A Sustainable Technology for the Treatment of Piggery Wastewaters

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Abstract — Twelve 0.6 m woodchip biofilters were used in a laboratory study for the treatment of the mechanically separated liquid fractions of (i) raw pig manure (SR) and (ii) pig manure after anaerobic digestion (SAD). Two loading rates were examined: 5 l/m^2/day (LLR) and 10 l/m^2/day (HLR). The SR and SAD biofilters operated for 390 and 350 days, respectively. Following a start-up period of 60 days the SR woodchip biofilters removed, at the LLR and HLR, respectively, an average of 59 and 43 % suspended solids (SS), 64 and 47 % unfiltered chemical oxygen demand (COD_{uf}), and 60 and 44 % ammonium-nitrogen (NH_4^+-N). Following a start-up period of 70 days, the SAD biofilters removed, at the LLR and HLR, respectively, an average of 56 and 50 % SS, 59 and 51 % unfiltered total nitrogen (TN_{uf}) and 90 and 71 % NH_4^+-N. For both the SR and SAD woodchip biofilters, unfiltered COD, filtered COD and NH_4^+-N removals were higher at the lower loading rate ($P<0.05$). More nitrification occurred at the lower loading rate as indicated by the higher nitrate production in both the SR and SAD woodchip biofilters ($P<0.05$).

Keywords —Anaerobic digestion, biofilters, nitrification, pig manure, woodchips

1 Introduction

The Irish Government's Nitrates Action Plan requires the Irish pig industry to investigate new methods of treating pig farm manure that reduce the need for extensive landspreading[1]. Pig manure contains high concentrations of nutrients with the potential to cause environmental damage to receiving waters; for instance, one of these nutrients, ammonium nitrogen is toxic to fish, and can lead to eutrophication and consequent depletion of dissolved oxygen levels in water [2].

Previous studies suggest woodchips can supply biodegradable carbon substrates that can be used for the balanced growth of beneficial heterotrophic microorganisms in the treatment of wastewaters where excess nitrogen and phosphorus are present. This balanced growth ensures that additional nitrogen (N) is incorporated into microbial cells thus: (i) reducing N concentrations in the filter effluent and (ii) leading to the beneficial slow release of N when the woodchips and microorganisms are spread on land as a soil conditioner [3].

The use of out-wintering woodchip pads has become popular in Ireland and Scotland as an environmentally beneficial means of over-wintering cattle, while also showing welfare benefits for the cattle[4]. Effluent passing through these shallow biofiltration woodchip pads is treated, to a limited degree, due to the physical, chemical and biological processes that occur in the pads [5]. The used woodchips from these pads, along with the trapped manure nutrients and biosolids, are spread on land as a soil conditioner. Analysis of these woodchip pads suggets that successful treatment of high strength piggery wastewater could be achieved using woodchips as biofilter media, and the spent woodchips from the biofilters could be beneficially land-spread, leading to reduced transport costs.

Buelna et al. [6] developed an organic bed biofiltration process (BIOSOR) to treat the liquid effluent and the gaseous emissions from pig farms. In this biofiltration system, biofilm media consisted of 70 % woodchips and 30 % peat. The average pollutant removal efficiency of the BIOSOR biofilter can be 95 % [7]. Even with a large variation in suspended solids concentrations at the inlet, the suspended solids concentrations at the outlet can be under 200 mg/l with a removal efficiency of up to 97 %.

The objective of this study was to investigate, in the laboratory, the use of indigenous woodchips as a biofilter media for the treatment of (i) the separated liquid fraction of raw pig manure and (ii) the separated liquid fraction of pig manure after anaerobic digestion.

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2 MATERIALS AND METHODS

2.1 Design and construction of the woodchip biofilters

The laboratory study provided controlled conditions for examining the nutrient removal efficiency of the woodchip biofilters at two different hydraulic loading rates.

Two steel frame units, each supporting six identical woodchip filters, were placed in a controlled temperature room at 11 ± 2 °C. The woodchips were contained within 225 mm diameter polyethylene pipes ("Corripipe"). A wire mesh base with 10 mm square openings was attached to the base of the Corripipe. The filters were open at the bottom to allow air flow through the filters to maintain an aerobic environment for the microorganisms.

Lodgepole pine (Pinus contorta) woodchips were chosen as the biofilter media. This wood is the second most popular conifer in Ireland, growing on over 65,000 hectares of the Irish Forestry Company (Coillte Teo.) holding land [8]. Studies suggest that lodgepole pine woodchips release biodegradable carbon when immersed in a liquid [3].

