
Ningzia Eppen Biotech (CN)	MSG	10100	2012
Kuaijishan Shaoxing Winery (CN)	distillery	900	2011
Rendac (NL)	rendering	5700	2011
Severn Trent (UK)	centrate	4000	2011
Xinjiang Meihua amino Acis (CN)	MSG	10710	2011
Jiangsu Hangguang Bio-engineering (CN)	sweetener	2180	2011
Confidential client (PL)	distillery (Wheat stillage)	1460	2011
Shandong Xiangrui (CN)	corn starch MSG	6090	2011
Waterschap Groot Salland (NL)	centrate	600	2011
Meihua II (CN)	MSG	9000	2010
Meihua I (CN)	MSG	11000	2009
Angel yeast (CN)	yeast	1000	2009
ARA Niederglatt (DE)	centrate	60	2008
Semiconductor plant ((JP)*	semiconductor	220	2006
Waterstromen Steenderen (NL)	potato	1200	2006
Industry Water Lichtenvoorde (NL)*	tannery	325	2004
WSHD (NL)*	centrate	500	2002

*Two reactor process configuration

2.2.2 Anita™ Mox

Anita™ Mox is provided by Veolia and is based on a single vessel Moving Bed Biofilm Reactor (MBBR) technology. In the reactor, AOBs and anammox bacteria grow on carriers which are kept in suspension by aeration and mixers. The MBBR is continuously aerated to oxidise the ammonium to nitrite. A principal set-up of the reactor and the biomass on carriers is shown in Figure 7.

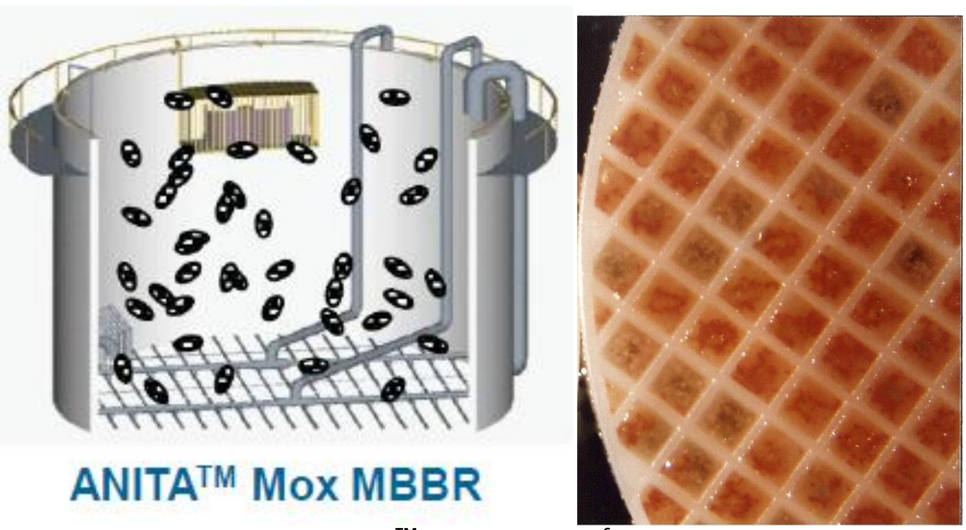


Figure 7: Schematic set-up of the ANITA™ Mox process (left)[6] and the biomass content (right)(left)

[6] From http://www.revistaseccion.com/images/stories/productos/Soporte_plstico_empleado_en_el_proceso_Anita_Mox_colonizado_con_bacterias_anammox.jpg

Anita™ Mox has now been applied at full scale for a range of wastewater types; a reference list of known full scale installations is shown in Table 5. Currently, the largest known Anita™ Mox plant in operation has a design capacity of approximately 400 kgN/d and is similar to the nitrogen load of an average Australian slaughterhouse.

Table 5: Anita™ Mox reference list (provided by Veolia 05-2013).

Installation	Wastewater	kgN/d	Year
Malmo -Sjolunda WWTP (SE)	municipal	200	2010
Vaxjo -Sundet WWTP (SE)	municipal	320-430	2011
Holbaek (DK)	municipal/industry	120	2012
Grindsted (DK)	municipal/industry	107	2013
Durham-Soth Durham WWTP (US)	municipal	330	2013
James River WWTP (US)	municipal	250	2013

2.2.3 DEMON®

Another anammox technology supplier is cyklar-stulz with the DEMON® process. The DEMON® process is based on sequence batch reactor (SBR) technology and with a hydro-cyclone for improved biomass retention. The SBR process occurs in a single reactor, run through a series of 5 operating modes (Figure 8). When applied to the DEMON® process, the aeration stage is intermittent to create an aerated phase and achieve the oxidation of ammonium to nitrite, followed by an anoxic phase for the anammox process. Figure 8 shows the principal phases of a SBR and the anammox biomass of the DEMON® process.

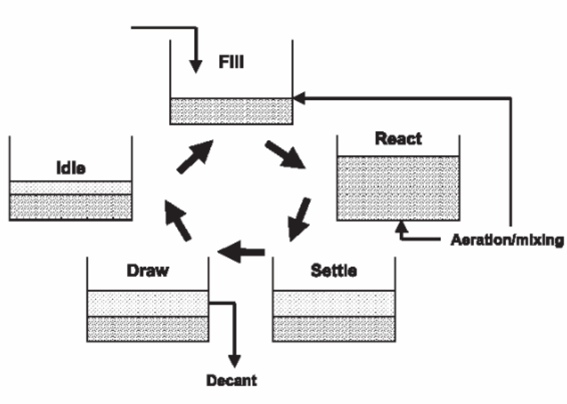


Figure 8: Principal phases of a SBR (left)[7] and the biomass content of the DEMON® technology (right)

DEMON® has now been applied at full scale for a range of wastewater types; a reference list of known full scale installations is shown in Table 6. DEMON® processes are also flexible in scale varying from 50 kgN/d to over 5,000 kgN/d.

[7] From http://www.inspectapedia.com/septic/tfs3fig1.gif

Table 6: List of known installations using DEMON® process (current to 2013)

Installation	Wastewater	kg/N	Year
Utrecht (NL)	filtrate	900	1997
Rotterdam (NL)	centrate	850	1999
Zwolle (NL)	centrate	410	2003
Beverwijk (NL)	centrate + condensate	1200	2003
Stass (AT)		600	2004
Groningen (NL)	filtrate + condensate	2500	2005
Den Haag (NL)	centrate	1200	2005
Glarnerland (CH)		250	2007
Plettenberg (DE)		80	2007
Thun (CH)		400	2008
Gengenbach (DE)		50	2008
Heidelberg (DE)		600	2008
New York (US)	centrate	5000	2009
Linkoping (SE)	centrate	500	2009
Etappi Oy (FI)		1000	2009
Balingen (DE)		200	2009
Appeldoorn (NL)		1900	2009
Geneva (CH)	centrate	1700	2010
Shell Green (UK)	centrate	1600	2010
Whitlingham (UK)	centrate (CAMBI)	1500	2010
Limmattal (CH)		250	2010
Zalaegerzeg (HU)		160	2010
Alltech (Serbia)		2400	2011
Seine Gresillions (FR)	centrate	3500	2012

2.2.4 DeAmmon®

The DeAmmon® process has been developed in co-operation between Purac, Ruhrverband, and ISAH, The University of Hanover, Germany. The DeAmmon® process works in an SBR configuration with biomass growing on plastic suspended carriers for biomass retention (like an MBBR). Specially designed mixers keep the carriers in suspension and sieves at the effluent outlet avoid carrier washout. The aeration is intermittent to achieve ammonium oxidation and the anammox reaction.

The process configuration and the carrier material are similar to the cyklar-stulz and the Veolia process. A reference list of known DeAmmon® installations is shown in Table 7.

Table 7: DeAmmon® reference list.

Installation	Wastewater	kg/N	Year
Hattingen (D)	centrate	120	2003
SYVAB Stockholm (SE)	centrate	600	2007

2.2.5 Cleargreen™

The Cleargreen™ process (distributed by Suez) also works in an SBR configuration that allows the successive completion of all treatment phases in the same tank. The different SBR phases are divided with 4 sub-cycles of aeration (nitrification) and anammox reaction. The sub-cycles can be adapted in duration and intensity depending on the wastewater characteristics. The biomass grows in flocks which are settled prior of withdrawing the effluent. The principal set-up and phases of an SBR were previously shown in Figure 8. The Cleargreen™ is a newer technology and a reference list of full scale plants in operation is not currently available.

2.2.6 Comparison of Different Process Features

A comparison of the process features of existing anaerobic ammonium removal technologies is shown in Table 8. The TN and NH_4-N removal efficiencies are expected to be comparable for all technologies since this is a feature of the anammox biochemical reactions and the capability is not significantly impacted by the process configuration.

The volumetric loading rates differ where SBRs have lower nitrogen loading rates compared to the ANAMMOX® continuous one-step system.

The energy input is expected to be similar in terms of the anammox reaction. However, design decisions including aerator selection (fine vs. coarse) and reactor geometry (and therefore head pressure); and process variables including nitrogen load, wastewater COD concentration and temperature will influence the energy demand. In many cases where the reactor is under-loaded the energy demand will increase (kWh/kgN).

Start-up and commissioning times should be comparable with sufficient seed sludge (>50% of biomass needed). Generally, anammox sludge can be transported when kept anoxic, however customs and quarantine regulations currently prevent importation of seed sludge to Australia. When starting a new installation without seed sludge, reactor start-up can take up to 2 years, depending on size and wastewater characteristics. This is also expected to be similar between technologies. Note, with sufficient knowledge of the microbiology and processes, it is possible to transfer anammox sludge from one technology type to another.

The NO_2-N toxicity for the anammox bacteria decreases with increasing aggregate size. Suspended sludge tolerates less NO_2-N than flocculent sludge and flocculent less than granular sludge.

Table 8: A comparison of the process features of different anammox technologies.

		ANAMMOX®	ANITA™MOX	DEMON®	DeAmmon®	Cleargreen™
Volumetric loading rates	kgN/m^3/d	1.7-2.0	0.7-1.2	07-1.2	0.7-1.2	0.7-1.2
Performance TN removal	%	~90NH_3-N ~85TN	~90NH_3-N ~85 TN	~90NH_3-N ~85TN	~90NH_3-N ~85 TN	~90NH_3-N ~85 TN
Energy demand	kWh/kg NH_3-N removed	1.0-1.3	1.45-1.75	1.0-1.3	TBC	TBC
Start-up	months	1-3 w seed	4-5 w 2-10% seeding	2-5 w seed & cyclone	2-5 w seed	TBC
Sensitivity/ Flexibility		Tolerates elevated NO_2	DO control Tolerates elevated NO_2	pH & DO control NO_2<5mg/L Pre-settling	pH & DO control NO_2<5mg/L Pre-settling	TBC

TBC: Data not available at time of writing this report

2.3 Application of Anammox to Municipal Wastewater Treatment

Municipal wastewater treatment plants commonly utilise an activated sludge process similar to the process schematic process schematic shown in

Figure 9. In the activated sludge process, organic solids in the wastewater feed may be removed at the start of the process using a primary settling tank, nutrients are typically then removed from the wastewater using a biological nutrient removal (BNR) process (mainline treatment). The BNR process generates large volumes of waste sludge which is separated and treated using anaerobic digestion (side stream treatment).

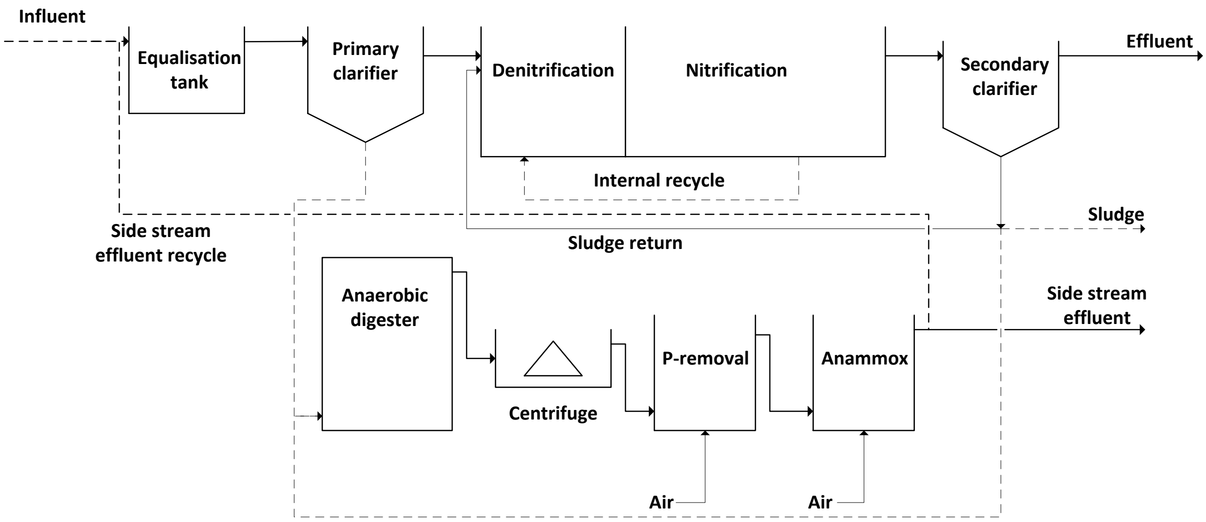


Figure 9: Schematic representation of conventional activated sludge treatment with side-stream treatment.

