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Performance of vertical flow constructed wetlands for faecal sludge drying bed leachate: Effect of hydraulic loading

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ABSTRACT

The discharge of raw faecal sludge directly into the environment is a common practice that threatens environmental and public health in low-income countries. Planted drying beds are a promising and low-cost option for treatment of faecal sludge and the production of fodder plants, but current research shows the leachate quality does not meet guidelines for discharge. This paper investigates the use of Vertical Flow Constructed Wetlands (VFCWs) planted with *Echinochloa pyramidalis* for polishing of leachate from faecal sludge drying beds. At a pilot-scale, three hydraulic loads (50, 100 and 150 mm/d) were applied with single loadings once a week on VFCWs (corresponding to 31 ± 9, 63 ± 19, 94 ± 28 g/m^2/month of BOD_5; 21 ± 6, 42 ± 12, 63 ± 19 TKN; and 2 ± 1; 4 ± 4; 6 ± 6 PO_4–P, respectively, for hydraulic loading rates of 50, 100 and 150 mm/d). Infiltration flow rate, plant growth, rhizospheric bacteria, and leachate characteristics were monitored. VFCWs were effective in reducing on average more than 80% of the pollutants monitored (COD, BOD_5, NH_4–N, TKN, PO_4–P, and faecal bioindicators), which met all National Cameroon and WHO guidelines for safe reuse in agriculture, except for total nitrogen and faecal indicators. Results confirmed a correlation between plant density and rhizospheric bacteria growth with increasing hydraulic load. These are important results, demonstrating that VFCWs can operate efficiently at multiple hydraulic loadings, and are hence adaptable to different sized treatment schemes. It also illustrates that if plant production for fodder is a goal, increased loading rates are preferable as they achieve overall treatment goals and result in greater plant production.

Keywords:
Developing countries
Faecal sludge
Loading rates
Resource recovery
VFCW
Leachate

1. Introduction

Faecal sludge (FS) is produced from onsite sanitation technologies (e.g. pit latrines, septic tanks), and is the result of the collection, storage or treatment of combinations of excreta and blackwater, with or without greywater from these technologies (Montangero and Strauss, 2002; Tilley et al., 2008). Adequate treatment of FS prior to end use or discharge to the environment are imperative to provide human and environmental health protection (Feachem et al., 1983; Koné and Strauss, 2004; Mara and Cairncross, 1989). There have been many attempts at implementing conventional wastewater treatment technologies (e.g. rotating biological contactors, activated sludge), but they have not been well adapted to the African context for a number of reasons, among which the high cost of installation, the availability of a reliable energy supply, and local skills and human resources are prominent. Hence, more "natural" or passive systems also termed low-cost technologies (Strauss et al., 1997) such as planted drying beds for sludge and wastewater (i.e. vertical flow constructed wetlands (VFCWs)) provide a promising alternative. Their use for treatment of wastewaters, including municipal, surface, storm, industrial, and agricultural has been well established (Cofie et al., 2006; Cooper, 2005; Kadlec and Wallace, 2009; Liénard et al., 2004; Stefanakis and Tsihrintzis, 2012; Vymazal, 2007). Preliminary research (Kengne et al., 2011; Stefanakis and

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Tsihrintzis, 2011; Koottatep et al., 2004) has demonstrated they also perform well at the pilot-scale for the treatment of FS in the tropics, which is much more concentrated than conventional wastewater (e.g. 400–2000 mg/L NH_4–N and 10,000–30,000 mg/L COD (NWSC, 2008)). Gagnon et al. (2012) also observed enhanced water quality for the effluent of a planted sludge drying bed for fish farm sludge loaded at a rate of 30 kg TSS m^2/yr by using specific plant species and by keeping the system partly saturated. A pilot-scale implementation of planted sludge drying beds with *Echinochloa pyramidalis* and *Cyperus papyrus* as emergent plants for the treatment of faecal sludge in Cameroon (sub-Saharan Africa) was able to remove ≥30% dry matter content of the faecal sludge. Resulting leachate has a NH_4–N, TKN and COD content ranging from 80–900, 20–800, 100–6300 mg/L, respectively (Kengne et al., 2011). This leachate still needs further treatment for nutrient and/or pathogen removal before it can be discharged to the environment, or for safe reclamation in agriculture. Therefore, the treatment of this sludge drying bed leachate by VFCWs can be a novel approach for high sludge loading rate (200 kg TSS m^2/yr).

The treatment effectiveness and plant growth with VFCWs depends on the hydraulic loading rate. European recommendations for hydraulic loadings of VFCWs when treating wastewater are 50 to 60 mm/d (Brix and Arias, 2005). In warm tropical regions, the hydraulic loading rates are generally higher (Dan et al., 2011; Kivaisi, 2001). Plant nitrogen and phosphorus uptake can increase with increasing hydraulic load (e.g. 18–68 mm/d), but decrease if loading rates are too high (e.g. 135 mm/d) (Tripathi et al., 1991). This correlation between hydraulic loading and VFCW efficiency has been confirmed by other authors (Cooper, 2005; Liénard et al., 2004; Molle et al., 2006). However, Maina et al. (2011) did not observe significant differences in nitrogen uptake based on effluent concentrations over hydraulic loadings of 14–174 mm/d. Brix and Arias (2005) recommend loading frequency of 8–12 times per day at nominal loading rate of 50–60 mm/d. However, at high hydraulic load, less loading times and long resting periods allow renewal of oxygen, as they lead to a better efficiency of filter beds and long term functioning (Molle et al., 2006). Therefore, for the same hydraulic load, the choice between numerous small volumes of batches or less batches of greater volume is determinant in operating VFCWs.

