Biological nutrient removal from meat processing wastewater using a sequencing batch reactor

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Abstract Meat processing effluents are rich in nutrients (nitrogen: 75–200 mg L^{-1} and phosphorus: 20–40 mg L^{-1}) and COD (800–2,000 mg L^{-1}) after primary treatment. A laboratory scale sequencing batch reactor (SBR) was operated for the treatment of a beef processing effluent from slaughtering and boning operations. An effective SBR cycle was found for removal of COD, nitrogen and phosphorus at 22°C. The solid retention time was 15 days while the hydraulic retention time (HRT) was 2.5 days. The total nitrogen in the wastewater was reduced to less than 10 mg L^{-1}, while the total phosphorus decreased to less than 1.0 mg L^{-1}. The residual effluent soluble COD was found to be non-biodegradable as reflected by no further soluble COD removal following prolonged aeration. Removal of biodegradable soluble COD, ammonia nitrogen and soluble phosphate phosphorus of greater than 99% was achieved in the SBR. Good prediction of ammonia and nitrate nitrogen removal was obtained using IWA Activated Sludge Model. The operating cycle is shown to be appropriate to achieve simultaneous removal of COD and nutrients from the meat processing wastewater. Alkalinity and pH have an inverse relationship during the initial anaerobic and aerobic stages due to production and stripping of CO_2. Use of a low level of DO in the final aerobic stage ensured complete ammonia removal and enhanced denitrification.

Keywords Meat processing wastewater; nutrient removal; sequencing batch reactor (SBR)

Introduction

The meat processing industry is one of the major export industries in New Zealand. The processing of meat requires large quantities of potable water, and much of this is discharged as high strength COD and nutrient contaminated wastewater. A typical New Zealand meat processing industry produces up to 10,000 m^3day^{-1} of wastewater with a pollution load equivalent to a city of 60,000–100,000 inhabitants (Bhamidimarri, 1991).

Currently in New Zealand meat-processing wastewater is treated primarily by screens, save-all or dissolved air flotation and secondarily by pond systems or land irrigation. Even though pond systems remove a substantial amount of carbon, nutrient removal is limited (Russell and Cooper, 1992). Effluent nitrogen from the biological treatment is in the form of ammonia and nitrate and both can have adverse impacts when discharged into the environment (Slaney and Van Oostrom, 1997). Although land irrigation can be used because meat processing wastewater contains the required nutrients for plant growth, the nitrate infiltration rate is high through soil and overloading by irrigation may cause groundwater pollution, especially, in the winter season (Russell *et al*., 1993).

The sequencing batch reactor (SBR) activated sludge process is known to have several advantages over conventional continuous flow systems (Irvine and Ketchum, 1989; Wilderer *et al*., 2001). Biological simultaneous nutrient and COD removal is possible by the SBR from meat processing wastewater (Keller *et al*., 1997).

Biological nitrogen removal was achieved in SBR by simultaneous nitrification and denitrification (SND) (Keller *et al*., 1997). Achieving SND in SBR has advantages in terms of less COD requirement and less sludge production. To achieve SND, the dissolved

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oxygen (DO) concentration has to be kept low and at about 0.5 mg L^{-1} the nitrification and the denitrification rates are equal (Münch *et al*., 1996). But, to avoid the filamentous bulking and increase the fat degradation rate, up to 4 mg L^{-1} of DO must be maintained (Travers and Lovett, 1984).

A fundamental requirement for biological phosphorus removal is an anaerobic stage (Okada and Sudo, 1986). Under anaerobic conditions, bio-P bacteria release the stored polyphosphate as orthophosphate (PO_4^{3-}) and convert the readily biodegradable COD (RBCOD) to poly-hydroxy-alkanoates (mainly in the form of poly-hydroxy-butyrate or PHB) that is stored within the cell. On re-exposure to aerobic (or anoxic) conditions, the PHB stored during the anaerobic phase is utilised as a source of organic carbon and energy for polyphosphate storage resulting in a net uptake of phosphorus.