The most common application of AAR in municipal wastewater processes is treatment of side stream effluent from the anaerobic digesters. During anaerobic digestion, organic bound nitrogen is released as soluble ammonium. Depending on the type of sludge entering the digester (primary sludge, waste activated sludge or a mixture) and the application of sludge pre-treatments (e.g. sonication, thermal hydrolysis), the NH_4^+ concentrations in digester effluent can vary from 500 – 2000 mg/L in the centrate.

Anaerobic digester centrate may be recycled to the mainline treatment process for biological nutrient removal (nitrification/denitrification). However, the side-stream effluent can contain up to 30% of the overall N load in the treatment plant, recycling this load would result in 30% additional volume requirement, energy consumption and sludge production. Application of AAR to the side stream effluent can therefore result in considerable cost savings through reduced energy demand and chemical consumption. AAR effluent may still be recycled to the mainline treatment process; however the recycled N load would be reduced by approximately 90% (
Figure 9).

Recently AAR technology suppliers have been working on the implementation of anammox as part of mainline treatment in a municipal wastewater treatment plant, shown in Figure 10. This includes challenges such as increased COD/N ratio, biomass retention, low temperature and resulting competition between AOBs and NOBs. The principal positioning of the AAR in the mainline would be after a high loaded A-step, in which the majority of the COD is removed as CO_2 and biomass. After clarification the supernatant contains a lower COD concentration that is not sufficient for denitrification (without the addition of extra COD e.g. methanol). Therefore, anammox can be applied in place of denitrification to remove the ammonium. However, there are still a number of operating challenges to be addressed around this process configuration.

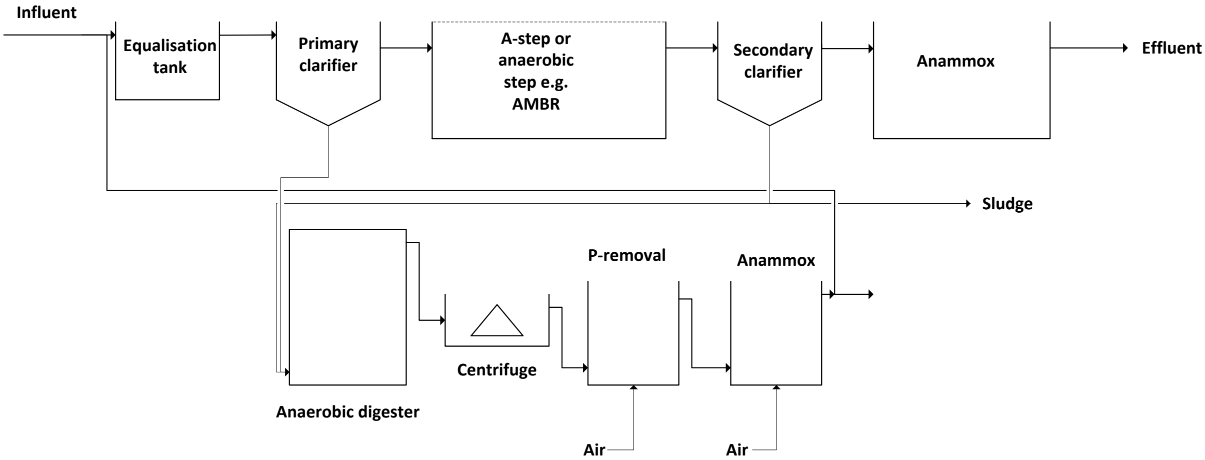


Figure 10: Potential setup with anammox in the main line of domestic wastewater treatment with side-stream treatment.

Table 9 presents a summary of performance data from 3 full scale SBRs at municipal treatment plants [30]. Ammonium removal was 92-97% in each of the 3 plants and is consistent with the benchmarks suggested by AAR technology suppliers. However, total nitrogen removal cannot be assessed as the concentrations of organically bound nitrogen were not presented. The ammonium nitrogen concentrations in the feed streams to the anammox processes were 650-760 mg/L and were 2-4 times higher than concentrations measured in slaughterhouse effluent, it is not clear from the full scale studies if this would impact the ammonium removal efficiency. Residual nitrogen concentrations from the 3 full scale plants ranged from 35 mgN/L to 150 mgN/L suggesting that additional treatment may be required prior to discharge.

Plant	Stream	Temp °C	pH	TSS mg/L	TCOD mg/L	SCOD mg/L	NH4-N mg/L	NO2-N mg/L	NO3-N mg/L	DOC mg/L
Zurich	Input	27	7.8	250	630	300	650	<0.2	<0.2	80
	Output	30	7.1	150	400	190	30	<0.2	5	N/A
St. Gallen	Input	N/A	N/A	370	770	N/A	890	N/A	N/A	N/A
	Output	18-30	8	120	325	190	73	1	80	72
Niederglatt	Input	20	7.7	5	N/A	N/A	760	<0.5	3	N/A
	Output	29	7.8	4	N/A	N/A	20	0.5	50	N/A

Table 10 presents an overview of typical average nitrogen concentrations and volume loading/removal rates for single reactor processes incorporating partial nitrification and anammox. Nitrogen removal efficiency was estimated by comparing the N loading rate and the N removal rate. The processes were generally operating within the nitrogen loading range expected for anammox processes (0.7-1.2 gN/L/d). Table 10 incorporates several studies where the ammonium concentration in the feed was similar to levels expected from slaughterhouse effluent; of these studies the nitrogen removal efficiency was generally poor at approximately 50%. However, in one study on landfill leachate where the nitrogen loading rate was lower (0.38 gN/L/d), nitrogen removal was estimated at 100%. This suggests lower ammonium concentrations may be a challenge, but are not a barrier to anaerobic ammonium oxidation.

Table 10: Overview of typical average nitrogen concentrations and volume loading/removal rates for existing single reactor anammox processes (adapted from [36])

Wastewater	Reactor Type	Influent	N loading	N removal	Efficiency	Reference
		mgN/L	gN/L/d	gN/L/d		
Sewage sludge digestate	SBR	650	0.54	0.51	0.94	Joss et al. [30]
Digested Black water	RBC[1]	1023	0.94	0.71	0.76	Vlaeminck et al. [37]
Sewage sludge digestate	SBR[2]	800	0.74	0.67	0.91	Jeanningros et al. [38]
Landfill leachate	RBC	209	0.38	0.38	1.00	Hippen et al. [39]
Landfill leachate	RBC	250	0.67	0.41	0.61	Siegrist [40]
Sewage-like nitrogen level	RBC	66	0.86	0.44	0.51	De Clippeleir et al. [36]
Sewage-like nitrogen level	RBC	31	0.84	0.38	0.45	De Clippeleir et al. [36]

1. RBC - rotating biological contactor
2. SBR - sequencing batch reactor

2.4 Applications to Broader Industrial Wastewaters

The anammox process can be applied for streams with ammonium concentrations >200 mg/l and with low COD and BOD content. Examples of industries with ammonium rich wastewaters that may be suitable for anammox include:

- Municipal wastewater treatment (reject water from a sludge digester)
- Organic solid waste treatment (landfills, composting, digestion)
- Food industry
- Manure processing industry
- Fertilizer industry
- (Petro)chemical industry
- Metal and Mining industry
- Slaughterhouse

The placement of the anammox process for industrial application is generally after an anaerobic treatment step, as suggested in Figure 11. The anaerobic step may be a high rate anaerobic process such as an Expanded Granular Sludge Blanket system (EGSB), an Internal Circulation system (IC) or Upflow Anaerobic Sludge Blanket reactor (UASB). The anaerobic step may also be after a conventional solids digester or an anaerobic lagoon. The requirements of the anaerobic step are similar to municipal treatment; the COD will be consumed in the anaerobic step and transformed to biogas and the effluent will contain insufficient degradable COD for denitrification. In this case anammox can substitute the nitrification and denitrification step. Depending on the discharge limits (sewer, irrigation or surface water) and regulations, the effluent of the anammox step may or may not be directly discharged.

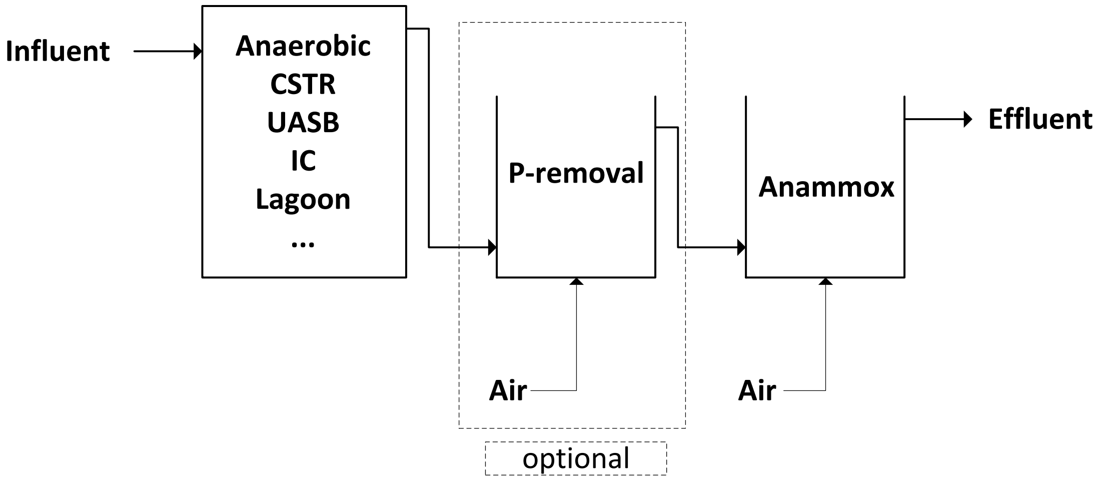


Figure 11: Principal side-stream setup and anammox positioning.

3 Alternative Technologies for Nitrogen Removal

3.1 Activated Sludge System/Biological Nitrogen Removal (BNR)

Biological nitrogen removal is achieved through nitrification and denitrification (Figure 12).

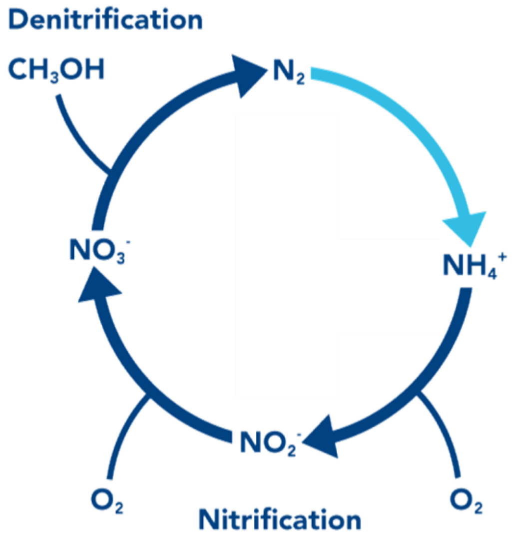


Figure 12: Nitrogen cycle showing nitrification and denitrification[8]

Nitrification, introduced in Section 2.1.1, is the process where ammonium is oxidized to nitrate. This process occurs in two steps under aerobic conditions; first the oxidation of ammonium to nitrite, and second the oxidation of nitrite to nitrate. These two processes are catalysed by two different groups of bacteria, called ammonia-oxidizing bacteria (AOB) and nitrite oxidizing bacteria (NOB),

[8] Adapted from http://en.paques.nl/pageid=66/ANAMMOX%AE.html (accessed 5/11/20130)

respectively. Nitrification is an aerobic process and oxygen is required for and consumed by both steps. Theoretically the aeration energy required for complete nitrification is 4.6 kWh/kgN.

Denitrification refers to the process that nitrate is reduced to nitrogen gas. The process occurs in several steps, namely the reduction of nitrate (NO_3^-) to nitrite (NO_2^-), the reduction of nitrite to nitric oxide (NO), the reduction of nitric oxide to nitrous oxide (N_2O), and finally the reduction of nitrous oxide to nitrogen gas (N_2). These processes are catalysed by a group of bacteria called denitrifiers. Organic carbon is required for all steps in denitrification to provide the electrons required for each of the reduction processes. Denitrifiers also assimilate some of the carbon sources for growth. This process occurs in the absence of oxygen.