The organic and nutrient concentrations of wastewater is also important to consider in the design of VFCWs (Kadlec and Wallace, 2009). Also important in operating treatment wetlands is the choice of macrophytes to use in VFCWs as shown by Brisson and Chazarenc (2009). In addition to its effective treatment potential, employing VFCWs planted with *E. pyramidalis* has a very good potential for plant production and harvesting, with an aerial production of 1662 g DW/m^2 in unsaturated conditions and 4207 g DW/m^2 in flooded conditions (both at salinity of 2 dS/cm) (Ngoutane et al., 2011). The biomass production represent a great economic potential, as this plant is commonly used as fodder. Based on a survey of fodder demand in Cameroon and the biomass production at the pilot-scale, projections at full-scale for a city of 100,000 inhabitants with 11,000 m^2 in treatment area, *E. pyramidalis* would generate an income of 12,375–39,600 USD in the dry season, and 24,750–59,400 USD in the rainy season to offset treatment costs (Ngoutane et al., 2012).

This paper investigates the possibility of employing VFCWs with *E. pyramidalis* for "polishing" or further treating the leachate from planted sludge drying beds prior to discharge. There is a lack of data on factors affecting the performance and design of VFCWs in subtropical regions. Based on this, hydraulic loadings were evaluated as a key factor in treatment performance.

2. Methods

2.1. Study site

This study was conducted with pilot-scale VFCWs at the University of Yaoundé I, in Cameroon. The field site is located 760 m above sea level at 3°45N and 11°32E, has a typical equatorial Guinean climate characterized by two rainy seasons (September to mid-November and mid-March to June), and two dry seasons (mid-November to mid-March and July to August). The annual average rainfall is 1600 mm, and the average daily temperature is 22° to 35 °C.

2.2. Experimental setup

As shown in Fig. 1, the pilot-scale setup consisted of 1 × 1 × 1 m (length, width, height) plastic skid containers (metal-caged, liquid-storage, pallet-sized containers), each representing 1 m^2 of planted sludge drying bed or VFCWs. There were six parallel flows, each consisting of two containers in series. All the containers were filled with three layers of filter media, increasing in grain size from top to bottom. The layers consisted of 15 cm of 0.3–2 mm diameter sand, 20 cm of 5–15 mm diameter semi-coarse gravel and 30 cm of 15–25 mm diameter coarse gravel. The uniformity coefficient from the sand, semi-coarse gravel and coarse gravel layers were 3.55, 1.67 and 1.37. Leachate produced from the dewatering of faecal sludge from Yaounde in planted drying beds was collected in barrels, and then applied to VFCWs and the effluent was monitored. All the beds were planted with *E. pyramidalis*. Robust cuttings were used to initially establish 12 plants per unit. To establish and acclimatize the plants, each bed was amended once with dried faecal sludge, and then irrigated with raw wastewater obtained from a student dormitory over a period of 2 months (September to November 2010).

2.3. Batch loading

The study was conducted during November 2010–March 2011. Leachate from the dewatering of raw faecal sludge applied once a week at 200 kg TSS/m^2/yr on sludge planted drying beds was collected in barrels. The faecal sludge was a mixture from public toilets and septic tanks, collected and transported with mechanical (vacuum) emptying trucks. This leachate was thoroughly mixed and applied on VFCWs beds once a week at 08:00 in a single batch. Three hydraulic loading rates (HLR) were tested: 50, 100 and 150 mm/d, with each loading rate applied in duplicate. The resting period after each batch loading was one week. Splash plates were used for dispersion of leachate at the surface and to protect the top sand layer. The loading was done according to the unsaturated downflow mode of loading as described by Crites et al. (2006).

2.4. Sampling

Effluent of the VFCWs was collected in polyethylene containers located at the drainage pipe. During the operation, leachate from planted sludge drying beds and effluents from VFCWs were analysed over 10 sampling events. The leachate sampling was done the day before application to VFCWs, and the effluent the same day of application. Two different sampling regimes were carried out for effluents. The first was at 18:00 the same day of application (same time and date for each system). This time was selected, as at least 99% of the volume of applied leachate that will theoretically pass through the beds should have drained through the beds at this time. The second, done only during the first six sampling events and without replication, was performed at 5, 15, 30, 45 and 60 min

