Available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/watres

Influence of sand layer depth and percolate impounding regime on nitrogen transformation in vertical-flow constructed wetlands treating faecal sludge

Atitaya Panuvatvanich[a,1], *Thammarat Koottatep*[a,1], *Doulaye Kone*[b,*]

[a]*School of Environment, Resources and Development (SERD), Asian Institute of Technology (AIT), P.O. Box 4, Pathumtani 12120, Thailand*
[b]*Eawag: Swiss Federal Institute of Aquatic Science and Technology, Department of Water and Sanitation in Developing Countries (Sandec), Ueberlandstrasse 133, P.O. Box 611, CH-8600 Dübendorf, Switzerland*

ARTICLE INFO

Article history:
Received 22 December 2008
Received in revised form
11 March 2009
Accepted 14 March 2009
Published online 26 March 2009

Keywords:
Denitrification
Faecal sludge treatment
Impounding regime
Nitrification
Vertical-flow constructed wetlands

ABSTRACT

Four laboratory-scale units of vertical-flow constructed wetlands (VFCW) were fed once a week with faecal sludge (FS) at a constant solids loading rate (SLR) of 250 kg TS/(m^2.year) (equivalent to 260–300 g N/(m^2.week)) for a period of 12 weeks to study: i) the nitrification and denitrification potential of the sand layer of VFCWs and ii) the effect of percolate impounding regime (permanent or batch-impounding) on nitrogen transformation. The TN content of raw FS was characterised by 65% org-N, 34% NH_4–N and 1% NO_x–N. After FS application and a six-day impounding period, 8–13% TN were recovered in the percolate exhibiting the following composition: 70–80% NH_4–N, 25–30% org-N and <1% NO_x–N. A large fraction of the influent organic N (55%) was filtered in the bed and 24–29% of initial NH_4–N were lost due to nitrification and volatilisation. In permanent impounding systems, 8–11% TN were recovered in the percolate versus 13% in batch-operated beds. N loss was increased with sand layer depth (20–40 cm) under permanent impounding regimes.

1. Introduction

In urban areas of developing countries, thousands of tons of faecal sludge (FS) collected daily from on-site sanitation installations are disposed of inappropriately. A survey conducted by the Department of Health (DOH, 2008) on faecal sludge management (FSM) practices in Thailand revealed that only 42% of the municipalities are equipped with FS treatment systems. However, existing facilities are operational in only 21.4% of the municipalities. FS treatment plants are still lacking in 58.1% of the municipalities in Thailand. Hence, raw FS is used in agriculture, aquaculture or discharged indiscriminately into lanes, drainage ditches, onto open urban spaces, into inland waters, estuaries or the sea. FS contains high concentrations of pathogens and may also be contaminated with hazardous pollutants (WHO, 2006; Pronk and Koné, 2008). Given its high nutrients concentration (especially N and P), (Koottatep et al., 2005), uncontrolled FS discharge into surface waters severely impacts urban ecosystems and leads to significant oxygen depletion and toxicity in aquatic systems (Yutani and Tanaka, 2007). However, the valuable nutrients contained in FS can be recovered for agricultural use (Hadsoi, 2005).

Vertical-flow constructed wetland (VFCW) is a potential treatment system to recover nutrients and biosolids for reuse

* *Corresponding author*. Tel.: +41 44 823 55 53; fax: +41 44 823 53 99.
E-mail addresses: atitaya.panuvatvanich@ait.ac.th (A. Panuvatvanich), thamarat@ait.ac.th (T. Koottatep), doulaye.kone@eawag.ch (D. Kone).
[1] Tel./fax: +66 2 524 55 95.
0043-1354/$ – see front matter
doi:10.1016/j.watres.2009.03.029

purposes. It has proven robust and presents a viable option with a promising N treatment performance (Koottatep et al., 2005; Lienard et al., 2005; Kengne et al., 2008). However, current design and operational criteria of planted drying beds do not allow to predict N recovery or removal from the system. Since focus is on dewatering, little focus has been placed on its N removal or recovery potential. An improved understanding of nitrogen transformation processes and reuse potential will help design dewatering beds either for N removal or resource and nutrient recovery.

1.1. *Planted drying beds (VFCW) for FS treatment*

For more than 12 years, planted drying beds or VFCWs have been used as sludge treatment option in Denmark (Nielsen, 2003, 2005) and France (Lienard et al., 2005). VFCWs, which are widely spread throughout Europe, Asia, Africa, and the United States (Koottatep et al., 2001, 2005; Nassar et al., 2006; Kengne et al., 2008), are regarded as a promising treatment option, since they are easy to operate (Koottatep et al., 2005; Paing and Voisin, 2005), cost-effective and provide a technically feasible approach for sludge dewatering, stabilisation and humification (Nielsen, 2003, 2005; Koottatep et al., 2005; Kengne et al., 2008).