This study was conducted in a laboratory scale SBR to simultaneously remove COD, nitrogen and phosphorus from primary treated meat processing wastewater. The SBR operating conditions were selected based on above discussion. The SBR performance was tested with the ASM1 (Henze *et al*., 1987) model incorporating modifications proposed by Oles and Wilderer (1991) with parameters estimated by experiments and from the literature.

Materials and methods

A laboratory scale SBR was operated for the treatment of a beef processing effluent from slaughtering and boning operations. Primary treated wastewater was collected from Manawatu Beef Packers, Fielding, New Zealand fortnightly and stored in a refrigerator at 4°C. Feeding, mixing, aeration and decanting were controlled by level controllers and a programmable time controller. The DO concentration was controlled by YSI model 57 DO meter with a YSI 57 probe and the recorder port connected to the DO switching box. The DO switching box was able to switch between higher and lower DO levels. The air flow rate was controlled at approximately 0.10 L air (L volume. min)$^{-1}$. A peristaltic pump fed the influent wastewater for 8 minutes in each cycle. The solids retention time (SRT) was kept at 15 days while the hydraulic retention time (HRT) was 2.5 days. The maximum operating volume of the reactor was 15 L. The SBR was located in a temperature controlled room at 22 ± 2°C. The operating conditions of the SBR is shown in Table 1.

Typical characteristics of the influent and effluent were measured at regular intervals of time using *Standard Methods* (1995). The sludge volume index (SVI) was determined from the settled sludge volume in 30 minutes in the reactor during the settling period. Nitrate and nitrite nitrogen and soluble phosphate phosphorus (PO_4^{3-}-P) were measured using ion chromatography (Dionex-100, column type AH9-HC, eluent 9.0 mM Na_2CO_3, flow rate 1.0 ml min^{-1} and the injection volume 25 µl).

Ammonia was measured by manual phenate method (*Standard Methods for the Examination of Water and Wastewater*, 1995). Total phosphorus was measured by ascorbic

Table 1 Operating conditions of laboratory scale sequencing batch reactor (6 hour cycle)

Reactor sequence	(HRT = 2.5 days)	
	Time (hours)	DO level (mg L^{-1})
Fill and non-aerated mix	2.0	0.0
Aerated mix 1	1.0	0.3–0.7
Non-aerated mix react (anoxic)	0.5	0.0
Aerated mix 2	1.0	3.5–4.0
Aerated mix 3 (sludge wasted at the end)	0.5	0.3–0.7
Settle	0.75	0.0
Decant and Idle	0.25	0.0

acid method after digesting with nitric and sulphuric acid (*Standard Methods for the Examination of Water and Wastewater*, 1995). Reactor performance during one cycle was monitored when the reactor was operating under quasi steady-state [reactor mixed liquor volatile suspended solid (MLVSS), effluent soluble COD (SCOD), nitrate nitrogen, ammonia nitrogen (NH_3-N) and PO_4^{3-}-P were stable over three weeks]. Inert COD fraction of wastewater origin and biomass origin, readily biodegradable COD (RBCOD) and maximum specific growth rate of autotrophs were determined in batch tests using the acclimated sludge obtained from the SBR for ASM1 model simulation. Inert SCOD from wastewater origin and biomass origin were determined by a method outlined by Orhon *et al*. (1999). The RBCOD of the wastewater was determined by the method described by Ekama *et al*. (1986). The maximum specific growth rate of autotrophs was measured according to the concept described by Sözen *et al*. (1996).

Results and discussion

The organic loading rate (OLR) during the operating period was between 0.56–0.72 kg COD $(m^3\ day)^{-1}$, comparable to the design values of 0.30–0.35 kg BOD_5 $(m^3\ day)^{-1}$ recommended for activated sludge systems treating slaughterhouse wastewater (Johns, 1995). The average ratio of COD:BOD_5 was reported to be 1:0.5 by Russell *et al*. (1993). The total COD (TCOD):TKN and TCOD:TP ratios were 9 and 46, respectively. These ratios were sufficient to remove nitrogen and phosphorus simultaneously by biological process alone in this study.