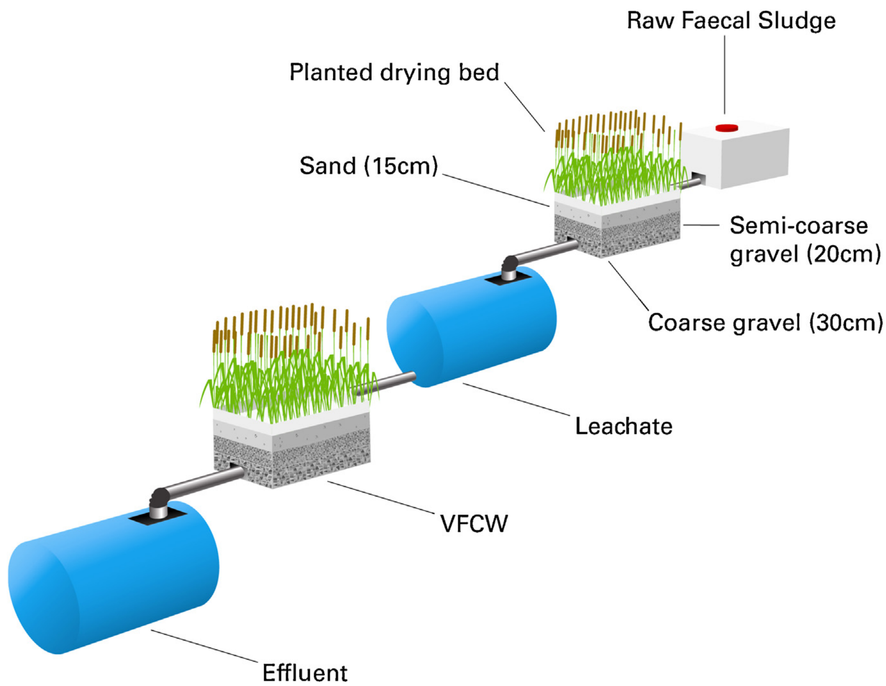

Fig. 1. Schematic of the pilot-scale faecal sludge treatment plant facility at the University of Yaoundé I, including how terminology is defined in this paper.

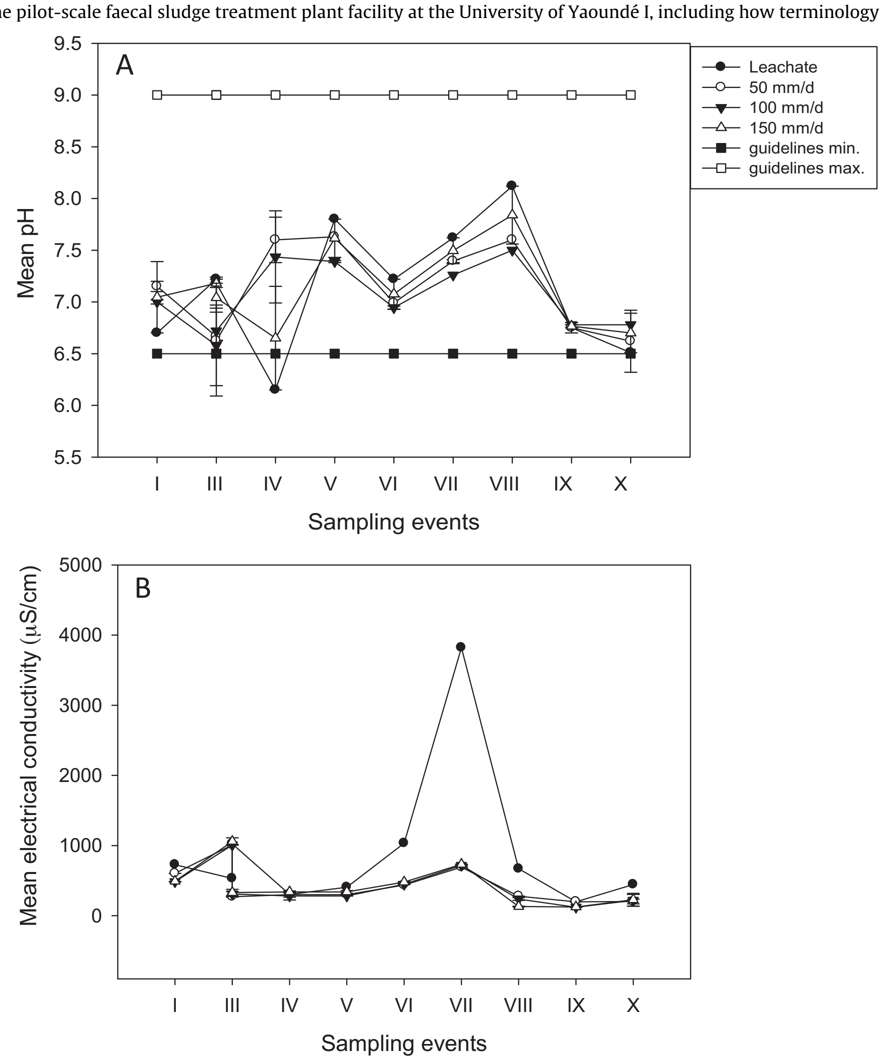

Fig. 2. Physical characteristics of applied leachate and effluent quality for all 10 sampling events for each of the three applied hydraulic loads. (A) pH; (B) electrical conductivity.