The solid loading rate (SLR), defined as the total solid weight of FS applied to the system per unit surface area and time, is often used as design criteria for FS treatment. Cooper (2007) suggests SLRs of 30–80 kg TS/(m^2.year) for reed beds treating excess sludge from a wastewater treatment plant in Europe. Nielsen (2003, 2005) suggests an optimal loading rate of 50–60 kg TS/(m^2.year) for reed beds treating activated and digested sludge. However, Koottatep et al. (2005) and Kengne et al. (2008) reveal that in tropical regions, the planted vertical-flow constructed wetlands can be loaded at much higher rates (100–250 kg TS/(m^2.year)). Aside from sludge dewatering, higher nitrification was observed with an NO_3–N percolate concentration amounting to $>$ 100 mg/L (Koottatep et al., 2005) in free drainage beds loaded once a week at 250 kg TS/(m^2.year). However, when operated at a loading frequency of once a week, plant wilting occurred and a limited system performance was observed. Hence, Koottatep et al. (2005) suggested batch-impounding of the percolate for six days to prevent water plant stress and achieve a high denitrification of NO_3–N. Nevertheless, despite high N removal efficiency, large-scale batch-impounding was subject to regular interventions by an operator, especially in areas where remote control is not a feasible option. Hence, further improvements of the operational pattern are necessary as well as investigations of the conditions under which N removal or recovery can be enhanced.

1.2. *Nitrogen transformation in planted dewatering beds*

Nitrogen transformation in constructed wetlands is quite a complex mechanism involving many biological and physico-chemical processes, such as ammonia volatilisation, ammonification, nitrification, denitrification, plant and microbial assimilation, remineralisation, sedimentation of particulate N, and also adsorption onto substrata.

To date, only few publications have discussed N transformation in planted dewatering beds treating FS. As reported (Koottatep et al., 2002), 63% N content of raw FS removed from the VFCW system was distributed as follows: (i) 0.002% accumulated in plant biomass, 55% in the dried sludge on top of the beds and 8% was recovered in the percolate. The remaining (unaccounted) 37% of TN–N input was assumed to be lost in the VFCW as a result of biological uptake by microbial organisms, adsorption of NH_4^+ and NO_3^- onto substrata, ammonia volatilisation, and nitrification/denitrification reactions. Other studies reveal that 57.6% of influent nitrogen was primarily removed by sedimentation and filtration on top of the beds, followed by denitrification (40.9%), direct plant uptake (0.5%) and 1% recovery from the percolate (Hamersley et al., 2004). Based on the literature, about 55–60% are removed by sedimentation and bed filtration, less than 1% by plant uptake and less than 10% end in the percolate. Theoretically, sequential microbial nitrification–denitrification is generally the main reason for ammonia disappearance in these systems (Tanner et al., 2002; Liu et al., 2005; Sun and Austin, 2007), yet, there is still no clear explanation of how configuration of VFCW's ''black box'' can influence N loss or recovery.

The literature (Von Felde and Kunst, 1997) suggests that 95% NH_4^+–N are nitrified at 20 cm soil depth and good oxygen supply. The results correlate very well with those of Guilloteau (1992), who found that nitrification occurs at 30 cm soil depth. Moreover, many researchers (Walker et al., 1973; Whelan and Barrow, 1984; Pell and Nyberg, 1989) report that the nitrification reaction was completed at 5–60 cm infiltration depth in beds fed with wastewater from treatment systems. Investigations of potential nitrification activity in different soil depths confirm that maximum nitrification takes place directly beneath the system's surface. Therefore, various approaches to promote aeration physically have been suggested and recently tested.

Bayley et al. (2003) suggest that higher N removal occurs at over 50 cm reed bed depth. These results correlate well with those of (Von Felde and Kunst, 1997), who suggest that in water-saturated and flooding columns, the denitrification process also takes place at 20 cm soil bed depth. However, Tanner et al. (1999) suggest that under drainage conditions, gaseous oxygen entering the interstitial space in the gravel matrix reduces anoxic conditions. Based on the literature, it can thus be concluded that electrons responsible for denitrification are available in deeper soil areas under saturated condition – a hypothesis yet to be confirmed.

To improve VFCW design for N recovery or removal, this study investigated: i) the relationship between sand layer depth and nitrogen transformation, focusing on nitrification reaction and ii) effect of percolate impounding regime on nitrogen transformation.