Sufficient readily biodegradable COD (primarily VFA), is required for complete nitrate removal by denitrification. The stoichiometric value of the required COD:$N_{eliminated}$ mass ratio for denitrification is 2.86, including sludge production, the ratio increases to 4 (Mulder *et al.*, 2006). Therefore, the challenge for the primary treatment of abattoir wastewater is to reduce the carbon content through anaerobic pond systems, and therefore reduce the amount of COD oxidised by aeration, but keep sufficient COD for complete N removal [41]. By-passing of raw influent might be an option to increase the COD for denitrification in some cases. However, the FOG might disturb the sludge settleability.

Activated-sludge systems are also employed in North America to remove the COD and some of the nutrients before land application. Advanced tertiary treatments using biological and physicochemical methods have been employed in the US and in Australia to achieve complete nitrification and partial or even complete denitrification together with chemical phosphorus removal. However, their use is very limited due to high-cost involved [42].

3.2 Sequencing Batch Reactor Biological Nutrient Removal

Biological nitrogen removal requires both aerated (for nitrification) and non-aerated (for denitrification) conditions. Sequencing Batch Reactors (SBRs) are able provide both aerobic and non-aerated conditions by controlling the aeration through different operating stages (Figure 13). During some periods of the SBR cycle, air is supplied to provide oxygen for nitrification, while in other periods air supply is stopped to create anoxic conditions and enable denitrification. SBRs also include a stage where sludge settles to allow the treated wastewater to be drawn from the top of the reactor. Therefore, sludge settleability is very important.

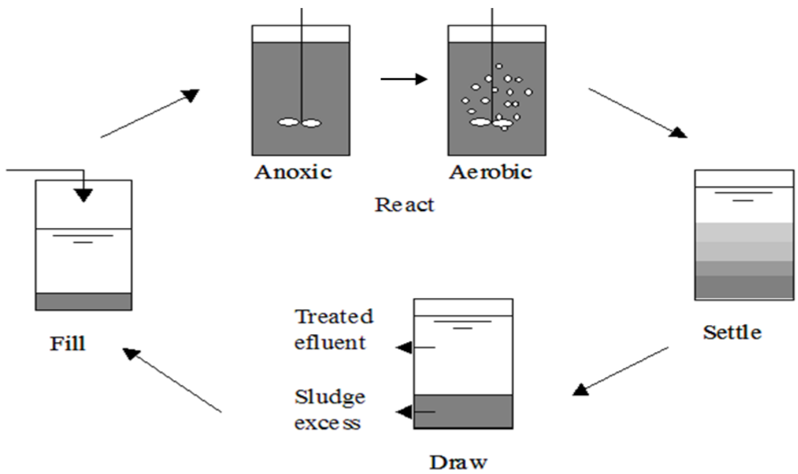


Figure 13: Principal operating modes of a Sequencing Batch Reactor (SBR) – Note: all stages occur within a single reactor.

SBRs have been applied to slaughterhouse wastewater at laboratory, pilot and full scale.

A lab scale study (5 L) treating wastewater collected from the anaerobic lagoon at a slaughterhouse [41] demonstrated simultaneous nutrient and COD removal in an SBR with flocculent biomass and intermittent aeration is possible. This study achieved 90% nitrogen removal and 90% phosphate removal without addition of an external carbon source. The nitrogen concentration in the final effluent (20 mgN/L) was in a similar range to that reported in anammox processes. Additional SBR lab-scale studies treating wastewater from the same location have achieved 95% total nitrogen removal and support these findings [43]. As a result of these studies an 8 m^3 SBR pilot was built in cooperation with the Australian Meat Research Corporation (ENV.044). The SBR pilot plant also reported over 90% removal of COD and both inorganic nitrogen (predominantly ammonia) as well as total Nitrogen. A summary of the challenges found operating the pilot plant is discussed in the final report to ENV.044.

The settling time of floccular biomass is a limitation in SBRs and poor settleability can lead to significant down-time between reaction cycles. Granular biomass is a strategy to address this limitation and has been successfully applied at lab-scale to treat abattoir wastewater with COD, TN and TP removal of was 98%, 97% and 98%, [44]; and 68%, 86% and 74%, respectively [45]. The results of these studies indicate that flocculent SBRs can be converted to granular SBRs by reducing the settle time.

Lab studies into SBRs emphasise the importance of biodegradable COD [43]. At longer SRT there is less biodegradable COD available for denitrification [3]. Therefore the performance of an SBR can depend on the sludge retention time (SRT) of the anaerobic pre-treatment step. Current research into SBR technology is investigating a high rate aerobic process where the hydraulic retention time (HRT) is reduced to 12 hours and the sludge retention time (SRT) to 2 days (A.ENV.0150). Under these conditions the SBR has achieved approximately 85% COD removal, 57% nitrogen (N) removal and up to 80% phosphate (P) removal. The efficiency of nutrient removal from the wastewater is considerable at such a short SRT. The removed was predominantly converted to biomass (through growth and/or accumulation), rather than oxidation, suggesting aeration requirements could be substantially reduced (corresponding to lower electricity requirements) compared to current aerobic/SBR operations.

3.3 Nitritation/Denitritation

Nitritation/denitritation is a concept introduced to reduce COD requirements and aeration costs associated with nitrogen removal. Nitritation/denitritation is a short cut in the nitrogen cycle based on the observation that nitrite is an intermediary compound in both nitrification and denitrification. Therefore, the partial nitrification of ammonium to nitrite and the subsequent denitrification from accumulated nitrite, instead of from nitrate should be feasible. In order for nitritation and denitritation to occur, nitrite oxidation should be controlled without affecting the ammonia-oxidizing bacteria (AOB) and denitrifying microorganisms must be adapted to high concentrations of nitrite. Theoretically, this process saves up to 25 % of the oxygen demand, up to 40 % of the carbon source, up to 30% of sludge production and reduces CO_2 emissions by 20 % compared to a conventional nitrification/denitrification process [46].

Laboratory studies have not reported significant technical barriers applying the nitritation/denitritation pathway to treat slaughterhouse wastewater with COD, N and P removal of 95%, 97% and 98%, reported respectively [23]. The implementation of the nitrite pathway also significantly reduced the demand for carbon addition to the BNR process [23]. Effluent treated using the nitritation/denitritation is reportedly suitable for irrigation [47], depending on discharge limits and land availability.

A pilot scale implementation of nitritation/denitritation is presented in MLA/AMPC project ENV.044. The final report for this project discusses the potential for full scale implementation.

3.4 Constructed Wetlands

Constructed wetlands can be used as a bio-filter for wastewater treatment. The constructed wetland system can remove solids, COD, nitrogen and phosphorus. Constructed wetland systems should be considered as one component within a treatment train. A schematic set-up of a constructed wetland system is shown in Figure 14.

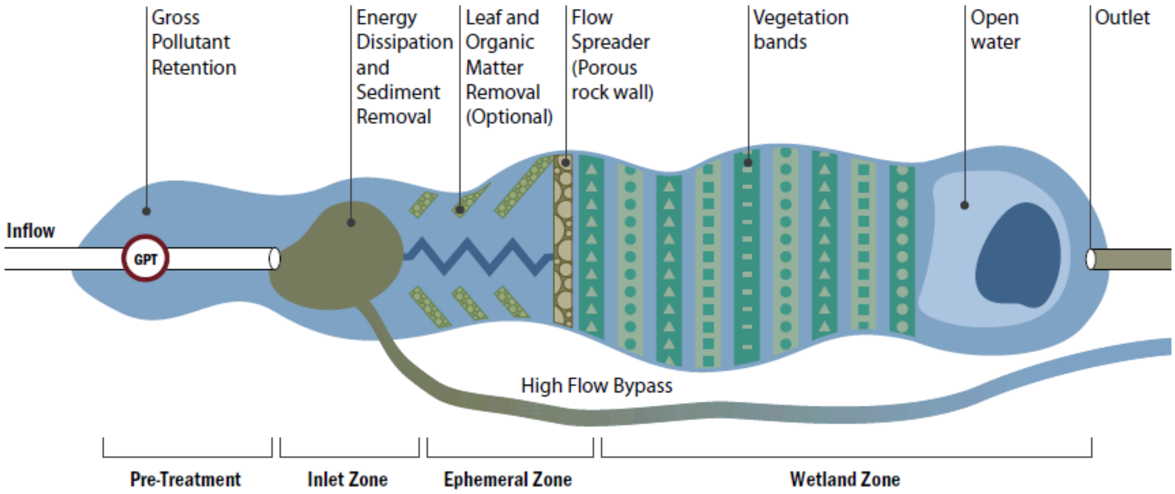


Figure 14: Schematic set-up of a typical constructed wetland (from Constructed wetlands systems design guidelines for developers 2005).

Experimental trench systems containing *Typha* (two species), *Phragmites* and *Scirpus* plants in a gravel substrate have been applied to treat effluent from poultry abattoirs. However, nutrient removal efficiencies were generally poor at 83-89%, 4.0 – 56% and 34 – 61%, for suspended solids, total nitrogen and total phosphorus respectively [48].

Rivera [49] reported some success using a two stage wetland system to treat abattoir wastewater (Pachuca, Mexico) with mean removal efficiencies for COD, SS and organic nitrogen of 87.4%, 89% and 73.6%. However removal rates for inorganic nitrogen such as NH_3-N and NO_3-N were generally poor. Rivera estimated a constructed wasteland with horizontal subsurface flow to treat 30kL per day would require a trench length of 960 – 1125 m [49]. The wastewater production expected from Australian slaughterhouses is in the range of 1000-3000 kL per day and could require a trench length in the range of 50-100 km.

Constructed wetlands are not considered as an option for slaughterhouse wastewater due to the relatively poor ammonium/nitrogen removal, the relatively poor phosphorus removal and the large footprint.

3.5 Stripping

Ammonium ions (NH_4^+) are converted to volatile ammonia gas (NH_3) at a pH of approximately 10.5 – 11.5. The volatile ammonia gas can be stripped from the wastewater using a stripping tower (Figure 15) however large volumes of air are often needed. Ammonia stripping will consume large quantities of caustic (to raise the pH above 10.5) and acid (e.g. H_2SO_4, used to wash the gas and produce $(NH_4)_2SO_4$). Lower concentrations of ammonium in the wastewater and lower discharge limits for the treated effluent will increase the air requirements and the energy demand. Ammonia stripping is generally considered economically feasible where the concentration >3 gNH_4-N/L; this is an order of magnitude higher than concentrations expected for slaughterhouse wastewater (200-300 mg/L). However, ammonia stripping will release odour and may breach EPA guidelines, therefore this process is not considered appropriate for slaughterhouse wastewater [50] and will not be investigated further.

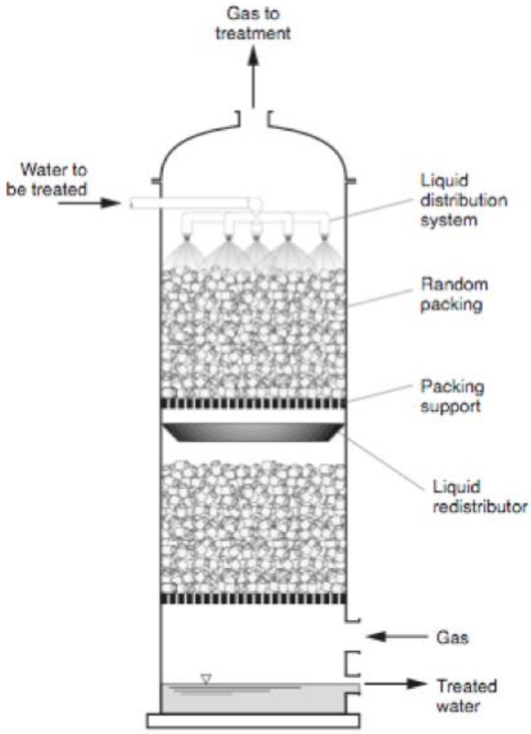


Figure 15: Schematic stripping tower where air in pumped into the bottom of the tower to remove volatile components of contaminated water added to the top of the tower (Crittenden et al 2005).

3.6 Crystallisation

Crystallisation refers to technologies for recovery of nitrogen and phosphorus through precipitation of compounds such as struvite ($MgNH_4PO_4.6H_2O$). This is an emerging technology option, rather than an established process in the Australian meat processing industry. Struvite precipitation is targeted towards P recovery, rather than N recovery. Struvite is a highly effective fertilizer that has a phosphorous content competitive with most commercial fertilizers, and requires only magnesium dosing, which removes phosphorous at a net cost of $1/kg P, compared to approximately $11/kg P for iron or alum dosing. Given the fertilizer value of phosphorous at $5/kg P, there is a substantial driver for phosphorous recovery.

When considering struvite crystallisation in the context of nitrogen removal, it is a chemical process and is relatively fast compared to biological nutrient removal processes. MLA/AMPC project

A.ENV.0154 determined a HRT of 1-2 hours was sufficient for the process. This corresponds to a nitrogen loading rate of 3-5 kgN/m^3/day for slaughterhouse wastewater. However, phosphorus is generally the limiting compound in struvite crystallisation. The ratio of nitrogen and phosphorus in Australian slaughterhouse wastewater is generally greater than 5:1 (adapted from Table 13 in Section 4.1.1). Considering the elemental composition of struvite, complete removal of P would result in removal of approximately 10% of N from the wastewater. Therefore struvite crystallisation is not suitable as a standard alone technology for N removal, but may provide significant benefits to processing plants where P removal is required.