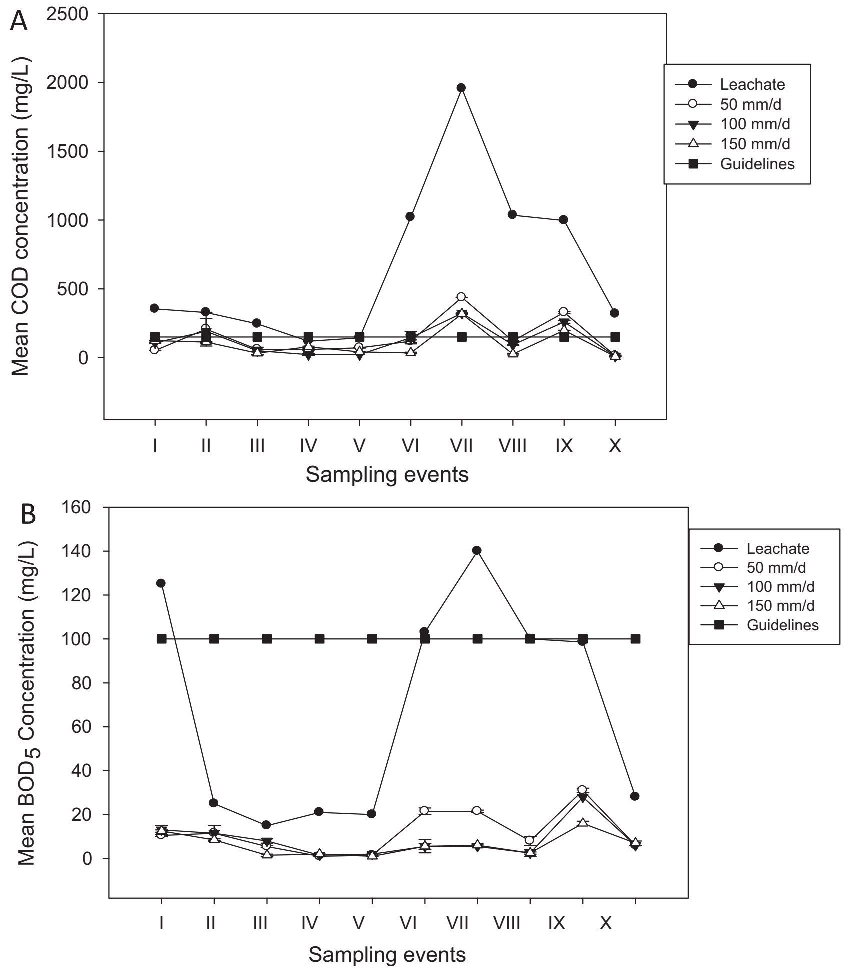

Fig. 3. Organic characteristics of applied leachate and effluent quality for all 10 sampling events for each of the three applied hydraulic loads. (A) BOD_5; (B) COD.

after batch loading to assess the infiltration rate (IR) and changes in nutrients and organics of the effluent. This infiltration rate is given by $D = dV/dt$, with dV being the volume of effluent collected (in litres) and dt being the infiltration time (in minutes). Leachate and effluent samples were collected in polyethylene bottles for chemical analyses and sterilized borosilicate bottles for microbiological analyses.

2.5. Characterization of leachate and effluent

pH, temperature and redox potential were measured in situ using a Hach pH-meter model HQ11d, and conductivity with a Hach Conductimeter model HQ14d. COD was quantified with the close-reflux dichromate reduction method at 150 °C for 2 h followed by a spectrophotometric quantification with a spectrophotometer model Hach DR 2010. BOD_5 was quantified after five days of incubation at 20 °C with Oxytop head gas sensors following inhibition of nitrification. NH_4–N was quantified with the Nessler method, and NO_3–N with the cadmium reduction (NitraVer 5) method for extraction and quantified with a Hach spectrophotometer DR 2010. These methods were carried out as described in Standard Methods for the Examination of Water and Wastewater (American Public Health Association et al., 2005). Total Kjeldahl Nitrogen (TKN) was quantified through wet acid digestion followed by distillation in a Bucchi K-350 distiller and back titration with H_2SO_4 0.1 N (AOAC, 1980). Faecal coliforms and faecal streptococci were quantified with the membrane filtration method on a solid Tergitol–TTC agar–agar and bile–esculine–agar media (American Public Health Association et al., 2005).

The mean concentrations of leachate characteristics applied during the 10 sampling events are reported in Table 1. For

Table 1
Mean value for physico-chemical and bacteriological characteristics of leachate from faecal sludge drying beds that was applied to VFCWs over 10 sampling events. Cameroon and WHO Guidelines are presented as a reference.

Parameters	Mean value and standard deviation	Guidelines for discharge of effluent	
		MINEP*	WHO**
pH	7.13 ± 0.58	6–9	6–8.5
EC (μS/cm)	916 ± 1003	NA	NA
COD (mg/L)	652 ± 558	<200	NA
BOD_5 (mg/L)	156 ± 47	<50	NA
TKN (mg/L)	105.39 ± 31.17	<30	<30
NH_4–N (mg/L)	38.70 ± 20.50		
NO_3–N (mg/L)	81.23 ± 112.73		
PO_4–P (mg/L)	10.50 ± 9.31	<10	<10
Faecal streptococci (Log CFU/100 mL)	4.87 ± 5.15	<3	NA
Faecal coliforms (Log CFU/100 mL)	5.26 ± 5.40	<3.30	<3

* Cameroon guidelines for wastewater effluent discharge from the Ministry of Environment and Nature Protection in Cameroon (MINEP, 2008).
** WHO (2006) guidelines for the safe use of wastewater, greywater and excreta in Agriculture.