The lodgepole pine logs were freshly cut, debarked and chipped with an industrial wood chipping machine. The chips were dried in a drying cabinet for 48 hours to a moisture content of approximately 20%. The woodchips were then sieved and segregated into various sizes 0 – 8 mm, 8 – 14 mm, 14 – 28 mm and greater than 28 mm. An initial filter height of 400 mm was used in each biofilter. The woodchips used were 14 – 28 mm in size. Each filter was initially seeded with two litres of activated sludge taken from a local wastewater treatment plant.

Based on operational results, a 100 mm layer of fine woodchips (0-8 mm) was added to the top of each biofilter on days 88 and day 47 for the SR and SAD biofilters, respectively. A second 100 mm layer was added to each biofilter on day 110 and 69 for the SR and SAD biofilters, respectively. All woodchip biofilters were seeded with 1 litre of activated sludge taken from a local nitrifying wastewater treatment plant immediately after the second layer of woodchips was added.

2.2 Influent wastewater; source and characteristics

The wastewaters treated by the biofilters were (i) the separated liquid fraction of raw pig manure (SR) and (ii) the separated liquid fraction of piggery wastewater after anaerobic digestion (SAD).

2.2.1 Separated raw pig manure liquid (SR)

The raw pig manure was sourced from Teagasc, Moorepark, Fermoy. Co. Cork. The liquid fraction of the raw pig manure (SR) was obtained using a decanter centrifuge (GEA Westfallia Separator UCD 205, Germany). A coagulant - aluminium salt in liquid form (PC31, Celtic Watercare, Cork, Ireland) and a flocculent - a water soluble PAM (C1900P, Celtic Watercare, Ireland) were used to increase the efficiency of separation. PC31 was used in the order of 2.5 litres per m^3 and C1900P was diluted with water to 0.4% by volume and added to the raw pig manure at approximately 17% by volume. The liquid fraction was collected and stored in a controlled temperature room at 11 °C.

Table 1 shows the average nutrient and solids values of the influent SR liquid over the study period. 14 % of the unfiltered total nitrogen (TN_{uf}) was in particulate form, while 80 % of the filtered total nitrogen (TN_f) was in the form of ammonium nitrogen (NH_4^+-N). The rest of the filtered nitrogen comprised total oxidized nitrogen (TON, including nitrite (NO_2^--N) and nitrate (NO_3^--N); 2 %) and soluble organic nitrogen (18 %).

Table 1. Separated raw influent: mean ± standard deviation of various parameters over 390 days of operation.

	Influent mg/l
Suspended Solids	1505 ± 950
Unfiltered COD	2705 ± 1225
Filtered COD	1553 ± 960
Unfiltered TN	1079 ± 615
Filtered TN	924 ± 400
TON	23 ± 23
NO_2^--N	4 ± 7
NO_3^--N	18 ± 19
NH_4^+-N	738 ± 282
pH	7.78 ± 0.73

2.2.2 Separated AD pig manure liquid (SAD)

The anaerobic digestate was generated in a mesophilic anaerobic digester in Roughty Valley Pig Unit, Kilgarven, County Kerry, Ireland. The digestate was passed through a belt press separator. The liquid fraction was collected and stored in a controlled temperature room at 11 °C.

Table 2 shows the average solid and nutrient values of the SAD over the study period. The biochemical oxygen demand (BOD_5) concentration was 3280 ± 563 mg/l, suggesting that over 74 % of unfiltered chemical oxygen demand (COD_{uf}) was not readily biodegradable. The TN_{uf} concentration was similar to the 3800 mg TN/l observed in the liquid fraction of anaerobically digested pig slurry separated by a screw press separator [9]. Approximately 21 % of

TN_{uf} was particulate nitrogen and 84 % of TN_f was NH_4^+-N. The remaining TN_f was present in the form of soluble organic nitrogen and a small quantity (3 %) of oxidised nitrogen.

Table 2. Separated AD influent: mean ± standard deviation of various parameters over 350 days of operation.

	Influent mg/l
Suspended Solids	4224 ± 2249
Unfiltered COD	12812 ± 1979
Filtered COD	6954 ± 1159
Unfiltered TN	3043 ± 442
Filtered TN	2417 ± 736
TON	77 ± 140
NO_2^--N	45 ± 94
NO_3^--N	32 ± 67
NH_4^+-N	2027 ± 714
pH	8.34 ± 0.25

2.3 Hydraulic loading rates

The SR and SAD influents were applied manually twice daily at 9 am and 6 pm. 99 ml (LLR) and 199 ml (HRL) were added to the top surface of the filter media over 2 and 4 minutes, respectively. This loading represented hydraulic loading rates of 5 l/m^2.day (LLR) and 10 l/m^2.day (HLR). Three replicates were examined for each loading rate and each influent type.