The ratio of the effluent SCOD to the influent TCOD was between 0.05–0.07. This ratio is similar to the total inert SCOD found for the influent (4%) and biomass origin (2%). These results suggest complete removal of biodegradable SCOD could be achieved by this operation.

Influent NH_3-N of 63–110 mg L^{-1} was reduced less than 0.8 mg L^{-1} in the effluent which indicates more than 99% NH_3-N removal from the system. Effluent and anoxic nitrite and nitrate were less than 1 and 5 mg L^{-1}, respectively. There was no settling problem observed during this experimental period. The measured SVI was in the range of 100–200 ml $(g\ TSS)^{-1}$.

Influent total phosphorus (TP) of 20–40 mg L^{-1} was reduced to less than 1 mg L^{-1} in the effluent. There was a reduction of the VSS:TSS ratio during the aerobic stage and this may be explained by the phosphorus storage in the aerobic stage with little change in the biomass concentration. The influent TSS of 480–655 mg L^{-1} was reduced to 11–45 mg L^{-1} in the effluent, during the experimental period.

Figure 1 shows SCOD, nitrogen and phosphorus profiles in a cycle study after system reached a quasi steady state. Even though the calculated SCOD was expected to be above 200 mg L^{-1} after the influent feeding time, the observed SCOD was 133.9 mg L^{-1}. This may be attributed to substrate adsorption or absorption and storage of biomass (Van Loosdrecht *et al*., 1997). The effluent SCOD fraction of 0.07 ensured complete removal of the biodegradable COD. Anaerobic hydrolysation rate could be assumed to be very low, as only a slight increase in ammonia concentration was observed during the anoxic/anaerobic period. The effluent ammonia concentration was slightly higher than that at the end of the aerobic period. This might be due to the resynthesis of ammonia from nitrate or ammonia release from endogenous biomass. The nitrite and nitrate nitrogen were less than 1 and 5 mg L^{-1}, respectively during the anoxic phase and good settling of sludge occurred. This is in line with the results of Lakay *et al*. (1999).

The influent ammonia was reduced from 101.8 mg L^{-1} to 0.6 mg L^{-1} in the effluent. The pH as shown in Figure 2 was between 6.9–7.6 during the cycle and therefore less than 2% of free ammonia was expected in the reactor (Anthonisen *et al*., 1976). Since the