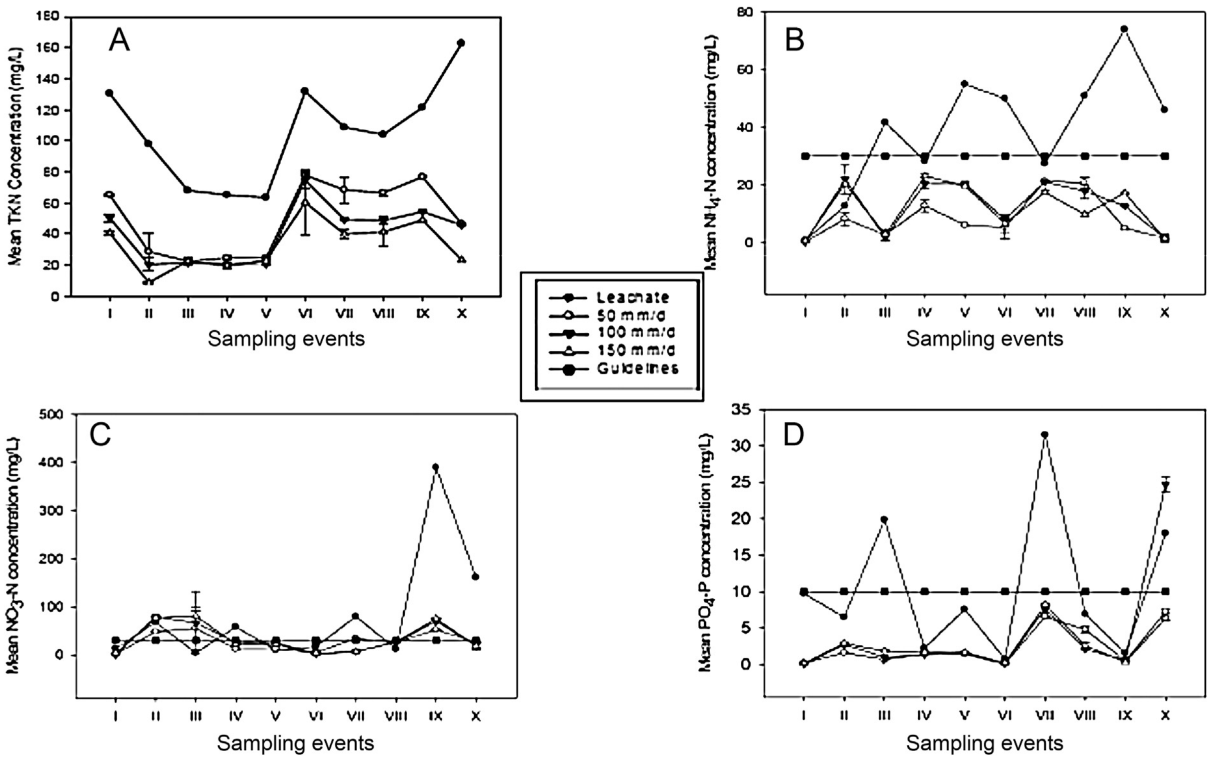

Fig. 4. Nutrient characteristics of applied leachate and effluent for all 10 sampling events for each of the three applied hydraulic loads. (A) TKN; (B) NH_4–N; (C) NO_3–N; (D) PO_4–P. WHO (2006) effluent guidelines are provided as a reference.

reference purposes, these leachate characteristics are accompanied with the Cameroon guidelines for wastewater effluent discharge from the Cameroon Ministry of Environment and Protection of Nature (MINEP, 2008) and the World Health Organization guidelines for the safe use of wastewater, greywater and excreta in Agriculture (WHO, 2006).

2.6. Plant growth

The growth response of *E. pyramidalis* was measured every two weeks during the study by measuring the plant stem height and stem diameter and by counting the plant density. Five plants in each bed were labelled and followed throughout the study for plant height and stem diameter. Plant height was measured from the bed surface to the appearance of terminal young leaves, stem diameter was assessed at the level of the third internode with a calliper. Plant density was counted for the entire bed.

2.7. Rhizospheric bacteria

Soil samples were collected every two weeks in all VFCWs during January–March 2011 (four last sampling events). Three core samples of 15 cm depth by 3 cm diameter were taken at the surface of the VFCWs with a 60 cm^3 modified syringe (cut at the bottom). Samples were analysed in triplicate at the Laboratory of Biotechnology and Environment of the Faculty of Science at the University of Yaounde I. Dilutions of soil samples were spread on a soil agar extract medium (solid culture medium) in petri dishes and incubated for 24 to 48 h at 24–28 °C in a Sartorious microbiological incubator. Black stains were counted using a microscope, according to the Germida (1993) method. The number of rhizospheric bacteria is expressed in log colony forming units (CFU)/g of substrate.

2.8. Statistical analysis

The non-parametric Kruskal–Wallis test were used to compare the effects of the different HLRs applied. Mean ± standard deviation was generally used to express the average of a given parameter and dispersion observed. The level of significance was set to $P < 0.05$. The statistical software Sigma plot 11.0 for Windows was used to performed the tests and to plot the graphs.

3. Results and discussion

3.1. Effluent characteristics

3.1.1. Volume recovered, pH and electrical conductivity

The total volume of effluent collected from VFCWs at 18:00, following application of leachate at 08:00 during the period of study, is reported in Table 2. The collected range of volumes was 5–22, 38–77 and 76–105 mm/d for 50, 100 and 150 mm/d, respectively.

Table 2
Mean and standard deviation of volume (*L*) of effluent collected from VFCWs at 18:00, following application of leachate at 08:00.