2.4 Sampling and analysis

Samples of influent and effluent were taken twice weekly and tested within 24 hours after sampling. The samples were tested for suspended solids (SS), COD_{uf}, COD_f, TN_{uf}, TN_f, NO_2^--N, NO_3^--N and NH_4^+-N. Filtered samples were obtained by passing the sample through Whatman GF/C 1.2 µm glass fibre filter paper. Tests for SS, COD_{uf} and COD_f were carried out in accordance with the standard APHA methods [10]. TN_{uf} and TN_f tests were carried out using Hach TN kits and a DR/2010 spectrophotometer. NH_4^+-N, NO_2^--N and NO_3^--N were measured using a KONELAB 20 Nutrient Analyser (Konelab, Thermo Clinical Labsystems, Vantaa, Finland). Descriptive statistics were used for analysis of influent and effluent means and standard deviations. Data were analyzed with sampling day as a repeated measure using the PROC mixed procedure of SAS [11]. The variables of interest were: SS, COD_{uf}, COD_f, TN_{uf}, TN_f, total oxidized nitrogen (TON), NO_2^--N, NO_3^--N and NH_4^+-N. The fixed effects were filter, loading rate and sampling day. Filter was the experimental unit. Statistical significance was assumed at $P<0.05$.

3 Results and Discussion

3.1 Separated raw manure

Table 3 shows the average concentrations of solids and nutrients in the SR biofilter effluents at the LLR and the HLR over a study period of 330 days. The presence of oxidized nitrogen was observed after 60 days of operation; percentage removals calculated from this time, which was assumed to be the end of the start-up period, are shown in table 3.

Table 3. Separated raw biofilter effluent: mean ± standard deviation of various parameters for over 330 days of operation and average percentage removal after a 60-day start-up period.

	Effluent 5 l/m^2/day mg/l		10 l/m^2/day mg/l		
	(% removal after start-up)				P
Suspended Solids	721 ± 576	*(59)*	933 ± 610	*(43)*	P<0.05
Unfiltered COD	1421 ± 1330	*(64)*	1833 ± 1285	*(47)*	P<0.05
Filtered COD	853 ± 791	*(63)*	1043 ± 841	*(51)*	P<0.05
Unfiltered TN	844 ± 380	*(30)*	854 ± 378	*(30)*	ns
Filtered TN	721 ± 235	*(24)*	749 ± 261	*(25)*	ns
TON	260 ± 179		184 ± 120		P<0.05
NO_2^--N	31 ± 37		36 ± 37		ns
NO_3^--N	228 ± 159		149 ± 94		P<0.05
NH_4^+-N	320 ± 155	*(60)*	443 ± 221	*(44)*	P<0.05
pH	7.75 ± 0.31		7.86 ± 0.36		ns

P values indicate statistical significance between loading rates.

3.1.1 Total suspended solids

Over the first 90 days of operation, 34 % and 26 % of the SS were removed at LLR and HLR, respectively. Following the addition of the fine woodchip layer on day 88, an average of 61 % and 41 % of SS was removed at LLR and HLR, respectively (Table 3). This increase in suspended solids removal suggests that the addition of a layer of fine woodchips increased the solids removal capacity of the biofilters. The increased solids removal may also indicate that there was a start-up period during which most of the solids passed through the system [12]. As biofilm growth and solids trapping occurred, the pore spaces were reduced within the woodchip filters. This in turn resulted in an increase in the solids being trapped in the system. A significant difference was noted between LLR and HLR indicating a higher solids removal at LLR ($P<0.05$).

3.1.2 Chemical oxygen demand

Following the start-up period of 60 days, an average of 64 % and 47 % of COD_{uf} was removed at LLR and HLR, respectively. On average, 63 % and 51 % of COD_f were removed at LLR and HLR, respectively. COD_{uf} and CODf removal rates were higher at LLR than at HLR ($P<0.05$).

3.1.3 Nitrogen

Fig. 1 shows the concentrations of oxidised nitrogen in the biofilter effluents. The presence of oxidised nitrogen indicates the occurrence of nitrification, which is the conversion of NH_4^+-N to NO_2^--N and NO_3^--N [13]. Higher complete nitrification occurred at the LLR, indicated by the higher NO_3^--N production ($P<0.05$). The loading rate did not seem to influence NO_2^--N production ($P>0.05$).