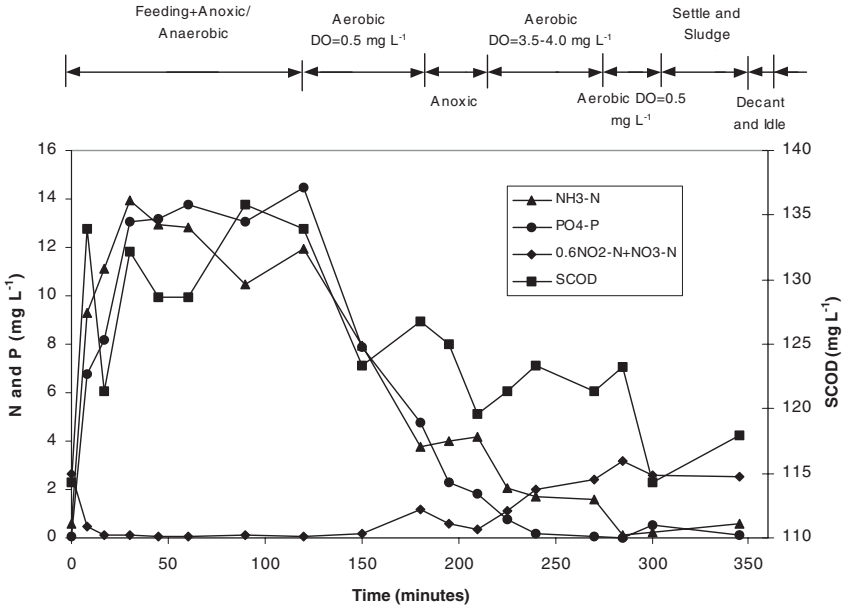

Figure 1 COD, nitrogen and phosphorus profiles during a cycle study

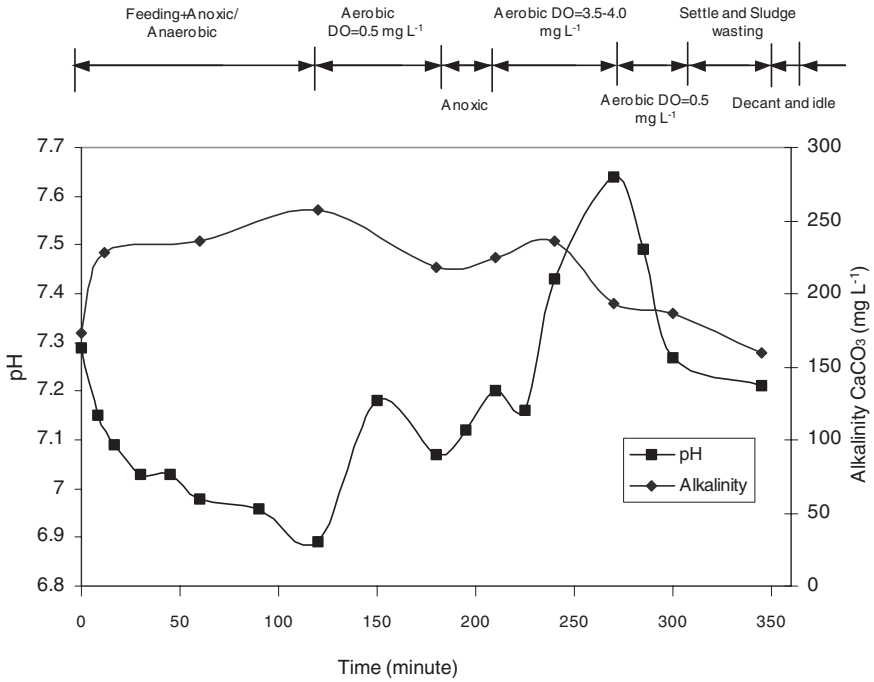

Figure 2 pH and alkalinity profiles in a cycle study

concentration of ammonia was low in the reactor (up to 12 mg L^{-1}), the loss of nitrogen due to ammonia volatilisation was assumed negligible (Philips and Verstraete, 2001). During the first aeration period 8.2 mg L^{-1} NH_3-N was removed, however, only 1.9 mg L^{-1} oxidised nitrogen was formed. This indicates most of the nitrogen was removed by SND. The low initial DO concentration between 0.3–0.7 mg L^{-1} during the first aeration phase permitted the SND (Münch *et al.*, 1996). During the anoxic period ammonia was formed at an

average rate of 0.33 mg NH_3-N (g VSS. h)$^{-1}$ due to anoxic hydrolysation and the nitrate was denitrified at the rate of 0.62 mg NO_3-N (g VSS h)$^{-1}$. The denitrification rate during the anoxic period is comparable to the endogenous denitrification rate, which shows that most of the biodegradable COD was removed prior to this period. The nitrogen balance in the cycle study revealed that 25.8% of influent nitrogen was removed by denitrification during the anoxic periods and 43.3% was removed by SND, probably in the initial aerobic period. Influent nitrogen of 27.9% was incorporated in the waste sludge. A higher percentage of SND indicates that the reactor operation is efficient.