Literature and data around application of crystallisation for nutrient recovery in the meat processing industry is discussed further in the final report to MLA/AMPC project A.ENV.0154.

3.7 Technology Comparison

A brief comparison of nitrogen removal technologies is included in Table 11. Benefits of AAR are expected to include reduced aeration costs, higher volumetric loadings (and reduced footprint) and lower sludge production. This will be investigated further in some case study based analysis in Section 5.

Table 11: A comparison of the process features of different anammox technologies.

		AAR	Nitrification/ Denitrification	Stripping	Wetlands	Crystallization
Volumetric loading rates	$kgN/m^3/d$	0.7-2.0	0.1-0.3	TBC	TBC	3-5
Performance TN removal	%	85-90% TN	Over 95% TN	TBC	Up to 70% TN	TP removal above 90%, but TN removal <20%
Energy demand	kWh/kgN removed	1.0-1.8	4.6	TBC	N/A	TBC
Chemical Costs	$/kgN removed	-	-	TBC	-	TBC
Sludge Production	kgTSS/kgCOD	~5%	20-40%	N/A	N/A	N/A
Start-up	months	Up to 4 months		Less than 1 month		Less than 1 month
Other process issues		Poor tolerance to FOG Tolerates elevated NO_2	TBC	TBC	TBC	TBC

4 Application of AAR to Slaughterhouse Wastewater

4.1 Implementation of AAR into Current Treatment Train

The wastewater treatment train used in the Australian red meat processing industry is shown in Figure 16. The current treatment train generally only considers the mainline wastewater, with coarse solids removed via dewatering processes prior to primary treatment and often transported off-site for treatment (usually composting). Anammox could potentially be integrated into the current mainline treatment train after primary screening (to remove TSS and FOGs) or after anaerobic treatment (to remove organic material), as shown in Figure 16. Suitability of these options will be discussed in the following sections.

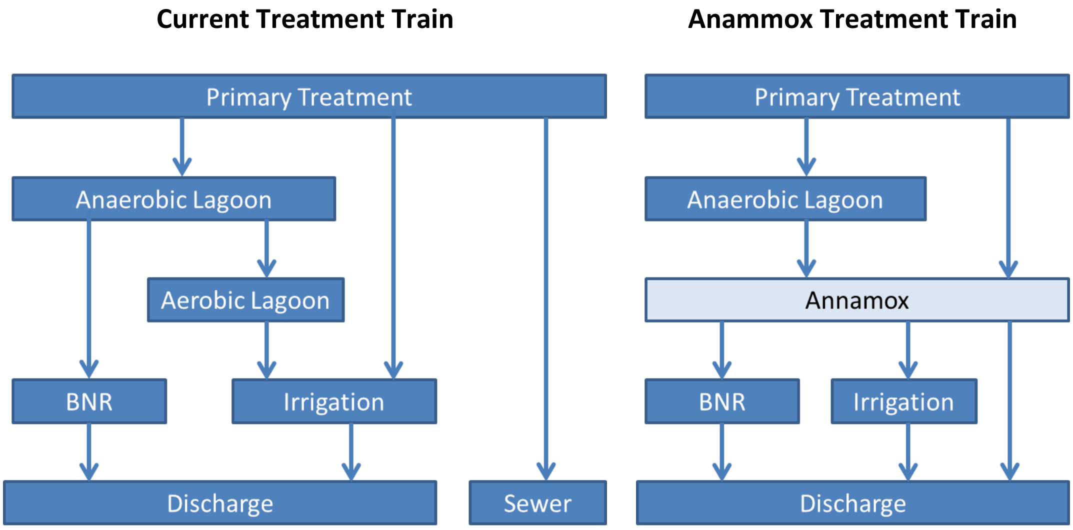


Figure 16: Integration of anammox into current process treatment train after primary treatment or anaerobic treatment

4.1.1 AAR applied to Mainline Wastewater after Primary Treatment

Assessment of AAR after primary treatment is based on treatment of raw wastewater after pre-treatment with a basic screen and a DAF to assist in removal of FOGs and suspended solids. Table 12 summarises the composition of combined wastewater (after primary treatment) reported in literature and measured from 6 Australian red meat processing facilities in 2012 and 2013 (A.ENV.0131 and A.ENV.0151).

Table 12: Production volume and chemical analysis of combined slaughterhouse wastewater in Australia (A.ENV.0151)

	Volume	Temp	TCOD	sCOD	TS	FOG	N	NH_3-N	P	PO_4-P	S
	kL/d	°C	mg/L	mg/L	mg/L[b]	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Lit [a]	-	-	2,000-10,000	-	-	100-600	100-600		10-100		
Site A	2,189	TBC	12,893	1,724	8,396	2,332	245	58	58	53	27
Site B	3,150	29	9,587	1,970	4,300	783	232	TBC	50	38	20
Site C	2,115	TBC	10,800	890	7,530	3,350	260	62	30	15	37
Site D	2,150	36	12,460	2,220	7,400	1,240	438	38	56	31	40
Site E	1,500	45	10,925	1,195	6,118	1,569	272	31	47	32	54
Site F	165	33	7,613	1,365	4,141	1,572	187	85	29	12	11

[a] Based on [1, 51-53]

Combined meat processing wastewater has a high organic strength with a TCOD consistently in the range of 10,000 mg/L. Resulting in COD:N ratios of over 20:1. Full scale nitritation-anammox reactors function effectively with influent COD/NH_4-N ratios around 1 or lower [30], however anammox activity is suppressed above COD/N ratio above 2.0 [33]. Therefore the COD/N ratio of combined slaughterhouse wastewater represents a challenge for anammox technologies.

In addition to high COD, the high FOG content (1000 to 3000 mg/L) of slaughterhouse wastewater presents a challenge for anammox processes. FOG is known to cause problems with sludge settleability and would therefore impact the performance of SBRs containing floccular biomass. This would be an issue for both anammox processes and conventional nitrification/denitrification technology. High FOG concentrations have been shown to decrease the integrity of granules in high rate anaerobic digestion technologies (e.g. UASBs), it is not clear if the high FOG content would cause similar problems with anammox granules and this is an area for possible investigation.

Recovery of FOGs is an ongoing challenge for Australian meat processors and may be influenced by increasing wastewater temperatures. The melting point of cattle fats varies from 29°C for subcutaneous fat to 46°C for intestinal fat and tallow [54]; the melting point influences the degree of emulsification and FOG particle size in respective DAF units. DAF units are also ineffective at temperatures above 40°C due to poor air solubility at these temperatures [55] (Induced air flotation is an alternative at higher temperatures). The higher FOG in slaughterhouse wastewater is likely due to poor remove of intestinal fat and tallow due to the higher wastewater temperature.

Total nitrogen concentrations in meat processing wastewater are typically 250 to 500 mg/L, while this is relatively low compared to anaerobic digester centrate (800 to 2000 mg/L) the total nitrogen concentration is in the range of where anammox is recommended (>200mg/L). However, slaughterhouse wastewater contains large amounts of organic bound nitrogen with typically only 20-50% in the form of NH_3 (required form for anammox). Therefore biological treatment to release nutrients may be an important step prior to either anammox or conventional BNR processes.

Anaerobic digestion was evaluated in A.ENV.0151 as a potential treatment process to generate energy from the wastewater and release nutrients to facilitate removal or recovery. Results demonstrated that the anaerobic biodegradability of combined Slaughterhouse wastewater was in the range of 90% of COD. Anaerobic treatment of combined slaughterhouse wastewater has also been successful in numerous previous laboratory studies with chemical oxygen demand (COD) removal in the range of 80-95% [1, 56-58]. High levels of subsequent biological nitrogen and phosphorus removal have also been reported [16].

Initially, there appear to be several technical barriers to application of anaerobic ammonium oxidation to mainline slaughterhouse wastewater after primary treatment. However, many of these barriers would also impact conventional BNR technologies.

4.1.2 AAR applied to Mainline Wastewater after Anaerobic Treatment

An alternative placement for AAR is after primary treatment to remove coarse solids and anaerobic treatment to remove organic matter. Table 13 summarises the average composition (recorded over 1 month in 2006) of combined red wastewater and treated effluent from an anaerobic lagoon (MLA/AMPC project ENV.044).

Table 13: Characteristics of Combined Red wastewater and anaerobic lagoon effluent (ENV.044)

Date	pH	Alkalinity mg $CaCO_3$/L	TSS mg/L	TCOD mg/L	SCOD mg/L	TKN mg/L	NH_4-N mg/L	TP mg/L	PO_4-P mg/L
Red Wastewater	7.18	154	1550	5373	1333	196	59.2	32.7	25.0
Anaerobic Lagoon Effluent	7.04	437	655	1097	207	217	196.1	36.0	35.1

After anaerobic treatment approximately 90% of total nitrogen is in the form of ammonium and should be available for anammox (or conventional nitrogen removal processes). The ammonium concentration in Table 13 is at the lower range of concentrations for anammox application. At these feed concentrations ammonium removal efficiencies may be lower than the 90-95% benchmarks listed by anammox technology suppliers, however there is potential to address this through process optimization.

The COD/ NH_4-N ratio in anaerobic lagoon effluent is 5, still significantly above the COD/NH_4-N ratios reported to suppress anammox activity (ratio of 2, [33]). The biodegradable fraction of COD in anaerobic lagoon effluent is expected to be very low, therefore there should not be enough available carbon for heterotrophic denitrifying bacteria to consume nitrite and out-compete anammox bacteria. It is not clear is non-degradable COD will also suppress anammox activity, and this area is still under investigation in the literature.

Anaerobic treatment appears to address many of the challenges/barriers to anaerobic ammonium oxidation of raw wastewater. Therefore, anaerobic lagoon effluent has been identified as the most likely place to implement anammox into existing wastewater treatment at Australian meat processers. However, further investigation is required to assess the impact of non-degradable COD on anammox activity; and to determine removal efficiencies and predict the total nitrogen and ammonium concentrations in anammox effluents compared to discharge limits for the Australian red meat processing industry.

4.2 Integration of Anammox with Developing Technologies

Possible implementation of anaerobic ammonium oxidation into developing wastewater management strategies in the red meat processing industry was based on process configurations developed in two current AMPC/MLA projects being conducted at UQ:

1. A.ENV.0151 NGERS and Wastewater Management – mapping waste streams and quantifying the impacts.
2. A.ENV.0150 Investigating high rate aerobic wastewater treatment with anaerobic digestion and anammox;

Example process configurations incorporating AAR are shown in Section 4.2.2 and Section 4.2.1.

4.2.1 Source Separation and Specialised Treatment of Wastewater (A.ENV.0151)

An example process configuration incorporating AAR into a process using separate specialised treatment technologies for anaerobic treatment and organic removal is shown in Figure 17. In this process configuration, estimating the size and cost of the upstream anaerobic treatments would be highly site specific and therefore were not included in the case study analysis in Section 5. However, as all streams undergo anaerobic treatment and all streams are directed to the AAR process, the combined AAR feed would be similar to CAL effluent.

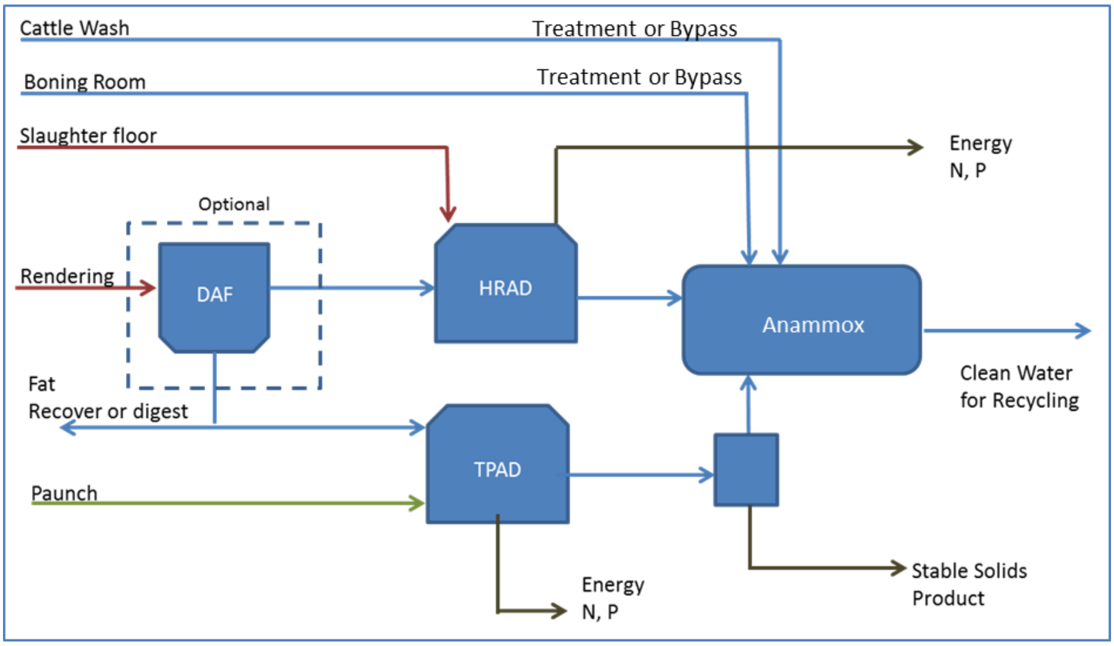


Figure 17: Separation of wastewater streams based on similar properties with primary treating using anaerobic digestion and potential anammox for treatment AD centrate (A.ENV.0151)

4.2.2 High Rate Aerobic Treatment Coupled to Anaerobic Digestion (A.ENV.0150)

An example process configuration incorporating AAR into a novel high rate aerobic process is shown in Figure 18. This process configuration has been included in the case study assessments in Section 5.