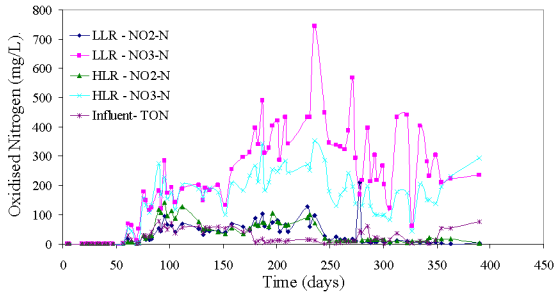

Fig. 1. TON concentrations in influent SR and NO_2^--N and NO_3^--N concentrations in effluent at LLR [13](5 l/m^2/day) and HLR (10 l/m^2/day)

Fig. 2 shows the average NH_4^+-N concentrations in the influent and effluent. The LLR resulted in a higher percentage removal of NH_4^+-N ($P<0.05$).

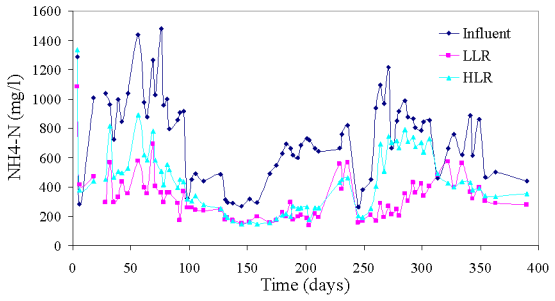

Fig. 2. .NH_4^+-N concentrations in SR influent and effluent at LLR (5 l/m^2/day) and HLR (10 l/m^2/day)

TN_{uf} removals of 30 % were observed at LLR and HLR following the 60 day start-up period. 24 % and 25 % of TN_f were removed at LLR and HLR, respectively. No difference ($P>0.05$) was noted in TN removals between LLR and HLR indicating that the loading rate did not have a significant effect on TN removal for SR. Total nitrogen removal could have occurred through a number of mechanisms filtration, simultaneous nitrification and denitrification; and biomass assimilation.

3.2 Separated AD manure

Table 4 shows the average concentrations of suspended solids and nutrients in the biofilter effluent for the SAD wastewater. The occurrence of nitrification was observed on about the 70 th day of operation. Percentage removals days 70 – 350 are shown in Table 4.

Table 4. Separated AD biofilter effluent: mean ± standard deviation and average percentage removals of various parameters over 280 days of operations after the 70 day start-up period.

	Effluent				
	5 l/m^2/day mg/l		10 l/m^2/day mg/l		
	(% removal after start-up)				P
Suspended Solids	2853 ± 2032	***(56)***	2924 ± 1694	***(50)***	ns
Unfiltered COD	8307 ± 2465	***(44)***	9148 ± 2392	***(38)***	<0.05
Filtered COD	5163 ± 862	***(22)***	5784 ± 599	***(16)***	<0.05
Unfiltered TN	1197 ± 545	***(59)***	1460 ± 657	***(51)***	<0.05
Filtered TN	1012 ± 483	***(55)***	1257 ± 641	***(44)***	<0.05
TON	623 ± 485		527 ± 513		<0.05
NO_2^--N	176 ± 151		258 ± 258		ns
NO_3^--N	446 ± 404		269 ± 320		<0.05
NH_4^+-N	261 ± 191	***(90)***	586 ± 288	***(71)***	<0.05
pH	8.51 ± 0.25		8.52 ± 0.18		ns

P values indicate statistical significance between loading rates.

3.2.1 Total suspended solids

During the first 69 days of operation the average percentage reductions in SS were 5 % and 13 % at LLR and HLR, respectively. Following the addition of the two 100 mm deep layers of fine woodchips, the reductions in SS increased to an average of 57 % and 51 % at LLR and HLR, respectively. The addition of the two layers of fine woodchips had a positive effect on the SS removal. The loading rate had no influence on the removal of SS ($P>0.05$).

3.2.2 Chemical oxygen demand

The COD_{uf} removal increased significantly after Day 70 (Fig. 3) once the two 100 mm deep fine woodchip layers had been added. Effluent from the LLR biofilters contained less COD_f than the HLR biofilters ($P<0.05$), indicating that the lower loading rate resulted in increased degradation of soluble organic matter by microorganisms on the woodchips.