Soluble phosphate phosphorus in the influent was 13.3 mg L^{-1}, which was reduced to 0.1 mg L^{-1} PO_4^{3-}-P in the effluent, that is greater than 99% of phosphorus removal in the system. The maximum phosphorus release rate was 11.4 mg P (g VSS h)$^{-1}$ in the anoxic/anaerobic period and the maximum phosphorus uptake rate was 4.8 mg P (g VSS h)$^{-1}$ in the first aerobic period. The phosphorus uptake rate during the first aerobic period was about 3.55 mg P (g VSS h)$^{-1}$. The anoxic phosphorus uptake rate was 2.14 mg P (g VSS h)$^{-1}$. The reduced anoxic bio-P bacteria activity (60% of aerobic activity) is similar to that suggested in activated sludge model 2d (Henze *et al*., 1999). The anoxic phosphorus uptake reduces the oxygen as well as the COD required for nitrogen removal. The measured sludge phosphorus content at the end of aerobic period (4.6%) confirms enhanced biological phosphorus removal.

The alkalinity profile is shown in Figure 2. The alkalinity during the anoxic/anaerobic period increased mainly due to removal of fermentation products during phosphorus release. During the initial aeration the alkalinity was reduced and this is attributed to nitrification. Figure 2 also shows variation in pH during the cycle. During the anoxic/anaerobic period even though the alkalinity increases, the pH decreases. This is probably due to CO_2 increase in the mixed liquor. The reverse phenomenon occurs during the aerobic period when the alkalinity decreases and pH increases as the CO_2 concentration decreases in the mixed liquor due to CO_2 stripping from the reactor (Dold and Marais, 1987). The increase in the CO_2 concentration in the mixed liquor is due to fermentation, partial functioning of the TCA cycle and utilization of glycogen for energy production during PHB formation (Hesselmann *et al*., 2000). During the anoxic period the pH and alkalinity increase during denitrification.

Total suspended solids were reduced from 505 mg L^{-1} in the influent to 33 mg L^{-1} in the effluent giving 94% removal. The higher DO concentration between 3.5–4.0 mg L^{-1} increases the rate of degradation of slowly biodegradable COD (Travers and Lovett, 1984). The prolonged aeration in starvation condition might have helped to achieve good settling sludge (Wilderer *et al*., 2001). The third aerobic phase at a low DO of 0.5 mg L^{-1} prevented any carryover of DO into the anaerobic stage of subsequent cycle and enhanced the denitification.

Simulation of the results with ASM1 model

Oles and Wilderer (1991) modified the ammonification of the soluble organic nitrogen process rate in aerobic and anoxic conditions in ASM1 (Henze *et al*., 1987) to suit SBR processes. The code for the modified ASM1 model was written in Matlab 5.3. The equations were solved using the function "ode45" for ordinary differential equations systems.

The characterisation of the wastewater shows that 17.0% of the total COD is readily biodegradable and 4% of each inert soluble and particulate COD. The maximum specific growth rate of autotrophs was 0.75 day^{-1}. All kinetic and stoichiometric parameters (except the default values in ASM1) of the model, which were used for computer simulation, are shown in Table 2.

Table 3 provides a summary of influent concentrations and initial conditions used for the

Table 2 Kinetic and stoichiometric parameter values used in ASM 1 simulation

Parameter	Nomenclature	Unit	Value	Reference
$\mu_{max,H}$	Maximum specific growth rate for heterotrophic biomass	day^{-1}	2.0	This work
K_S	Half saturation coefficient for heterotrophic biomass	g COD m^{-3}	8.00	This work
b_H	Decay coefficient for heterotrophic biomass	day^{-1}	0.24	Görgün *et al*. (1995)
$\mu_{max,A}$	Maximum specific growth rate for autotrophic biomass	day^{-1}	0.75	This work
$K_{O,A}$	Oxygen half saturation coefficient for autotrophic biomass	g O_2 m^{-3}	1.20	Henze *et al*. (1987)
b_A	Decay coefficient for autotrophic biomass	day^{-1}	0.10	Henze *et al*. (1987)
k_a	Ammonification rate	m^3 (g COD $day)^{-1}$	0.40	Oles and Wilderer (1991)
Y_H	Yield coefficient for heterotrophic biomass	g COD (g COD)$^{-1}$	0.63	This work
Y_A	Yield coefficient for autotrophic biomass	g COD (g N)$^{-1}$	0.20	Henze *et al*. (1987)