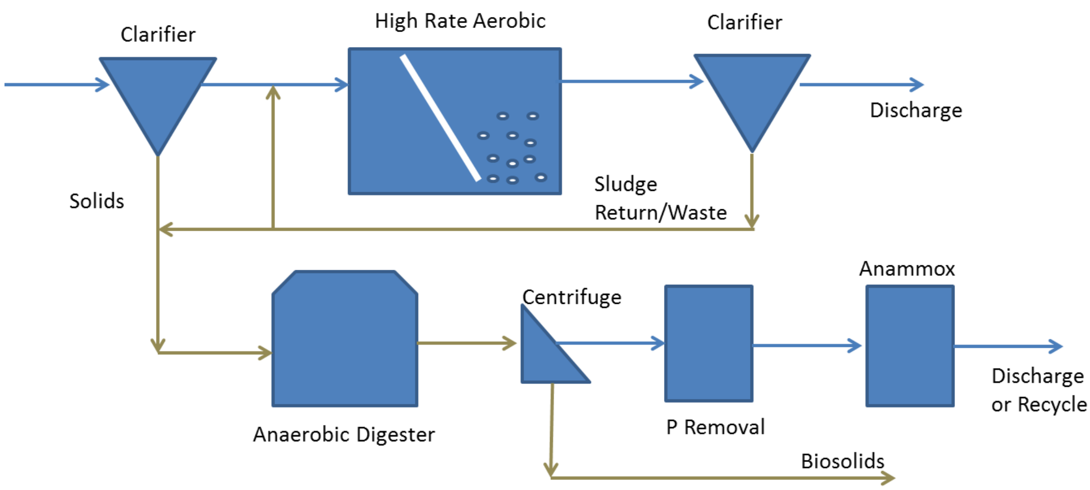


Figure 18: High rate aerobic wastewater treatment and anaerobic digestion with anammox integrated for treatment of side stream AD centrate (A.ENV.0150)

5 Case Studies to Assess AAR and Alternate Nitrogen Removal Technologies

This section includes a basic assessment of ARR applied in the Australian red meat processing industry against the more conventional nitrification/denitrification process and the developing high rate aerobic nitrogen removal technology.

The costing information in this analysis is based on order of magnitude estimates and is not intended as a detailed feasibility analysis; it is intended as an indication of the relative contributions of the organic removal and nitrogen removal steps to capital and operating costs.

Capital costs are generally estimated using plant/vessel size and a linear cost basis. However, there are likely some economies of scale, particularly for larger process vessels. Final vessel cost will be dependent on final design, construction material selection/availability (e.g. concrete, stainless steel, mild steel, glass panelling) and local suppliers or contractors.

5.1 Basis used in Case Study Analyses

The case study used to examine nitrogen removal technologies is based on treatment of the combined wastewater for a processing plant after primary solids removal and before anaerobic treatment, the cost associated with the anaerobic treatment and the value of biogas recovered is included in the assessment. The analysis is based on an averaged sized processing facility processing 600 head of cattle per day, with total effluent flow of 1.7 ML per day. Inputs are based on nutrient and organic contaminant production (per THSCW) as reported in recent MLA and AMPC projects (A.ENV.0131 and A.ENV.0151).

Each treatment technology has been developed to achieve a total nitrogen discharge of approximately 50 mg/L this corresponds to approximately 80% total N removal. Trade waste discharge fees or the cost of irrigation are not included in the analysis as the effluent quality is similar for each process, the costs associated with final discharge would be similar.

Table 14: Wastewater flow, concentration and load for case study to assess anammox and process alternatives

	Concentration	Load
Production level		600 head d^{-1}
Wastewater Volume		1730 kL d^{-1}
COD	10,000 mg/L	17,300 kg/d
Solids	3,480 mg/L	10,000 kg/d
Nitrogen	250 mgN/L	936 kg/d
Phosphorous	50 mgP/L	144 kg/d

Phosphorus (P) recovery using struvite crystallisation ($NH_4MgPO_4{\cdot}6H_2O$) is an emerging technology option that may be integrated into the treatment process where P removal is required. The costs and value recovery around struvite crystallisation would be similar for each process included in this analysis; therefore the specific costs around P recovery are not included. However, the process flowsheets demonstrate where the P recovery unit could be placed in each process.

5.2 Covered Anaerobic Lagoon with Nitrification/Denitrification

Treatment using a covered anaerobic lagoon (CAL) followed by aerated lagoons or SBRs for nitrification and denitrification (BNR) is currently the most commonly applied process configuration for treatment of slaughterhouse wastewater. Therefore, treatment using a CAL and BNR will represent the default treatment option (Figure 19). The specific process assessed in this report was developed in ENV.044. In this process, approximately 20% of raw wastewater is diverted past the CAL to provide a carbon source for the denitrification step, pre-fermentation can be used to produce VFA and assist in P removal. Alternatively, an external carbon source such as methanol could be supplied; this would result in significant chemical consumption costs and is not considered in this analysis. The nitrification/denitrification steps will produce waste sludge that requires treatment and disposal; this could be done in the CAL, in a separate in-vessel digester or off site.

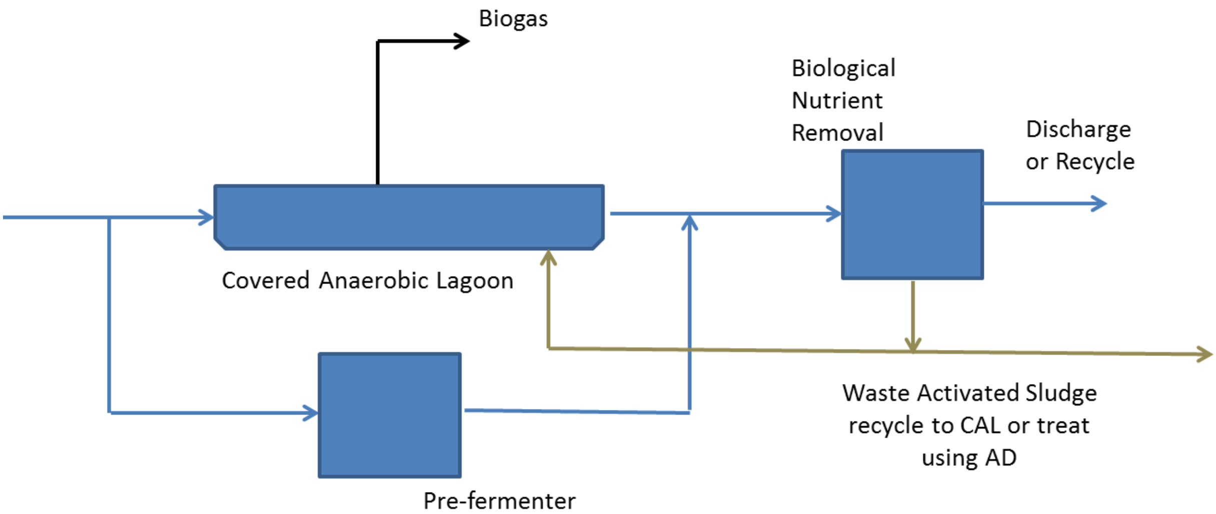


Figure 19: Process flowsheet representing a covered anaerobic lagoon followed by nitrification/denitrification in an SBR. The process is similar to that presented in ENV.044.

The covered anaerobic lagoon is sized based on a hydraulic retention time of 30 days. The anaerobic biodegradability of the organic material is 90% (based on findings from MLA/AMPC projects A.ENV.0131 and A.ENV.0151). CAL efficiency is set to 80% of degradable COD. Where the WAS is recycled through the CAL, the anaerobic biodegradability of the WAS is estimated at 40%.

The nitrification/denitrification is based on the BNR pilot plant designed and operated in MLA project ENV.044. The SBR operated at a HRT of 2 days. Results from ENV.044 demonstrate this is sufficient for COD and N removal at 90%.

Sludge production was calculated based on a yield of 0.4 kgSS/kgCOD feed, which is high for BNR processes. The composition of activated sludge produced in the BNR was 0.08 kgN/kgCOD. Energy demand was calculated as 4.6 kWh per kgN (removed as N_2) and 1 kWh/kgCOD that was oxidised.

For the CAL, capital costing was estimated at \$10 per m^3 for excavation, \$20 per m^2 for pond lining and \$25 per m^2 for the pond cover (*personal communication, Dr Stephan Tait*). For the BNR, capital costing was estimated at \$800 per m^3 tank volume (*personal communication, A/Prof Damien Batstone*).

Ancillary costs include foundations, pumps, piping and instruments and were correlated with the capital cost of the process vessels. Operating costs are estimated based on current pricing, including electricity at $0.1/kWh, personnel at $80,000 per full time equivalent, maintenance of 2-4% of initial capital per annum. Value recovery is based on cogeneration efficiency of 0.35 and $0.1/kWh, this corresponds to a gas value of $10/GJ.

A summary of capital costs for a CAL and BNR process is shown in Table 15, a summary of operating costs for a CAL and BNR process is shown in Table 16. The operating expenses shown in Table 16 do not consider the costs of sludge disposal which may be in the range of $30 per tonne. Processors may operate the SBR to minimise sludge production, if the sludge yield is reduced to 0.2 kgSS/kgCOD added the aeration costs would increase by $75,000 per year to oxidise additional COD and maintain the COD and N removal rates.

Table 15: Summary of capital costs for a covered anaerobic lagoon followed by nitrification and denitrification

	Basis	Estimated Capital
Covered Anaerobic Lagoon	6912 m^2 area and 6 m depth	$726,000
Cogeneration Unit	562 kW @ $1,500/kW	$844,000
BNR	3460 m^3 @ $800/$m^3$	$2,765,000
Installation and ancillaries		$344,000
Engineering Costs	10% of capital	$468,000
Total Estimated Capital		$5,147,000

Table 16: Summary of operating costs for covered anaerobic lagoon followed by nitrification and denitrification

	Basis	Estimated Expenditure
Operator support	0.35 FTE at $80,000	$27,000
Vessel and pipe maintenance	2-4% capital	$119,000
CAL energy demand	0.01 kWh per m^3 per day	$15,000
BNR energy demand	4.6 kWh/kgN and 1kWh/kgCOD	$58,000
Electricity generation from Co-gen	$0.1 per kWh	-$493,000
Renewable energy credits	$0.034 per kWh	-$172,000
Total Estimated Operating		-$446,000

Initial calculations were based on transport of waste sludge offsite for processing (costs not considered), additional calculations were conducted where the waste sludge was recycled into the CAL for treatment. Recycling the waste sludge to the CAL had little impact on capital cost due to the relatively low volume. Interestingly, recycling the waste sludge into the CAL also had little impact on the overall operating costs as the increased methane production from sludge degradation offset increased aeration demand from recycling the sludge COD into the BNR.

5.3 Covered Anaerobic Lagoon coupled to Anaerobic Ammonium Removal

The process configuration recommended for application of AAR to treat slaughterhouse wastewater is a covered anaerobic lagoon (CAL) to remove organic contaminants, followed by AAR in an SBR style reactor. A simplified process flowsheet is presented in Figure 20.

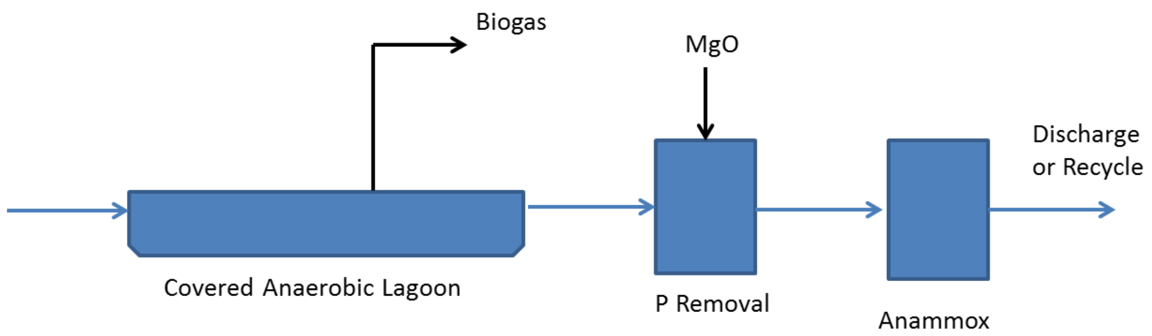


Figure 20: Process flow sheet representing covered anaerobic lagoon followed by anaerobic ammonium removal; Phosphorus removal is optional and is not included in cost calculations.