Sampling events	50 mm/d	100 mm/d	150 mm/d
I	13.73 ± 1.28	52.35 ± 0.65	105.30 ± 0.70
II	16.63 ± 0.38	46.09 ± 1.41	86.98 ± 1.02
III	8.67 ± 0.33	38.30 ± 3.30	76.05 ± 0.95
IV	7.49 ± 0.49	47.95 ± 1.06	85.03 ± 0.97
V	10.67 ± 0.67	46.62 ± 0	87.06 ± 2.06
VI	5.02 ± 0.02	49.18 ± 0.82	87.50 ± 2.50
VII	8.10 ± 0.90	51.15 ± 0.85	85.05 ± 1.05
VIII	11.45 ± 1.45	59.78 ± 0.78	89.38 ± 1.38
IX	21.90 ± 0.10	72.90 ± 0	96.90 ± 1.90
X	20.45 ± 0.55	77.40 ± 2.40	99.25 ± 1.25

Volume of effluent collected from VFCWs at 18:00, following application of leachate at 08:00.

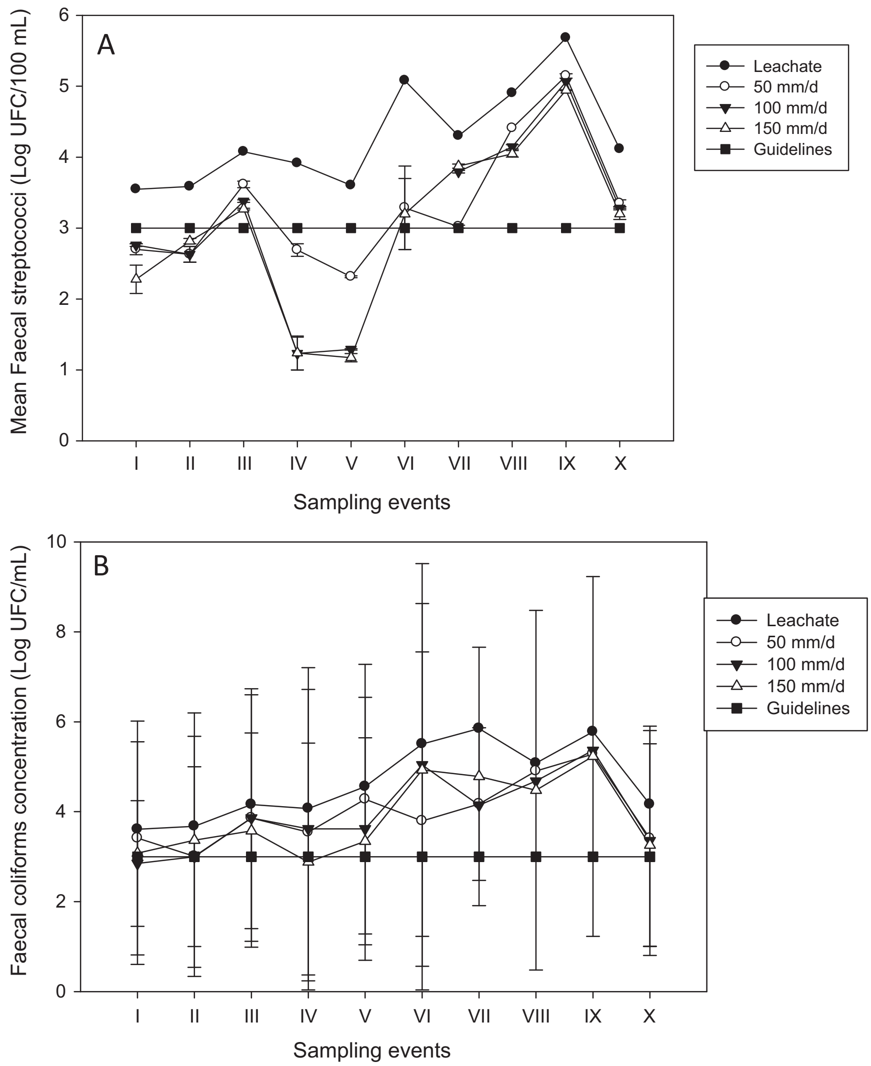

Fig. 5. Microbiological characteristics of applied leachate and effluent for all 10 sampling events for each of the three applied hydraulic loads. (A) faecal streptococci; (B) faecal coliforms. WHO (2006) effluent guidelines are provided as a reference.

Over all 10 sampling events, this is an average of 27%, 55% and 60% of the total applied volumes. The remaining liquid is likely to have been trapped in matrix pore spaces, and lost to transpiration and evapotranspiration as reported by Kadlec and Wallace (2009). The nominal detention time $\left(\tau_n = V_{\text{nominal}}/\text{flow rate}\right)$ was equal to 0.52 d, 0.65 d and 1.3 d for 50, 100 and 150 mm/d, respectively.

During the investigation, the pH values ranged between 6.5 and 8.1 in the leachate and 6.4 and 7.6 in the effluent (Fig. 2A). The electrical conductivity was between 198 and 3820 μS/cm in the leachate and 121 and 1010 μS/cm in the effluent (Fig. 2B). These values are favourable for biological processes, and are compatible with the WHO (2006) guidelines for safe reuse of wastewater in agriculture.