Table 3 Characterised influent concentrations and initial conditions

Parameter	Nomenclature	Unit	Value
Volume exchange rate		% of total volume	10.00
MLVSS	Mixed liquor volatile suspended solids	g L^{-1}	2,750.00
S_S	Readily biodegradable COD	mg COD L^{-1}	295.00
X_S	Slowly biodegradable COD	mg COD L^{-1}	1,300.00
S_I	Soluble inert COD	mg COD L^{-1}	70.00
X_I	Particulate inert COD	mg COD L^{-1}	70.00
S_{NH}	Soluble ammonia nitrogen	mg NH_4-N L^{-1}	101.80
S_{ND}	Soluble biodegradable organic nitrogen	mg L^{-1}	19.00
X_{ND}	Slowly biodegradable organic nitrogen	mg L^{-1}	24.00
S_{NO}	Soluble biodegradable organic nitrogen	mg NO_3-N L^{-1}	0.50
$X_{B,H}$	Heterotrophic biomass	mg COD L^{-1}	0.00

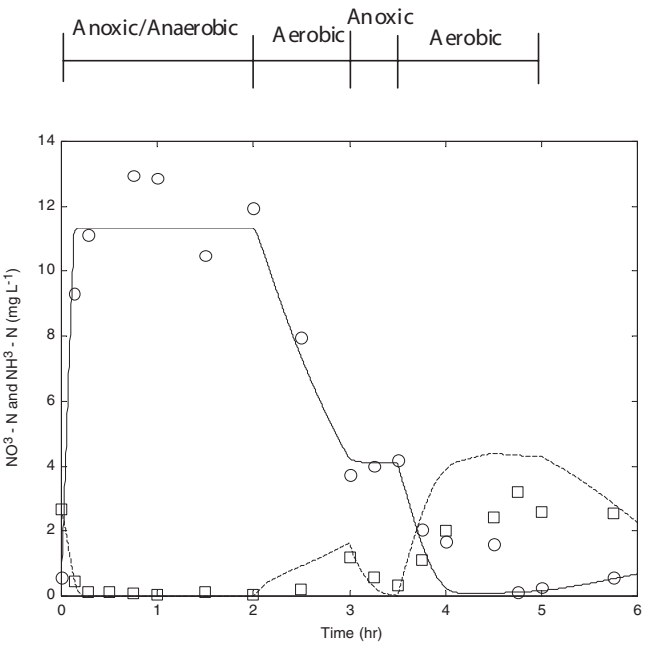

Figure 3 Simulated and experimental NH_3-N (○) and NO_3-N (□) of the SBR cycle

computer simulation of the SBR process. Based on Ekama *et al*. (1986) the total active biomass was calculated to be 68% of the measured VSS at the end of aerobic phase. Nitrifier fraction of 4.5% of the total active biomass gave the best simulation results.

The simulated results with the experimental values are compared in Figure 3. Good prediction of ammonia and nitrate nitrogen removal was obtained using the ASM1 model. The use of low DO level in the third aerobic period ensured complete removal of ammonia and enhanced denitrification.

Conclusions

Removal of biodegradable COD, ammonia nitrogen and dissolved phosphorus of greater than 99% was achieved in the sequency batch reactor. Good prediction of ammonia and nitrate nitrogen removal was obtained using the ASM1 model. Alkalinity and pH have an inverse relationship during the initial anaerobic and aerobic stages due to production and stripping of CO_2. Use of a low level of DO in the final aerobic stage ensured complete removal of ammonia and enhanced denitrification. The operating cycle is shown to be appropriate to achieve simultaneous removal of COD and nutrients from the meat processing wastewater.

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