2. Materials and methods

The studies were conducted in four lab-scale VFCW units at the Environmental Research Station of the Asian Institute of Technology (AIT), Pathumtani, Thailand (14°04′33.89″N/100°36′18.41″E; 14°05′00.31″N/100°37′00.69″E). Each consisted of a square plastic tank 1.0 m × 1.0 m × 0.6 m in size (width × length × substrata depth) planted with cattail (*Typha angustifolia*) of 10–12 plants/m^2 initial density. The substrata of the VFCW units were composed of large-sized gravel of

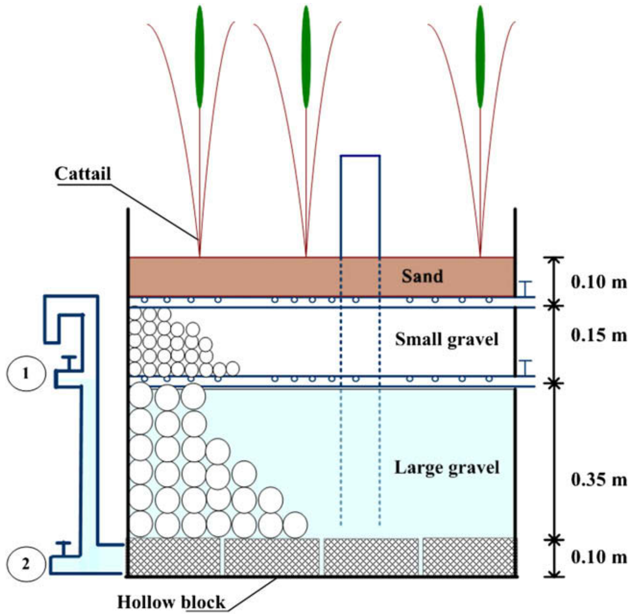

Fig. 1 – Schematic diagram of VFCW units: i) batch-impounding: valve 1 + 2 remained closed for six days after feeding, valve 2 was opened one day for percolate discharge prior to the next application; ii) permanent impounding: valve 2 was maintained closed and valve 1 remained open throughout the experiment.

25–50 mm placed at the bottom of the units, small-sized gravel of 10–25 mm in the middle and fine-sized sand of 0.30–0.75 mm diameter on top of the tanks. The drainage system, consisting of hollow concrete blocks, each 18 cm × 38 cm × 6.5 cm in size (width × length × hollow space), was equipped with a 75 mm diameter PVC ventilation pipes. The influent was evenly distributed on top of the constructed wetland and drained at the bottom by gravity. Two drainage patterns were investigated: i) batch-impounding and ii) permanent impounding. In batch-impounding (Fig. 1), the percolate was retained for six days at the bottom of the unit (gravel layer) by closing valves 1 and 2. The retained percolate was drained out for one day prior to applying the next FS load. In the permanent impounding system (Fig. 1), the outlet was raised with a PVC pipe to maintain the percolate at a constant level of 40–50 cm in the gravel layer. Valve 1 was opened and valve 2 maintained closed during the whole experiment. Table 1 provides details of media type, media depth and impounding regime.

Experimental runs 1 and 2 were designed to assess the effect of percolate impounding regimes (batch or permanent). Experimental runs 2–4 aimed at investigating the influence of sand layer depth on N transformation. The sand layer in these experiments varied from 10 to 40 cm.

Raw FS was applied once a week to all experimental units at 250 kg TS/(m^2.year) SLR for a period of 12 weeks, and samples were taken daily at the outlet. FS and percolate from sampling points were analysed for TN–N, NH_4^+–N, NO_2–N, and NO_3–N according to the ‘‘Standard Methods for the Examination of Water and Wastewater’’ (APHA, 1998).

2.1. Nitrogen mass balance

N mass balance was measured in the raw FS, percolate, plant biomass, accumulated sludge, and NH_3 volatilisation from the VFCW unit. N mass balance was calculated according to the following equation:

$$\mathrm{INF} = \mathrm{EFF} + \mathrm{PB} + \mathrm{DS} + \mathrm{NL} + \mathrm{UNA}$$

Where INF = N content in influent FS, g N/(m^2.week); EFF = N content in percolate, g N/(m^2.week); PB = N content in plant biomass, g N/(m^2.week); DS = N content in accumulated sludge, g N/(m^2.week); NL = NH_4–N loss in VFCW system, g N/(m^2.week); UNA = Unaccounted for balance, g N/(m^2.week)

2.2. Determination of ammonia volatilisation

Ammonia volatilisation (NH_3 gas) from accumulated sludge was measured and analysed in all lab-scale units throughout the experiment. Ammonia volatilisation measurements were conducted in an acrylic box of 20 cm × 20 cm × 20 cm (width × length × height) with four sides and top closed. One side of the box was opened and placed on the accumulated sludge surface. The top was connected to a pump driving air into the box at the same wind velocity as environmental conditions. The NH_3 gas was trapped in two flasks laid out in series, each flask contained 100 mL of 2% boric acid solution. The tubes in the trap flask were placed at about 1 cm below the surface of the boric acid solution. The ammonium concentration was analysed sequentially, and the volatilisation rates of the total accumulated sludge surface area were extrapolated from the measured values.

2.3. Determination of nitrogen plant uptake

At the beginning and end of each experimental run, aboveground and belowground the biomass of all units was sampled for plant analysis. In each unit, a 0.25 m^2 core of plant biomass was harvested. The N content of aboveground and belowground biomass was analysed by digesting dried

Table 1 – Media size and depth used in the VFCW units as well as percolate impounding regime.