3.1.2. Organic content

On average, there was 92% removal of COD and BOD_5 from the leachate for all loading rates. The greatest removal efficiency was observed with the 50 mm/d loading. However, the differences were not statistically significant ($P > 0.05$). The VFCW polishing step ensured that the effluents were always within the requirements of COD < 200 mg/L and BOD_5 < 50 mg/L (Fig. 3A and B) for the Cameroon guidelines for wastewater effluent discharge from the Cameroon Ministry of Environment and Protection of Nature (MINEP, 2008). These removals are similar to those reported for VFCWs treating wastewater with batch loading, as was done in this study. In fact, for wastewater characteristics almost similar to that of the FS leachate, Prochaska et al. (2007) reported 98% removal

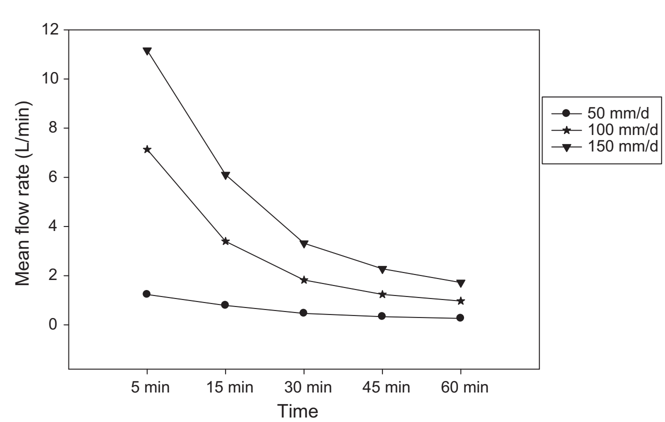

Fig. 6. Infiltration rates reported as volume of collected effluent as a function of hydraulic load and time over a 1 h time period.

of COD with a hydraulic loading rate of 80 mm/d (i.e. from 458 to 9 mg/L) and 92% with a hydraulic load of 170 mm/d (i.e. from 458 to 37 mg/L) in Greece; Molle et al. (2006) reported 94% removal (i.e. from 250 to 16 mg/L) in France. High removal rates of COD are attributed to batch loading, as the unsaturated flow allows oxygen to pass through the matrix during resting periods, which can then be used by rhizospheric microorganisms and results in increased mineralization of organic matter (Molle, 2003; Karathanasis et al., 2003). Suspended organic matter can also be removed by accumulation at the filter surface by the sand layer (Kadlec and Wallace, 2009) although it was not the case in this study, since no accumulation of solids was observed.

3.1.3. Nutrients

The removal of TKN was effective for all loading rates, ranging from 79 to 89% (Fig. 4A). The NH_4–N contents was also substantially reduced throughout the study period (Fig. 4B) with a removal performance of 70–93%. The nitrate content was very variable over the study with most often, an increase above the leachate concentration (Fig. 4C). Although NO_3–N is an indicator of good functioning of the VFCWS beds, it contributed to raise the total nitrogen concentration. The reduction in nitrogen at the outlet VFCW beds is likely due to a combination of plant uptake (mainly NH_4–N), adsorption, ammonia volatilization, and denitrification (Reddy and Patrick, 1984). Although the removal of nitrogen was the greatest at 50 mm/d, no statistically significant differences were seen between the different hydraulic loading rates ($P > 0.05$). For six sampling events over 10, the effluents TKN were still above the Cameroon guidelines for wastewater effluent discharge in the environment (<30 mg/L) (MINEP, 2008). Failing to meet the nitrogen discharge standards has been reported elsewhere for VFCWs, unless coupled with another treatment step (Vymazal, 2007). However, since this nutrient is beneficial for plant grow, the nitrogen effluent concentrations obtained could still be suitable for irrigation if carefully monitored for overloading and microbial standards.

Possibilities to increase the performance of VFCWs for nitrogen removal include periodic flooding of the beds to create alternating anaerobic and anoxic conditions to achieve nitrogen removal (nitrification and denitrification) (Panuvatvanich et al., 2009). Another possibility to increase nitrogen removal would be to impound the drainage layer of a vertical flow bed, such as described by Langergraber et al. (2008) for a 2-stage VFCW system. Increased nitrogen and phosphorus removal has also been reported with hybrid systems which use for example VFCWs followed by horizontal flow constructed wetlands (HFCW) or free water surface (FWS) wetlands. For these systems, hydraulic loads of 200–1200 mm/d have been employed (Johansen et al., 2002; Weedon, 2003). Another possibility, is if the effluent is used for reclamation in agriculture as opposed to direct discharge, in which case the nutrients can be of beneficial use.

The nitrogen removal efficiency obtained in this study is greater than those that have generally reported with VFCWs for wastewater treatment in Europe (≤40%) (Molle et al., 2005; Kadlec and Wallace, 2009), most likely because the warm climate increases the metabolic activity of plants and microorganisms, as well as nitrogen transformations.

The VFCW was effective at meeting Cameroon guidelines for wastewater effluent discharge (MINEP, 2008) as well as WHO's (2006) guidelines for safe use of wastewater in agriculture for orthophosphate of <10 mg/L at all loading rates (Fig. 4D). No significant difference was observed between the different hydraulic loads ($P < 0.05$) with performances ranging from 84 to 93%. Retention of phosphorus in VFCW is common: for example 75 g/m^2/yr as reported by Vymazal (2007). Phosphorus reduction in VFCWs was most likely due to sorption and plant uptake than accumulation of sludge at the filter surface (Kadlec and Knight, 1996; Reddy and