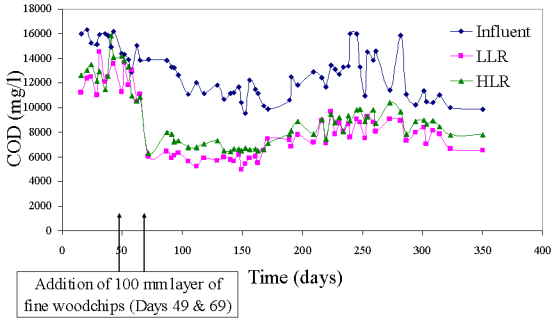

Fig. 3. Unfiltered COD concentrations for SAD influent and effluent at LLR (5 l/m^2/day) and HLR (10 l/m^2/day)

3.2.3 Nitrogen

On average, 261 mg NH_4^+-N/l and 586 mg NH_4^+-N/l were observed in the filter effluent at LLR and HLR, respectively (Table 4), indicating 87 % (LLR) and 71 % (HLR) removals of NH_4^+-N from the SAD wastewater over the 350-day study period. An average of 90 % (LLR) and 71 % (HLR) removal of NH_4^+-N occurred after the start-up period (days 70 – 350).

Table 2 indicates that an average of 626 mg/L of TN in the influent was in particulate form. Approximately 185 mg/L and 203 mg/L of particulate TN were detected in the filter effluent at LLR and HLR respectively (Table 4).

Fig. 4 shows the concentrations of NH_4^+-N present in the influent and effluent SAD liquid over the study period.

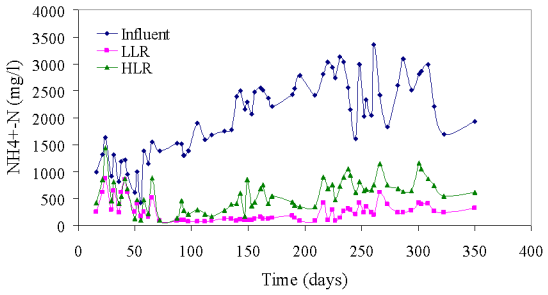

Fig. 4. Unfiltered NH_4^+-N concentrations for SAD influent and effluent at LLR (5 l/m^2/day) and HLR (10 l/m^2/day)

In the first 70 days the influent NH_4^+-N was 1115 ± 330 mg /l. The effluent NH_4^+-N for this period was 427 ± 249 and 553 ± 357 mg/L at LLR and HLR, respectively. The average TON observed in the effluent was 9 mg TON/l and 2 mg TON/l for LLR and HLR, respectively indicating that very little nitrification occurred in the system at the beginning of operation. This was due to the fact that heterotrophic microorganisms were faster growing than autotrophs, which are responsible for nitrification [13]. In this 70-day period an average of 2021 and 1017 mg COD_f/l were removed from the system at LLR and HLR, respectively. This reduction in COD_f and absence of oxidised nitrogen production may indicate that the NH_4^+-N removal in the first 70 days was partly due to its assimilation into cell material in the biofilm [14]. Hanaki et al. [14] indicates that the amount of ammonia that is assimilated into the cells of biomass is equivalent to between 2.5 and 3 % of the COD removal. This would equate an average of 61 mg NH_4^+-N/L and 31 mg NH_4^+-N/L being assimilated into cells during the first 70 days at LLR and HLR, respectively.

The concentration of free ammonia (NH_3) is dependent on the pH and temperature of the wastewater. Free ammonia-nitrogen (NH_3-N) concentrations can be calculated using the following equation [15];

$$[NH_3 - N]_{free} = \frac{[NH_4^+ - N]10^{pH}}{(K_a / K_w) + 10^{pH}}$$

$$\text{where } (K_a / K_w) = \exp[6334/(273+T)]$$

The average NH_3-N concentration in the influent during the study period was 88 mg/L which corresponded to 4.3% of total NH_4^+-N at a temperature of 11°C and a measured pH of 8.34. Nitrogen reduction from wastewater through volatilisation of NH_3 was possible, but it was difficult to quantify in this study.

From around Day 100 significant concentrations of oxidised nitrogen were observed in the biofilter effluent indicating the occurrence of nitrification. This lengthy start-up period was similar to that found by Rodgers et al. [16], who reported the occurrence of nitrification after 120 days in stratified sand filters [16]. Gilbert et al. [17] reported a start-up time of 40 to 50 days before oxidised nitrogen was formed in biofilters with media composed of peat, woodchips, calcite and pozzolana. A long start-up time may be explained by the fact that the nitrifiers have to compete with the heterotrophs for space and dissolved oxygen in the biofilm. Numbers of heterotrophs were found to be four to five times higher than those of nitrifiers due to their higher initial growing rates [18].