Experimental runs	Impounding types	Media depth (cm)		
		Sand (0.30–0.75 mm)	Small gravel (10–25 mm)	Large gravel (25–50 mm)
1	Batch-impounding	10	15	35
2	Permanent impounding	10	15	35
3	Permanent impounding	20	20	20
4	Permanent impounding	40	–	20

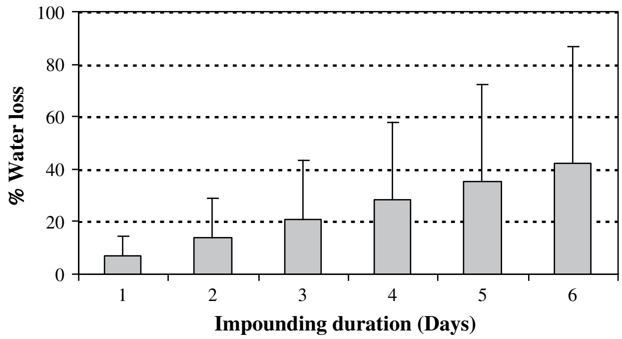

Fig. 2 – Mean percentage of water loss from a VFCW system during a six-day percolate impounding period.

samples of plant material and measuring nitrogen in the remaining solution (Lisamarie and Joan, 2003).

2.4. Water loss

A separate set of six experimental VFCW units, similarly configured as experimental run 1, were used to investigate the daily water loss over a six-day percolate impounding period. Each unit was loaded simultaneously with raw FS at 250 kg TS/(m^2.year) SLR. From day one to day six, the percolate was drained out every day from only one unit to determine the accumulated water volume loss over time.

3. Results and discussion

3.1. Water loss from VFCW systems

An average FS load of 300 L/week (HLR 30 cm/week) was fed to each experimental bed. Fig. 2 illustrates in % the average water loss from all experimental runs during the impounding period. After six days of impounding, about 173 L (58%) of percolate were recovered. Some 127 L (42%) of the inflow were lost during the six-day impounding period, thus corresponding to a daily water loss of 21.16 mm/(m^2.day).

3.2. Nitrogen mass balance

Total nitrogen (TN) loading of FS applied to the experimental beds ranged from 260 to 300 g N/(m^2.week). As shown in Table 2, the influent contained about 65% org-N, 34% NH_4–N and 1% NO_x–N. After a six-day impounding period, the NH_4–N predominating in the effluent (70–80%) contained 25–30% org-N. NO_x–N concentration mass remained negligible also in the influent (1%). Average TN mass content in experimental runs 1, 2, 3, and 4 after a six-day impounding period amounted to about 37, 32, 25, and 22 g N/week, respectively (Table 3).

N mass balance revealed that, on average, 55% of inlet TN were recovered in the sludge retained on top of the beds. N loss due to plant uptake and volatilisation was negligible, i.e. 0.2% and 0.01%, respectively. Percolate TN recovery varied between 8 and 13%, while 4.8–12.8% in the initial TN load were unaccounted for and probably attributed to denitrification.

TN recovery in the percolate was higher in batch-operated systems at a similar sand layer depth. In permanent impounding systems, less recovery was achieved, the greater the sand layer depth; 11%, 8% and 8% for sand layer depth of 10, 20 and 40 cm, respectively. No significant differences were observed in the percolate collected from beds with 20 and 40 cm sand layer depth.

3.3. TN removal efficiency

As show in Fig. 3, 90% of influent org-N mass was removed within the first day, and an additional 5% removal was

Table 2 – Influent and effluent characteristics of experimental VFCWs units in (mg/L) (for all parameters) and in (g) and percentage (for all nitrogen species).