Design and costing of the covered anaerobic lagoon is similar to Section 5.2. The anaerobic biodegradability of the raw slaughterhouse wastewater is 90%. CAL efficiency is set to 80% of degradable COD. Therefore 72% of COD entering the pond is converted to biogas. This this process, the CAL would be approximately 20% larger as raw wastewater is no longer diverted to the nitrogen removal step.

The AAR process is based on an SBR with a nitrogen loading rate of 0.7 kg/m^3/day. The energy demand for N removal was 1.2 kWh/kgN removed. The effluent quality was set at 25mg/L NH_4^+, 10 mg/L NO_2 and 20 mg/L NO_3 (10% of NH_4^+ removed) as discussed in Section 2.1.4, this results in an effluent concentration of 55 mg/L total nitrogen. In addition to N removal, the AAR reactor was assumed to oxidise 20% of the COD feed. Energy demand for the COD removal was calculated at 1 kWh/kgCOD removed. For the AAR reactor, capital costing was estimated at \$800 per m^3 tank volume.

A summary of capital costs for a CAL and AAR process is shown in Table 17, a summary of operating costs for a CAL and AAR process is shown in Table 18.

Table 17: Summary of capital costs for anaerobic ammonium removal coupled to covered anaerobic lagoon

	Basis	Estimated Capital
Covered Anaerobic Lagoon	8640 m^2 area and 6 m depth	$907,000
Cogeneration Unit	703 kW @ $1,500/kW	$1,055,000
Anammox Reactor	555 m^3 @ $800/m^3	$444,000
Installation and ancillaries		$186,000
Engineering Costs	10% of capital	$259,000
Total Estimated Capital		$2,851000

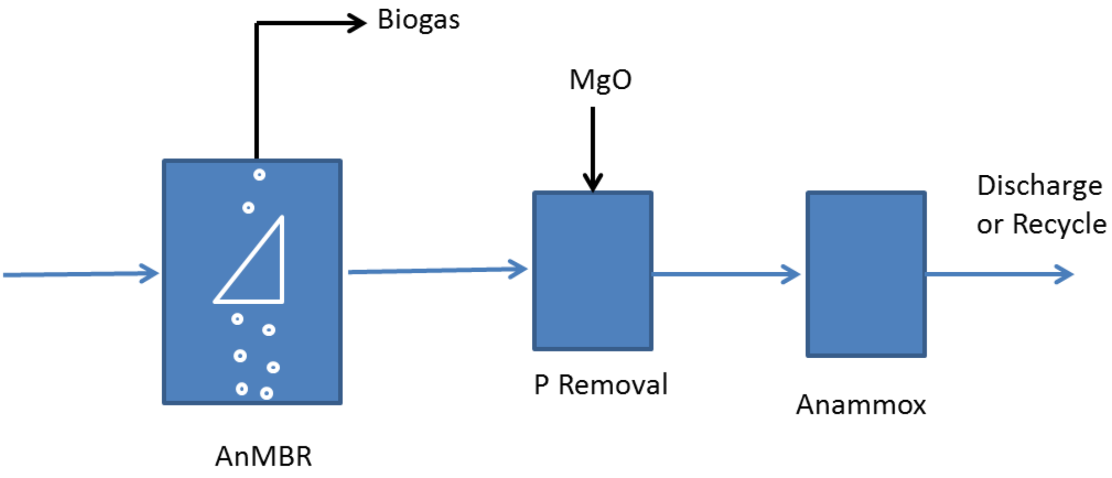
	Basis	Estimated Expenditure
Operator support	0.35 FTE at $80,000	$27,000
Vessel and pipe maintenance	2-4% capital	$83,000
CAL energy demand	0.01 kWh per m^3 per day	$19,000
AAR energy demand	1.2 kWh per kgN removed	$28,000
Electricity generation from Co-gen	$0.1 per kWh	-$616,000
Renewable energy credits	$0.034 per kWh	-$216,000
Total Estimated Operating		-$675,000

Ancillary costs include foundations, pumps, piping and instruments and were correlated with the capital cost of the process vessels. Operating costs are estimated based on current pricing, including electricity at \$0.1/kWh, personnel at \$80,000 per full time equivalent, maintenance of 2-4% of initial capital per annum. Value recovery is based on cogeneration efficiency of 0.35 and \$0.1/kWh, this corresponds to a gas value of \$10/GJ.

5.4 Anaerobic Membrane Bioreactor coupled to Anaerobic Ammonium Removal

Anaerobic membrane bioreactors (AnMBR) are emerging as an alternative technology to CALs for treatment of organic materials. AnMBRs are a high rate anaerobic technology that utilise a membrane to retain biomass and residual substrate within the reactor. The membrane separates the hydraulic retention time and the solids retention time, as a result AnMBRs are able to operation at very short hydraulic retention times compared to CALs.

In this process configuration an AnMBR is used to remove organic contaminants, followed by AAR in an SBR style reactor. A simplified process flowsheet is presented in Figure 21.



The AnMBR is designed based on a hydraulic retention time of 1 day. The anaerobic biodegradability of the organic material is 90% (based on findings from MLA/AMPC projects A.ENV.0131 and A.ENV.0151). AnMBR efficiency will be higher than a CAL and is set to 95% of degradable COD (based on findings from MLA/AMPC projects A.ENV.0133 and A.ENV.0149). Therefore 86% of COD entering the AnMBR is converted to biogas. The increased efficiency in the AnMBR also results in a greater conversion of organic N and a higher concentration of N transferred to the AAR reactor. For the AnMBR, capital costing was again estimated at \$1,000 per m^3 tank volume including an allowance for membranes.

The AAR process was designed using the guidelines discussed in Section 5.3. The AAR was sized using a loading rate of 0.7 kgN/m^3/d. Energy demand was based on 20% of feed COD being oxidised (1kWh/kgCOD) and 1.2 kWh/kgN removed. The effluent quality was set at 25mg/L NH_4^+, 10 mg/L NO_2 and 22.5 mg/L NO_3 (10% of NH_4^+ removed) as discussed in Section 2.1.4, this results in an effluent concentration of 58 mg/L total nitrogen. For the SBR, capital costing was again estimated at \$800 per m^3 tank volume.

Again, ancillary costs include foundations, pumps, piping and instruments and were correlated with the capital cost of the process vessels. Operating costs are estimated based on current pricing, including electricity at $0.1/kWh, personnel at $80,000 per full time equivalent, maintenance of 2-4% of initial capital per annum. Value recovery is based on cogeneration efficiency of 0.35 and $0.1/kWh, this corresponds to a gas value of $10/GJ.

A summary of capital costs for an AnMBR and AAR process is shown in Table 19, a summary of operating costs for an AnMBR and AAR process is shown in Table 20.

Table 19: Summary of capital costs for anaerobic ammonium removal coupled to covered anaerobic lagoon

	Basis	Estimated Capital
AnMBR	1730 m^3 @ $1000/$m^3$	$1,728,000
Cogeneration Unit	835 kW @ $1,500/kW	$1,252,000
Anammox Reactor	617 m^3@ $800/$m^3$	$494,000
Installation and ancillaries		$218,000
Engineering Costs	10% of capital	$369,000
Total Estimated Capital		$4,061,000

Table 20: Summary of operating costs for anaerobic ammonium removal coupled to covered anaerobic lagoon

	Basis	Estimated Expenditure
Operator support	0.35 FTE at $80,000	$27,000
Vessel and pipe maintenance	2-4% capital	$111,000
AnMBR energy demand	0.4 kWh per m^3 per day	$25,000
AAR energy demand	1.2 kWh/kgN and 1kWh/kgCOD	$23,000
Electricity generation from Co-gen	$0.1 per kWh	-$731,000
Renewable energy credits	$0.034 per kWh	-$256,000
Total Estimated Operating		-$801,000

5.5 High Rate Aerobic Treatment Coupled to Anaerobic Digestion (A.ENV.0150)

High rate aerobic treatment coupled to anaerobic digestion is a technology option designed to treat slaughterhouse wastewater and is currently under development in AMPC/MLA project A.ENV.0150. A simplified process flowsheet is presented in Figure 22 and mainly consists of a sequencing batch reactor (SBR) for carbon removal and partial nutrient removal, an anaerobic digester for solids stabilization, a struvite crystallizer for nutrient recovery and an anammox reactor for effluent polishing to achieve a discharge concentration of approximately 50 mg N/L.

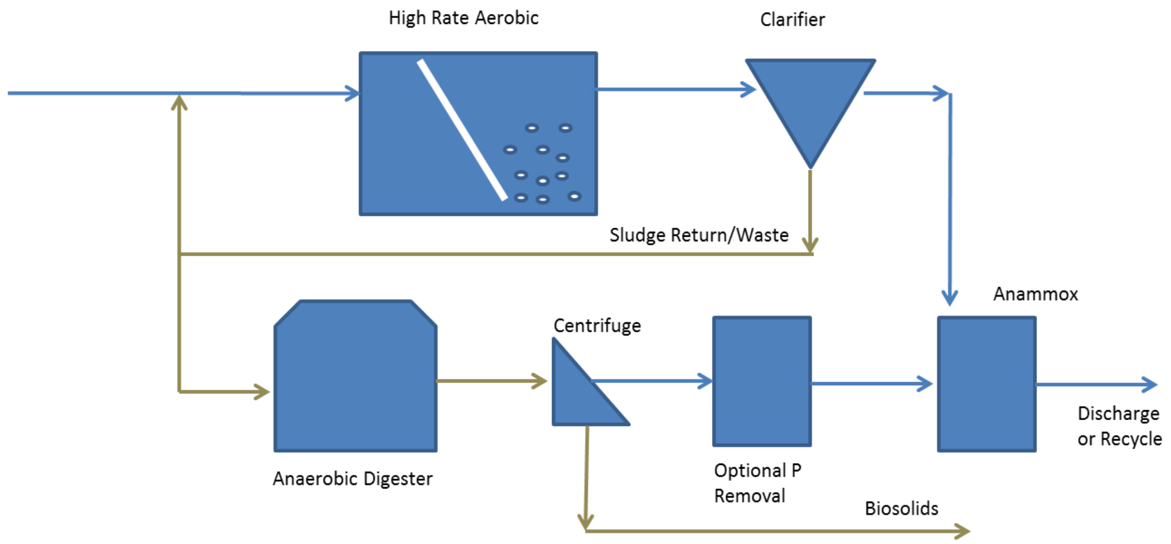


Figure 22: High rate aerobic wastewater treatment and anaerobic digestion with anammox integrated for treatment of side stream AD centrate (A.ENV.0150)

The high rate SBR is not a nitrification/denitrification process; the nitrogen removed through biomass growth only and does not leave the process as N_2 gas. The high-rate SBR is designed to have a hydraulic retention time (HRT) of 0.5 days and sludge retention time (SRT) of 2 days, this is significantly shorter than a conventional SBR for nutrient removal and aims to convert organic matter (measured as COD) into biomass, instead of oxidising it. This will reduce aeration requirements (resulting in lower energy demands) while achieving partial nutrient capture in the biomass growth (e.g. approx. 60% total nitrogen capture and 70% total phosphorus capture). The biomass generated from the SBR is thickened to 4% solids and treated in a mesophilic anaerobic digester (37°C and 12 day HRT), where approximately 60% of the biomass is converted to biogas. The stabilized solids stream is dewatered by centrifuge, with the solids cake being transported for land application, the cost of transport and land application is not included in this analysis.

The effluent streams from the high rate SBR and the digester are combined for further treatment using anaerobic ammonium removal. The AAR process is designed according to Section 5.3.

Again, ancillary costs include foundations, pumps, piping and instruments and were correlated with the capital cost of the process vessels. Operating costs are estimated based on current pricing, including electricity at $0.1/kWh, personnel at $80,000 per full time equivalent, maintenance of 2-4% of initial capital per annum. Value recovery is based on cogeneration efficiency of 0.35 and $0.1/kWh, this corresponds to a gas value of $10/GJ.

A summary of capital costs for a high rate aerobic nitrogen removal followed by an in-vessel anaerobic digester is shown in Table 21, a summary of operating costs is shown in Table 22.