The LLR resulted in a higher NO_3^--N production than the HLR ($P<0.05$). The effluent NO_2^--N was not affected by the loading rate ($P>0.05$). A maximum NO_2^--N concentration of 503 mg /l and a maximum NO_3^--N concentration of 1437 mg/l was noted at the LLR (Fig. 5).

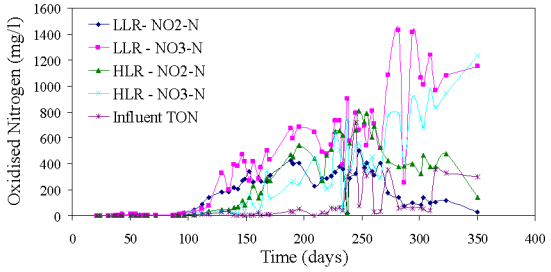

Fig. 5. Oxidised nitrogen concentrations in the SAD influent and effluent at LLR (5 l/m^2/day) and HLR (10 l/m^2/day)

A higher NO_2^--N was noted at HLR, reaching a maximum of 808 mg NO_2^--N/l on Day 248. After this, the production of NO_3^--N increased while NO_2^--N decreased, indicating an increase in complete nitrification. The high values of NO_2^--N observed in the biofilter effluent at both loading rates indicate the occurrence of partial nitrification. According to Anthonisen et al. [15], free ammonia inhibition of *Nitrobacter*, which is responsible for the oxidation of NO_2^- to NO_3^-, occurs at NH_3 concentrations between 0.1 and 1 mg N/l. NH_3 concentrations of 10 – 150 mg N/L can inhibit *Nitrosomonas*, which is ammonium oxidizing bacteria. The level of NH_3 in the woodchip biofilters could have inhibited the nitrite oxidizing bacteria and as a result, the partial nitrification occurred.

Simultaneous nitrification and denitrification was likely to have occurred in the biofilters. Though the biofilters were primarily aerobic, denitrification was likely to have occurred in anoxic zones within biofilms or within the biofilters in areas of high saturation. The increase in effluent pH suggests that denitrification occurred within the biofilters.

Construction and analysis of pilot scale woodchip biofilters is currently ongoing. Following on from the results of this study it is recommended that in practice a 1 m deep aerobic biofilter would result in better treatment. This additional depth may increase nitrification in the system due to the increase in the surface area for nitrifiers to grow. The increase in depth also provides additional carbon from the woodchips, resulting in an increase in heterotrophic bacterial growth and therefore increased nitrogen removal.

As the majority of the nitrogen in the filter effluent was in the form of NO_2^--N and NO_3^--N, it is recommended to pass the effluent through an anoxic zone. The pilot scale study incorporates a saturated layer of woodchips at the base of the filter; supplying an anoxic environment for the conversion of oxidised nitrogen to nitrogen gas. Studies have shown that biofilters are effective in the removal of ammonia gas [19] and hydrogen sulphide [20], so NO and N_2O released during denitrification may be captured in the biofilms in the aerobic woodchip layer thus possibly reducing greenhouse gas emissions.

4 Conclusions

Results from the laboratory-scale study indicate:

(i) 0.6 m deep woodchip biofilters comprising 0.2 m of fine woodchip on top of 0.4 m of coarse woodchip were successful in removing an average of 59 % and 54% SS from separated raw piggery wastewater (SR) and separated piggery anaerobic digestate (SAD), respectively.

(ii) For both piggery wastewaters (SR and SAD) effluent COD_{uf}, COD_f and NH_4^+-N concentrations were dependent on the loading rate, with the LLR (5 $l/m^2/d$) resulting in higher average percentage removals.

(iii) The lower loading rate resulted in higher complete nitrification in biofilters treating both SR and SAD liquids ($P<0.05$).

(iv) Nitrogen removal was observed in the woodchip biofilters. The removal mechanisms may include filtration, simultaneous nitrification and denitrification, ammonium volatilisation, and biomass assimilation.

(v) The majority of NH_4^+-N removed in the system was converted to NO_2^--N and NO_3^--N. The addition of a saturated layer of woodchips under the aerobic biofilters could improve the nitrogen removal performance of the system as this anoxic zone would encourage denitrification, thereby reducing the oxidised nitrogen in the effluent.

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