Parameter	Influent[a]	Effluent[b]			
		Experimental run 1	Experimental run 2	Experimental run 3	Experimental run 4
Temperature (°C)	28.67 ± 1.5	32.50 ± 1.80	33.10 ± 1.30	32.40 ± 1.80	31.30 ± 1.60
ORP (mV)	−291 ± 30	−229 ± 38	−290 ± 24	−264 ± 35	−249 ± 33
pH	7.48 ± 0.5	7.21 ± 0.2	7.28 ± 0.3	7.25 ± 0.2	7.24 ± 0.2
TS (mg/L)	22,420 ± 7702.6	2245 ± 350	1929 ± 310	1745 ± 210	1665 ± 190
TSS (mg/L)	19,500 ± 7,250	1100 ± 350	995 ± 320	945 ± 290	915 ± 310
BOD_5 (mg/L)	2225 ± 395	298 ± 95	283 ± 85	278 ± 85	264 ± 75
TN (mg/L)	950 ± 99.18 (285, 100%)	215 ± 40 (37, 13%)	185 ± 30 (32, 11%)	145 ± 30 (25, 9%)	130 ± 25 (22, 7.8%)
NH_4–N (mg/L)	320 ± 70.89 (95, 34%)	170 ± 30 (30, 10%)	120 ± 20 (21, 7.4%)	110 ± 20 (19, 6.5%)	85 ± 20 (15, 5.3%)
Org-N (mg/L)	625 ± 148.66 (185, 65%)	32 ± 25 (5.7, 2%)	58 ± 25 (9.3, 3.2%)	30 ± 20 (5.5, 1.9%)	40 ± 25 (6.2, 2.2%)
NO_3–N (mg/L)	4.81 ± 1.65 (1.5, 0.5%)	10.70 ± 4.88 (1.85, 0.65%)	7.32 ± 1.58 (1.30, 0.44%)	6.52 ± 1.02 (1.13, 0.40%)	5.67 ± 2.41 (0.98, 0.34%)
NO_2–N (mg/L)	0.82 ± 0.46 (0.25, 0.08%)	0.10 ± 0.03 (0.02, 0.01%)	0.08 ± 0.03 (0.01, 0.004%)	0.10 ± 0.04 (0.02, 0.007%)	0.40 ± 0.30 (0.07, 0.02%)

(a, b%) mean (g N/m^2.week, % of TN).
a Data based on 30 samples.
b Data based on 10 samples.

Table 3 – TN mass balance as a function of sand layer depth in VFCWs treating raw faecal sludge.

Experimental runs	N mass (g N/(m^2.week))						
	Raw FS TN	N–NH_4 loss in VFCW	N accumulated sludge	N Plant uptake	N–NH_3 volatilised	N Percolated	N Unaccounted
1	280 ± 23 [a]	75.71 ± 17.96 [a] (27%)	154.00 ± 12.64 [a] (55%)	0.56 ± 0.05 [a] (0.2%)	0.035 ± 0.002 [a] (0.01%)	37.10 ± 7.07 [a] (13%)	12.63 (4.8%)
2	285 ± 25 [a]	73.92 ± 15.51 [a] (26%)	156.75 ± 16.50 [a] (55%)	0.57 ± 0.05 [a] (0.2%)	0.036 ± 0.002 [a] (0.01%)	31.71 ± 5.53 [b] (11%)	22.05 (7.8%)
3	305 ± 33 [a]	73.22 ± 15.16 [a] (24%)	167.75 ± 18.12 [a] (55%)	0.61 ± 0.07 [a] (0.2%)	0.035 ± 0.002 [a] (0.01%)	25.38 ± 4.22 [b] (8%)	38.04 (12.8%)
4	290 ± 19 [a]	83.40 ± 20.13 [b] (29%)	159.50 ± 10.28 [a] (55%)	0.58 ± 0.04 [a] (0.2%)	0.037 ± 0.002 [a] (0.01%)	22.42 ± 3.75 [c] (8%)	24.10 (7.8%)

Data based on 10 samples.
Data with different letters within the same row indicate a significant difference at $p < 0.05$.

achieved over the remaining five days of percolate impounding. NH_4–N removal increased slightly with sand layer depth in the VFCW and yielded 45%, 50%, 55%, and 56% after one day percolate impounding for runs 1–4, respectively. Between days 2–6, additional removal of 30%, 28%, 23%, and 28% was achieved in runs 1, 2, 3, and 4, respectively.

After the first impounding day, a total of 72% and 77–79% TN were removed in run 1 and runs 2–4, respectively. Higher removal efficiencies (87%, 89%, 92%, and 92%) were achieved after six days of percolate impounding in all units.

3.4. *Effect of sand layer depth and percolate impounding regime on nitrification*

NO_x–N mass increased about 80–85% to reach a maximum load on day 1 in runs 1, 3 and 4, and after day 2 in run 2. The highest NO_x–N load in the percolate (5.28 and 5.12 g N/(m^2.day)) was achieved on day 1 of percolate impounding in beds containing 20 and 40 cm of sand and representing 5.8% and 5.2% of the initial NH_4–N load. In beds containing a 10 cm sand layer (runs 1 and 2), the NO_x–N load in percolate was higher in batch-operated beds, i.e. 7.8% and 2.6% of the initial NH_4–N load. NO_x–N loads decreased between day 2 and day 6 of percolate impounding. In all experiments, the NO_x–N load remained higher in batch-operated beds.

3.5. *Effect of sand layer depth and percolate impounding regime on denitrification*

In this experiment, the unaccounted for N was assumed to be lost by denitrification, a key factor controlling N disappearance in sand filtration systems (Koottatep et al., 2002; Hamersley et al., 2004). Fig. 3 reveals that after six days of impounding, all the NO_x–N loads equal those in the raw sludge effluent. These results suggest that the total amount of NH_4–N nitrified in the first days of percolation was denitrified after six days of impounding, irrespective of impounding regime and bed configuration. They also reveal that nitrification–denitrification was the main factor controlling N transformation in VFCWs. Table 3 reveals that N loss by denitrification was lower in batch-operated systems (4.8%) than in permanent impounding beds (7.8–12.8%). Consequently, ‘‘batch impounding systems’’ are efficient in N recovery as their percolate releases more NO_x–N. In contrast, ‘‘permanent impounding systems’’ create favourable conditions for denitrification, which is enhanced when sand layer depth is increased.