	Basis	Estimated Capital
Anaerobic Digester	2200 m^3@ \$800/m^3	\$2,228,000
Cogeneration Unit	562 kW @ \$1,500/kW	\$844,000
SBR	864 m^3 @ \$800/m^3	\$691,000
Anammox Reactor	358 m^3@ \$800/m^3	\$286,000
Installation and ancillaries		\$293,000

Engineering Costs	10% of capital	$434,000
Total Estimated Capital		$4,776,000

Table 22: Summary of operating costs for high rate nitrogen removal coupled to in vessel anaerobic digestion

	Basis	Estimated Expenditure
Operator support	0.35 FTE at $80,000	$27,000
Vessel and pipe maintenance	2-4% capital	$102,000
Digester mixing energy	0.1 kWh per m^3 per day	$10,000
Dewatering energy		$60,000
SBR energy demand	1 kWh per kg COD	$56,000
AAR energy demand	1.2 kWh per kg N	$11,000
Electricity generation from Co-gen	$0.1 per kWh	-$492,000
Renewable energy credits	$0.034 per kWh	-$172,000
Total Estimated Operating		-$398,000

5.6 Comparison of Nitrogen Removal Options

A comparison of the capital and operating costs of the nitrogen removal processes presented in this report is shown in Table 233. The costing information in Table 23 is not intended as a detailed feasibility analysis; it is intended as an indication of the relative contributions of the organic removal and nitrogen removal steps to capital and operating costs.

The capital costs of CALs appear to be around 20% cheaper when using a BNR process compared to an AAR process, this is because a portion of the raw wastewater was diverted past the CAL to provide a carbon source for BNR. In practice, the CAL would likely be designed to handle the full wastewater volumetric flowrate as a contingency for situations where the BNR process was not in operation. This would remove the capital saving.

The capital costs of AAR processes were considerably lower than the more conventional BNR processes. This was largely due to the difference in nitrogen loading rates and the subsequent difference in vessel size. The N loading rate for the BNR processes was approximately 0.1 kgN/m^3/d and was based on MLA project ENV.044. By comparison the N loading rate for the AAR processes was 0.7 kgN/m^3/d, while this is typical of the nitrogen loading rates achieved in existing AAR installations, the ammonium concentrations are also much higher in the wastewater treated by the full scale implementations. It is not clear if an N loading rate of 0.7 kgN/m^3/d would be achieved in red meat processing plants, however if the loading rate was reduced to 0.35 kgN/m^3/d and the capital cost of the AAR processes increased proportionally, the capital comparison of AAR processes to BNR processes would still be very favourable.

Operating costs for the conventional BNR process was 2-4 times higher than the AAR process; this was due to much higher aeration costs associated with nitrification/denitrification and oxidation of degradable COD. Additionally, the BNR processes required a portion of raw wastewater to bypass the CAL to provide carbon for denitrification; this resulted in a reduction in biogas produced by the CAL and a reduction in the potential revenue recovered. The combinations of these factors suggest a payback of 4-6 years for a CAL + AAR process compared to over 15 years for a CAL + BNR process. Payback relies heavily on the value of recovered energy from the CAL and does not consider interest on the capital.

High rate aerobic treatment coupled to anaerobic digestion is a very different process configuration, in this process the capital required for the in-vessel anaerobic digester is higher than the capital required for the CAL based processes. While the digester treats a concentrated sludge side-stream and has a much smaller volume than the CALs, the reactor cost per volume is much higher. The capital cost of the high rate SBR is much lower than the conventional BNR processes due to the lower HRT, however the high rate SBR also requires an AAR process to treat N released during the digestion step, as a result the capital of the high rate SBR is higher than AAR alone. Operating costs of the high rate SBR and anaerobic digestion process appear relatively poor in this comparison; this was a combination of increased aeration costs (resulting from some COD oxidation) and loss of methane potential (through both COD oxidation and storage of COD in biomass). The high rate SBR and anaerobic digestion process has significant advantages around plant foot print and will continue to improve as process development continues. The economics of this process would also be improved if the mitigation of sludge disposal costs were included in the analysis.

Table 23: Comparison of Nitrogen Removal Case Studies

Parameter	CAL + BNR	CAL + BNR (WAS recycle to CAL)	CAL + AAR	AnMBR + AAR	High Rate SBR +AD + AAR
Organic Removal	$1,858,000	$2,026,000	$2,303,000	$3,454,000	$3,598,000
Nitrogen Removal	$3,289,000	$3,418,000	$549,000	$607,000	$1,178,000
Total Capital	**$5,147,000**	**$5,444,000**	**$2,852,000**	**$4,061,000**	**$4,776,000**
Organic Operating	-$577,000	-$559,000	-$725,000	-$848,000	-$500,000
Nitrogen Removal Operating	$132,000	$204,000	$51,000	$48,000	$101,000
Total Operating	**-$445,000**	**-$355,000**	**-$674,000**	**-$800,000**	**-$399,000**

6 Conclusions and Recommendations

Australian red meat processing facilities generate large volumes of wastewater that require treatment to remove organic and nutrient contaminants in order to comply with water discharge regulations. This project reviewed the application of anaerobic ammonium removal technology against a range of current and developing technologies and processes for treatment of combined slaughterhouse wastewater. A brief summary of reviews outcomes are:

- Biological nutrient removal processes, such as nitrification/denitrification are an established and low risk option and have been applied to slaughterhouse wastewater at laboratory, pilot and full scale achieving nitrogen and COD removal above 90%. Effective, BNR processes have a higher demand for aeration energy, higher production of waste sludge and in some cases may require chemical carbon addition.
- Anaerobic ammonium removal is a relatively new technology that utilises a short-cut in the nitrogen cycle and results in theoretical aeration cost savings of approximately 60%. High nitrogen loading rates reduce both the footprint and the investment costs of AAR in comparison the BNR processes and creates capacity with improved sustainability. AAR is generally targeted towards streams with NH_4-N concentrations >200 mg/L and with low COD and BOD content. However there are limited examples of application to slaughterhouse wastewater and this does result in some risk.
- Ammonia stripping is not considered an appropriate option for treating slaughterhouse wastewater due to high chemical costs, high aeration energy costs and the potential to release odour that may breach EPA guidelines.
- Constructed wetlands are not a viable option for slaughterhouse wastewater due to the relatively poor ammonium/nitrogen removal, the relatively poor phosphorus removal and the high footprint.
- Struvite crystallisation is not suitable as a standard alone technology for N removal, but may provide significant benefits to processing plants where P removal is required.

AAR has emerged as a promising candidate for treating slaughterhouse wastewater. When considering the placement of AAR into the wastewater treatment train, AAR could potentially be integrated into the current mainline treatment train after primary screening (to reduce TSS and FOGs) or after anaerobic treatment (to remove organic material).

There appear several technical barriers to application of AAR to mainline slaughterhouse wastewater directly after primary treatment including the presence of FOG, the high degradable COD/BOD content, the high COD/N ratio and the low fraction of N as ammonium. This application is not recommended; however many of these barriers would also impact conventional BNR technologies.

Anaerobic treatment appears to address many of the challenges/barriers to AAR of raw wastewater. Therefore, anaerobic lagoon effluent has been identified as the most likely place to implement AAR into existing wastewater treatment at Australian meat processers. However, further investigation is recommended to assess the impact of non-degradable COD on anammox activity; and to determine removal efficiencies and predict the total nitrogen and ammonium concentrations in AAR effluents compared to discharge limits for the Australian red meat processing industry.

The concentrations of ammonia, phosphate and chloride present in combined slaughterhouse wastewater are not expected to cause inhibition, while the concentrations of DO and nitrite will be subject to process control, but are also not expected to cause inhibition. However, in addition to specific inhibitors, anammox bacteria are sensitive to high COD with COD/NH_4-N ratios above 2 reported to suppress anammox activity. This will be a challenge for the meat processing industry.

7 References

1. Johns, M.R., *Developments in wastewater treatment in the meat processing industry: A review.* Bioresource Technology, 1995. **54**(3): p. 203-216.
2. Liu, Y.Y. and R.J. Haynes, *Origin, nature, and treatment of effluents from dairy and meat processing factories and the effects of their irrigation on the quality of agricultural soils.* Critical Reviews in Environmental Science and Technology, 2011. **41**(17): p. 1531-1599.
3. Lemaire, R.L.G., *Development and fundamental investigations of innovative technologies for biological nutrient removal from abattoir wastewater*. 2007: University of Queensland.
4. Borja, R., C.J. Banks, and Z. Wang, *Performance and kinetics of an upflow anaerobic sludge blanket (UASB) reactor treating slaughterhouse wastewater.* Journal of Environmental Science and Health - Part A Environmental Science and Engineering, 1994. **29**(10): p. 2063-2085.
5. Caixeta, C.E.T., M.C. Cammarota, and A.M.F. Xavier, *Slaughterhouse wastewater treatment: Evaluation of a new three-phase separation system in a UASB reactor.* Bioresource Technology, 2002. **81**(1): p. 61-69.
6. Li, C.T., et al., *CHEMICAL/BIO-FLUIDIZED BED TREATMENT OF SLAUGHTERHOUSE WASTEWATER.* Journal of Environmental Engineering, 1986. **112**(4): p. 718-728.
7. Manjunath, N.T., I. Mehrotra, and R.P. Mathur, *Treatment of wastewater from slaughterhouse by DAF-UASB system.* Water Research, 2000. **34**(6): p. 1930-1936.
8. Martinez, J., et al., *Treatment of slaughterhouse wastewater*. 1995. p. 99-104.
9. Núñez, L.A. and B. Martínez, *Anaerobic treatment of slaughterhouse wastewater in an Expanded Granular Sludge Bed (EGSB) reactor*. 1999. p. 99-106.
10. Russell, J.M., R.N. Cooper, and S.B. Lindsey, *Soil denitrification rates at wastewater irrigation sites receiving primary-treated and anaerobically treated meat-processing effluent.* Bioresource Technology, 1993. **43**(1): p. 41-46.
11. Sachon, G., *Waste water from slaughterhouses: Management and treatment.* LES EAUX RESIDUAIRES DES ABATTOIRS DE BETAIL GESTION ET TRAITEMENT, 1986. **39**(516): p. 27-45.
12. Sayed, S., L. van Campen, and G. Lettinga, *Anaerobic treatment of slaughterhouse waste using a granular sludge UASB reactor.* Biological Wastes, 1987. **21**(1): p. 11-28.
13. Sayed, S. and W. De Zeeuw, *The performance of a continuously operated flocculent sludge UASB reactor with slaughterhouse wastewater.* Biological Wastes, 1988. **24**(3): p. 199-212.
14. Stebor, T.W., et al. *Operating experience: anaerobic treatment at packerland packing*. in *44th Purdue Industrial Waste Conference*. 1990. Chelsea, MI.
15. Thayalakumaran, N., R. Bhamidimarri, and P.O. Bicker, *Characterisation of aerobic bio treatment of meat plant effluent*. 2003. p. 53-60.
16. Lemaire, R., et al., *A sequencing batch reactor system for high-level biological nitrogen and phosphorus removal from abattoir wastewater.* Biodegradation, 2009. **20**(3): p. 339-350.
17. Beccari, M., et al., *Design of single sludge systems for the treatment of wastewaters with high ammonia content.* TRIB. CEBEDEAU, 1984. **37**(491): p. 387-394.
18. Frose, G. and R. Kayser. *EFFECTIVE TREATMENT OF WASTEWATER FROM RENDERING PLANTS*. 1985.
19. Willers, H.C., et al., *Temperature-dependency of nitrification and required anoxic volume for denitrification in the biological treatment of veal calf manure.* Bioresource Technology, 1993. **43**(1): p. 47-52.
20. van der Star, W.R.L., et al., *Startup of reactors for anoxic ammonium oxidation: Experiences from the first full-scale anammox reactor in Rotterdam.* Water Research, 2007. **41**(18): p. 4149-4163.