4. Discussions

4.1. *Water loss*

According to this study, average water loss from VFCW systems corresponds to 42% (127 L) during six days of percolate impounding. This is equivalent to a daily loss of 21.16 mm/(m^2.day). Water loss can be attributed to plant evapotranspiration or storage in the top sludge layer, adsorption to the filter media surface or to interstitial water. Similar results are reported from other studies, where water loss from evapotranspiration in wetlands treating wastewater

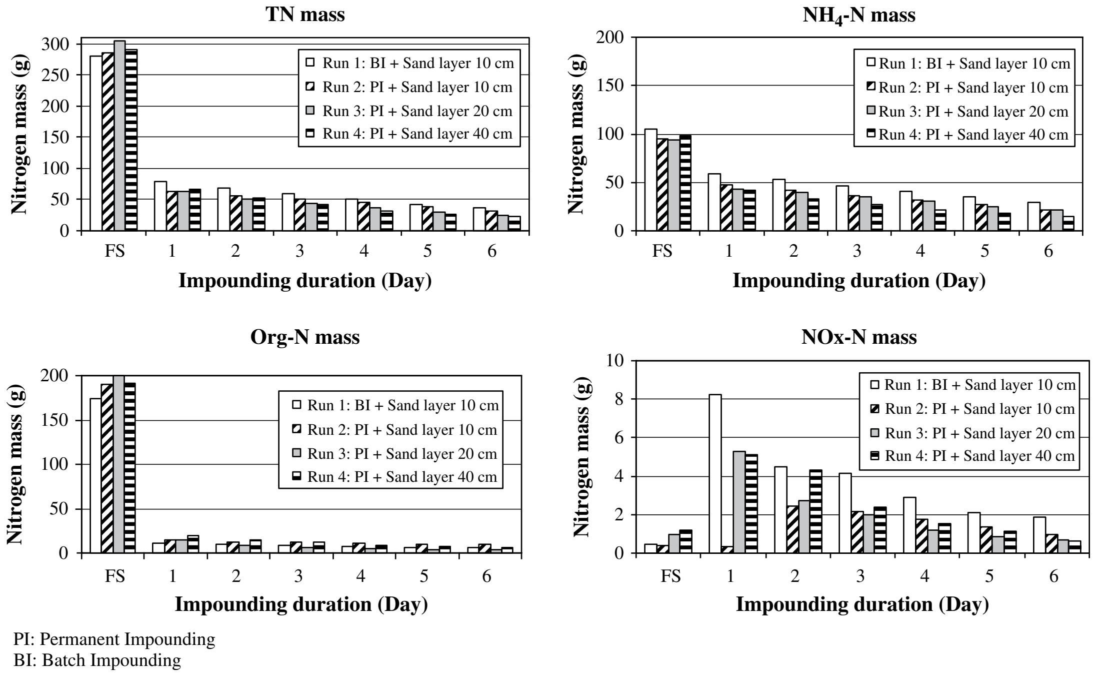

Fig. 3 – Nitrogen mass flow as a function of sand layer depth in VFCW systems during a six-day percolate impounding period.

in Tanzania (temperature 23–34 °C) and Italy varied between 21 and 32 mm/(m^2 d) (Ranieri, 2003; Mbuligwe, 2005). Based on this study, a six-day impounding period requires a minimum hydraulic load of 12.7 ± 4.48 cm/(m^2.week) in the VFCW to prevent plant wilting.

4.2. *Nitrification*

VFCW systems of greater sand layer depth achieve faster and higher nitrification for the first 1–2 days before reaching a steady decrease from day 3 to day 6. As suggested by other authors (McNevin et al., 1999; Lienard et al., 2005), NH_4–N is adsorbed onto bed media and organic matter during loading and nitrified mainly between feeding periods. According to Tanner et al. (1999), nitrification in the filter material is maintained by the oxygen stored in the biofilms surrounding sand surfaces and also in the interstitial spaces of the sand matrix, thus resulting in rapid consumption of available N substrates by aerobes. It can thus be concluded that an increase in sand layer depth of VFCWs could improve nitrification reaction in VFCW systems. However, the nitrate load released in the percolate of this study does not allow assessment of nitrification rates. Further investigations of filter material NH_4–N adsorption (McNevin et al., 1999; Lienard et al., 2005) and nitrification capacity are required to refine bed configuration and operational conditions for enhanced nitrogen removal or recovery. Nevertheless, our results suggest that frequent feeding of three to four days can enhance nitrification and denitrification in VFCW treating faecal sludge.

4.3. *Denitrification*

This study reveals that within 1–3 days, denitrification rates are higher in "permanent impounding systems" than in "batch impounding systems". However, no significant differences at $p < 0.05$ were observed on days 3–6 of percolate impounding. Indeed, the "batch impounding systems" may allow air to be sucked into the systems during drainage (Lienard et al., 2005) Hence, at the start of each new batch feeding operation, denitrification rates reached minimum values. An increase in denitrification was observed only after a prolonged impounding period, as impounding leads to increased anaerobic conditions (Koottatep et al., 2005; Lienard et al., 2005). This confirms the assumption that "permanent impounding systems" could provide more appropriate conditions for denitrifying reactions than "batch impounding systems"; a view also held by Von Felde and Kunst (1997) and Molle et al. (2008).

4.4. *Effects of percolate impounding period*

The results of this study reveal that a prolonged impounding period of six days does not significantly affect the overall performance of N removal. Indeed, after a period of six days, the nitrate load in the percolate ranged from 0.6 to 1.8 g, equivalent to 0.6–1.7% in inlet NH_4–N load. Koottatep et al. (2005) suggest an optimal six-day impounding period to prevent plant witling under tropical conditions. However, the VFCW configuration proposed in this study (Fig. 1) indicates that by maintaining a permanent water level in the drainage layer (percolate impounding), water is supplied through capillary exchange to meet plant requirements. The permanent impounding systems provide flexible operating conditions, as they do not require operator interventions to open and close a valve before and after each feeding. Moreover, bed configuration can be adapted to meet N removal or recovery by modifying the sand layer depth or loading frequency as suggested in the previous paragraph. Such a system will best fit secondary FS treatment, following the solids–liquid separation phase.

5. Conclusions

This work aimed at studying the influence of sand layer depth (10, 20 and 40 cm) and percolate impounding regime (batch and permanent impounding) on nitrogen transformation in VFCW treating FS. Based on the results obtained, the following conclusions can be formulated:

- Overall TN mass removal increases with sand layer depth in faecal sludge-fed VFCW. However, TN removal efficiency varies from 87 to 92% and is not significantly influenced by bed configuration nor impounding regime.
- VFCWs, fed once a week with faecal sludge at SLR 250 kgTS/(m^2.year), reach higher nitrate loads in the percolate, i.e. 3–5 g N/(m^2.day) within the first 1–2 days of impounding, irrespective of bed configuration or impounding regime. After three days of percolate impounding, the nitrate load in the percolate decreases steadily to reach its initial value. Except for batch-operated beds, all the NO_x–N produced in the VFCW beds was denitrified.
- On average, 4.8% of initial TN was lost in batch-impounding beds, and 7.8–12.8% in permanent impounding systems. An increase in sand layer depth in VFCW enhanced N nitrification and denitrification.
- Impounding periods of six days did not lead to an improved N removal or recovery. Nitrification appears to be the governing factor of N loss. However, nitrification rates in filter media remain unknown.
- Further investigations on NH_4 adsorption and nitrification capacity of filter material are necessary to improve bed configuration and operating conditions for enhanced nitrogen recovery or removal.

Acknowledgements

This research, conducted by the Department of Water and Sanitation in Developing Countries (Sandec) of the Swiss Federal Institute of Aquatic Science and Technology (Eawag), Switzerland and the Asian Institute of Technology (AIT), was sponsored by grants from the Velux Foundation and from the Swiss Development Cooperation.

REFERENCES

APHA, 1998. Standard methods for the examination of water and waste water, Washington DC.

Bayley, M.L., Davison, L., Headley, T.R., 2003. Nitrogen removal from domestic effluent using subsurface flow constructed wetlands: influence of depth, hydraulic residence time and pre-nitrification, pp. 175–182.

Cooper, P., 2007. In: Novais, J.M. (Ed.), The constructed wetland association UK database of constructed wetland systems, pp. 1–6.

DOH, 2008. Faecal Sludge Management in Thailand. Bureau of Environmental Health, Department of Health (DOH).

Guilloteau, J.A., 1992. Traitement des eaux rećsiduaires par infiltration percolation. Performances biomasse et renouvellement des gaz. Universiteć' Louis Pasteur, Strasbourg, France.

Hadsoi, S., 2005. Reuse and Recycle of Bio-residue (Percolate) from Constructed Wetland Treating Septage. Asian Institute of Technology, Thailand.

Hamersley, M.R., Howes, B.L., White, D.S., 2004. Denitrification and carbon availability in a septage-treating ecologically engineered wetland. Journal of Environmental Quality.

Kengne, I.M., Akoa, A., Soh, E.K., Tsama, V., Ngoutane, M.M., Dodane, P.H., Kone, D., 2008. Effects of faecal sludge application on growth characteristics and chemical composition of echinochloa pyramidalis (lam.) hitch. and chase and cyperus papyrus l. Ecological Engineering 34 (3), 233–242.