21. Kuenen, J.G., *Anammox bacteria: from discovery to application.* Nat Rev Micro, 2008. **6**(4): p. 320-326.
22. Strous, M., J.G. Kuenen, and M.S.M. Jetten, *Key physiology of anaerobic ammonium oxidation.* Applied and Environmental Microbiology, 1999. **65**(7): p. 3248-3250.
23. Lemaire, R., M. Marcelino, and Z. Yuan, *Achieving the nitrite pathway using aeration phase length control and step-feed in an SBR removing nutrients from abattoir wastewater.* Biotechnology and Bioengineering, 2008. **100**(6): p. 1228-1236.
24. Jin, R.-C., et al., *The inhibition of the Anammox process: A review.* Chemical Engineering Journal, 2012. **197**(0): p. 67-79.
25. Strous, M., et al., *The sequencing batch reactor as a powerful tool for the study of slowly growing anaerobic ammonium-oxidizing microorganisms.* Applied Microbiology and Biotechnology, 1998. **50**(5): p. 589-596.
26. Wett, B., et al., *Syntrophy of aerobic and anaerobic ammonia oxidisers.* Water Science and Technology, 2010. **61**(8): p. 1915-1922.
27. Güven, D., et al., *Propionate oxidation by and methanol inhibition of anaerobic ammonium-oxidizing bacteria.* Applied and Environmental Microbiology, 2005. **71**(2): p. 1066-1071.
28. Cema, G., *Comparative Study on Different Anammox Systems*, in *Land and Water Resources Engineering*. 2009, KTH Royal Institute of Technology: Stockholm.
29. Abma, W.R., et al., *Full-scale granular sludge Anammox process*. Vol. 55. 2007, London, ROYAUME-UNI: International Water Association. 7.
30. Joss, A., et al., *Full-Scale Nitrogen Removal from Digester Liquid with Partial Nitritation and Anammox in One SBR.* Environmental Science & Technology, 2009. **43**(14): p. 5301-5306.
31. Abma, W., et al., *Upgrading of sewage treatment plant by sustainable and cost-effective separate treatment of industrial wastewater.* 2010.
32. Dapena-Mora, A., et al., *Evaluation of activity and inhibition effects on Anammox process by batch tests based on the nitrogen gas production.* Enzyme and Microbial Technology, 2007. **40**(4): p. 859-865.
33. Chamchoi, N., S. Nitisoravut, and J.E. Schmidt, *Inactivation of ANAMMOX communities under concurrent operation of anaerobic ammonium oxidation (ANAMMOX) and denitrification.* Bioresource Technology, 2008. **99**(9): p. 3331-3336.
34. Jung, J.Y., et al., *Factors affecting the activity of anammox bacteria during start up in the continuous culture reactor*. 2007. p. 459-468.
35. Van De Graaf, A.A., et al., *Autotrophic growth of anaerobic ammonium-oxidizing micro-organisms in a fluidized bed reactor.* Microbiology, 1996. **142**(8): p. 2187-2196.
36. De Clippeleir, H., et al., *OLAND is feasible to treat sewage-like nitrogen concentrations at low hydraulic residence times.* Applied Microbiology and Biotechnology, 2011. **90**(4): p. 1537-1545.
37. Vlaeminck, S.E., et al., *Nitrogen removal from digested black water by one-stage partial nitritation and anammox.* Environmental Science and Technology, 2009. **43**(13): p. 5035-5041.
38. Jeanningros, Y., et al., *Fast start-up of a pilot-scale deammonification sequencing batch reactor from an activated sludge inoculum.* Water Science and Technology, 2010. **61**(6): p. 1393-1400.
39. Hippen, A., et al., *Aerobic deammonification: A new experience in the treatment of wastewaters.* Water Science and Technology, 1997. **35**(10): p. 111-120.
40. Siegrist, H., et al., *Nitrogen loss in a nitrifying rotating contactor treating ammonium-rich wastewater without organic carbon.* Water Science and Technology, 1998. **38**(8-9 -9 pt 7): p. 241-248.
41. Keller, J., et al., *Nutrient removal from industrial wastewater using single tank sequencing batch reactors.* Water Science and Technology, 1997. **35**(6): p. 137-144.

42. Mittal, G.S., *Treatment of wastewater from abattoirs before land application - A review.* Bioresource Technology, 2006. **97**(9): p. 1119-1135.
43. Pochana, K. and J. Keller, *Study of factors affecting simultaneous nitrification and denitrification (SND).* Water Science and Technology, 1999. **39**(6): p. 61-68.
44. Cassidy, D.P. and E. Belia, *Nitrogen and phosphorus removal from an abattoir wastewater in a SBR with aerobic granular sludge.* Water Research, 2005. **39**(19): p. 4817-4823.
45. Yilmaz, G., et al., *Simultaneous nitrification, denitrification, and phosphorus removal from nutrient-rich industrial wastewater using granular sludge.* Biotechnology and Bioengineering, 2008. **100**(3): p. 529-541.
46. Gustavsson, D., *Biological sludge liquor treatment at municipal wastewater treatment plants–a review.* Vatten, 2010. **66**(3): p. 179-192.
47. Pijuan, M. and Y. Zhiguo, *Development and optimization of a sequencing batch reactor for nitrogen and phosphorus removal from abattoir wastewater to meet irrigation standards.* Water Science and Technology, 2010. **61**(8): p. 2105-2112.
48. Max Finlayson, C. and A.J. Chick, *Testing the potential of aquatic plants to treat abattoir effluent.* Water Research, 1983. **17**(4): p. 415-422.
49. Rivera, F., et al., *The application of the root zone method for the treatment and reuse of high-strength abattoir waste in Mexico.* Water Science and Technology, 1997. **35**(5): p. 271-278.
50. Johns, M. and P. Greenfield. *Nutrient removal from abattoir Wastewater*. in *Proc. of Abattoir waste water and odour management Conference Bris*. 1992.
51. Cowan, J.A.C., et al., *Membrane treatment strategies for red meat abattoir effluents.* Water Science and Technology, 1992. **25**(10): p. 137-148.
52. Mittal, G.S., *Characterization of the effluent wastewater from abattoirs for land application.* Food Reviews International, 2004. **20**(3): p. 229-256.
53. Tritt, W.P. and F. Schuchardt, *Materials flow and possibilities of treating liquid and solid wastes from slaughterhouses in Germany. A review.* Bioresource Technology, 1992. **41**(3): p. 235-245.
54. Yilmaz, M.T., M. Karakaya, and N. Aktaş, *Composition and thermal properties of cattle fats.* European Journal of Lipid Science and Technology, 2010. **112**(3): p. 410-416.
55. Tchobanoglous, G., et al., *Wastewater engineering : treatment and reuse*. 4th ed. / revised by George Tchobanoglous, Franklin L. Burton, H. David Stensel. ed. 2003, Boston :: McGraw-Hill. xxviii, 1819 p. :.
56. Massé, D.I. and L. Masse, *Treatment of slaughterhouse wastewater in anaerobic sequencing batch reactors.* Canadian Biosystems Engineering / Le Genie des biosystems au Canada, 2000. **42**(3): p. 131-137.
57. Del Nery, V., et al., *Long-term operating performance of a poultry slaughterhouse wastewater treatment plant.* Resources, Conservation and Recycling, 2007. **50**(1): p. 102-114.
58. Gannoun, H., et al., *Mesophilic and thermophilic anaerobic digestion of biologically pretreated abattoir wastewaters in an upflow anaerobic filter.* Journal of Hazardous Materials, 2009. **170**(1): p. 263-271.

Attachment 1: Anaerobic Biodegradability of Slaughterhouse Wastewater

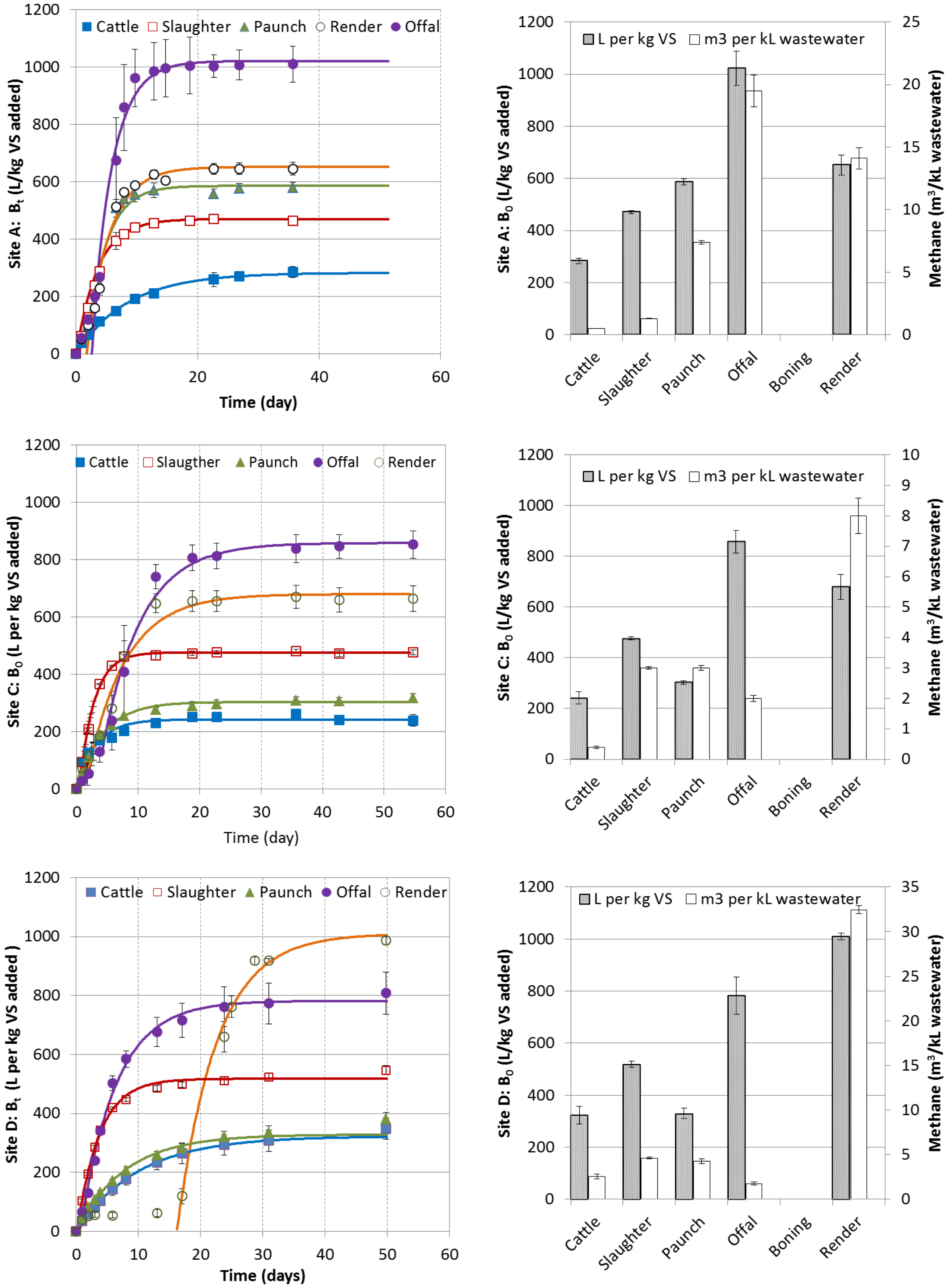


Figure 23: Cumulative methane production over time (Bt) from BMP tests at Site A, C and D; and Summary of B_0 determined from first order model and parameter estimations at Site A, C and D.

Attachment 2: Anaerobic Biodegradability of Slaughterhouse Wastewater

Table 24: Characteristics of Combined red wastewater and anaerobic lagoon effluent (ENV.044)

Red Wastewater									
Date	pH	Alkalinity	TSS	TCOD	SCOD	TKN	NH4-N	TP	PO4-P
		as mg CaCO3/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
13/03/2006	7.05	187.5	2040.0	9150.0	1512.0	239	71.0	46.0	34.8
15/03/2006	6.95	150.0	2400.0	6150.0	1365.0	208	64.7	33.0	26.4
17/03/2006	6.95	150.0	1670.0	5270.0	1596.0	190	54.5	35.0	27.1
20/03/2003	7.86	150.0	1030.0	3220.0	870.0	133	54.9	20.0	13.9
22/03/2006	6.93	125.0	2590.0	7070.0	1305.0	178	51.0	27.0	21.9
24/03/2006	7.41	150.0	0.0	4130.0	1077.0	187	68.9	29.0	23.1
27/03/2006	7.33	165.2	2410.0	4776.0	1557.0	245	65.8	41.0	28.8
29/03/2006	7.05	175.0	1810.0	4752.0	1278.0	188	54.3	33.0	25.0
31/03/2006	7.13	137.5	0.0	3840.0	1437.0	195	48.0	30.0	23.8
Average	**7.18**	**154.5**	**1550.0**	**5373.1**	**1333.0**	**196**	**59.2**	**32.7**	**25.0**
Stdev	**0.30**	**18.9**	**996.3**	**1838.2**	**236.4**	**33**	**8.4**	**7.6**	**5.6**
Anaerobic Pond Effluent									
Date	pH	Alkalinity	TSS	TCOD	SCOD	TKN	NH4-N	TP	PO4-P
		as mg CaCO3/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
13/03/2006	6.95	480	960	1632	193	220	202.0	34.0	35.4
15/03/2006	7.05	388	550	950	202	214	191.8	37.0	35.9
17/03/2006	7.10	425	950	1100	214	223	198.3	40.0	36.1
20/03/2003	7.05	450	450	888	190	228	208.1	36.0	36.0
22/03/2006	7.02	450	560	1002	250	215	196.7	34.0	34.0
24/03/2006	7.04	475	600	1146	225	215	192.0	33.0	33.8
27/03/2006	7.03	413	470	794	196	210	198.4	35.0	34.4
29/03/2006	7.10	425	700	1176	189	219	191.0	37.0	34.9
31/03/2006	7.00	425		1188	200	210	186.8	38.0	35.5
Average	**7.04**	**437**	**655**	**1097**	**207**	**217**	**196.1**	**36.0**	**35.1**
Stdev	**0.05**	**30**	**200**	**242**	**20**	**6**	**6.5**	**2.2**	**0.9**