Koottatep, T., Polprasert, C., Oanh, N.T.K., Surinkul, N., Montangero, A., Strauss, M., 2001. Septage dewatering in vertical-flow constructed wetlands located in the tropics. Water Science and Technology 44 (2–3), 181–188.

Koottatep, T., Polprasert, C., Oanh, N.T.K., Surinkul, N., Montangero, A., Strauss, M., 2002. Constructed wetlands for septage treatment – towards effective faecal sludge management, Avignon, France, pp. 719–735.

Koottatep, T., Surinkul, N., Polprasert, C., Kamal, A.S.M., Kone, D., Montangero, A., Heinss, U., Strauss, M., 2005. Treatment of septage in constructed wetlands in tropical climate: lessons learnt from seven years of operation. Water Science and Technology 51, 119–126.

Lienard, A., Molle, P., Boutin, C., Dodane, P.H., 2005. Treatment of water using artificial wetlands: action of plants and development of the technique in France [Traitement des eaux uses par marais artificiels: Action des plantes et developpement de la technique en France]. (11), 45–55.

Lisamarie, W., Joan, G.E., 2003. Net impact of a plant invasion on nitrogen-cycling processes within a brackish tidal marsh. Ecological Applications 13 (4), 883–896.

Liu, W., Dahab, M.F., Surampalli, R.Y., 2005. Nitrogen transformations modeling in subsurface-flow constructed wetlands. Water Environment Research 77 (3), 246–258.

Mbuligwe, S.E., 2005. Comparative treatment of dye-rich wastewater in engineered wetland systems (ewss) vegetated with different plants. Water Research 39 (2–3), 271–280.

McNevin, D., Barford, J., Hage, J., 1999. Adsorption and biological degradation of ammonium and sulfide on peat. Water Research 33 (6), 1449–1459.

Molle, P., Prost-Boucle, S., Lienard, A., 2008. Potential for total nitrogen removal by combining vertical flow and horizontal flow constructed wetlands: a full-scale experiment study. Ecological Engineering 34 (1), 23–29.

Nassar, A.M., Smith, M., Afifi, S., 2006. Sludge dewatering using the reed bed system in the Gaza strip, Palestine. Water and Environment Journal 20 (1), 27–34.

Nielsen, S., 2003. Sludge drying reed beds. Water Science and Technology 48 (5), 101–109.

Nielsen, S., 2005. Sludge reed bed facilities: operation and problems. Water Science and Technology 51 (9), 99–107.

Paing, J., Voisin, J., 2005. Vertical flow constructed wetlands for municipal wastewater and septage treatment in French rural area. Water Science and Technology 51 (9), 145–155.

Pell, M., Nyberg, F., 1989. Infiltration of wastewater in a newly started pilot sand-filter system: III. Transformation of nitrogen. Journal of Environmental Quality 18 (4), 463–467.

Pronk, W., Koné, D., 2008. Option for urine treatment in developing countries. In: Richards, B.S., Schafer, A.l. (Eds.). School of Engineering & Electronics, University of Edinburgh, United Kingdom, Scotland, UK, pp. 421–430.
Ranieri, E., 2003. Hydraulics of sub-superficial flow constructed wetlands in semi arid climate conditions. Water Science and Technology 47, 49–55.
Sun, G., Austin, D., 2007. In: Novais, J.M. (Ed.), A mass balance study on nitrification and deammonification in vertical flow constructed wetlands treating landfill leachate, pp. 117–123.
Tanner, C.C., D'Eugenio, J., McBride, G.B., Sukias, J.P.S., Thompson, K., 1999. Effect of water level fluctuation on nitrogen removal from constructed wetland mesocosms. Ecological Engineering 12 (1–2), 67–92.
Tanner, C.C., Kadlec, R.H., Gibbs, M.M., Sukias, J.P.S., Long Nguyen, M., 2002. Nitrogen processing gradients in subsurface-flow treatment wetlands – influence of wastewater characteristics. Ecological Engineering 18 (4), 499–520.
Von Felde, K., Kunst, S., 1997. N- and cod-removal in vertical-flow systems. Water Science and Technology 35 (5), 79–85.
Walker, W.G., Bouma, J., Keeney, D.R., Magdoff, F.R., 1973. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: I. Soil transformations. Journal of Environmental Quality 2 (4), 475–480.
Whelan, B.R., Barrow, N.J., 1984. The movement of septic tank effluent through sandy soils near Perth. I. Movement of nitrogen. Australian Journal of Soil Research 22 (3), 283–292.
WHO, 2006. Guidelines for the Safe Use of Wastewater, Excreta and Greywater. World Health Organization, Geneva.
Yutani, K., Tanaka, N., 2007. In: Kim, I.S., Kim, S., Cho, J. (Eds.), Calculating nitrogen-removal efficiency with a one-dimensional nitrogen budget model of a reed-wetland soil system, pp. 129–138.