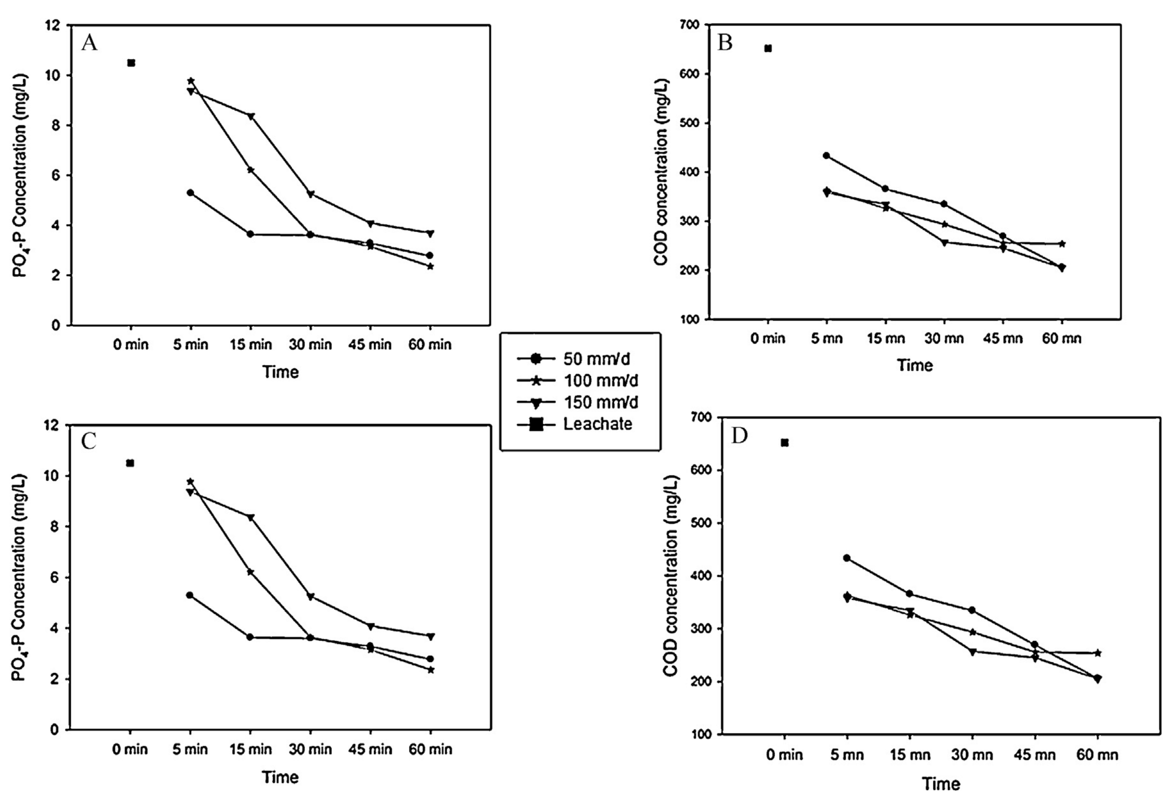

Fig. 7. Characterization of applied leachate and resulting effluent as a function of time over 1 h and hydraulic loading. (A) PO_4–P; (B) COD; (C) NH_4–N; (D) NO_3–N.

D'Angelo, 1997) since no accumulation of sludge was observed in this study.

3.1.4. Faecal indicators

As shown in Fig. 5B, respectively, faecal streptococci and faecal coliforms were slightly reduced for all hydraulic loads, with an average removal less than 2 log units. No statistical difference was observed between hydraulic loading rates ($P > 0.05$). The effluent values were still higher than those recommended by WHO for reuse in non-restricted agriculture (WHO, 2006). The low reduction of faecal indicators observed here can be explained by the high strength of faecal indicator as the removal efficiency decreases with the increase in the leachate like observed on the graphs. Similar low removals have been reported for VFCWs treating wastewater (Kadlec and Knight, 1996; Ottova et al., 1996). Reductions are thought to be a result of reduced organic matter depriving bacteria of growth substrate (Vymazal, 2007). Plants are also thought to aid the process through the secretion of antibiotics (tannins and gallic acid), and through oxygen transfer to the rhizosphere (Gersberg et al., 1989; Kadlec and Wallace, 2009) although the media in this study was not water saturated and thus, the oxygen supply through plants would be negligible. Williams et al. (1995) also suggested removal of pathogens to be highly dependent on the grain size distribution of the filter media used. This inefficient bacterial treatment through this process implies that if the effluent is to be used for reclamation in irrigation, then a multiple barrier approach should be applied for public health protection of pathogens.

3.2. Infiltration rate

From the first six sampling events used for the measurement of the infiltration rate, an average of 81%, 95% and 88% of the total volume that was collected after 10 h, had percolated through the bed during the first hour following application of leachate at 50, 100 and 150 mm/d, respectively, as can be seen by comparing Fig. 6 to Table 2. The hydraulic flow rate was greatest at 5 min of infiltration and decreased with time to be quite low by 60 min, confirming good drainage of the VFCWs for all hydraulic loads. Similar drainage of VFCWs has been reported by Dittner et al., 2004 and Molle et al. (2006) for 24, 48 and 95 mm/d. The initial peak infiltration is explained by the head pressure gradient exerted on the filter surface and by the progressive saturation of the filter pores, especially the sand layer (Molle et al., 2006).

Concentrations of PO_4–P, COD, NH_4–N and NO_3–N in the effluent over the first hour are reported in Fig. 7A–D, respectively. The concentrations in the effluent was reduced with time for all

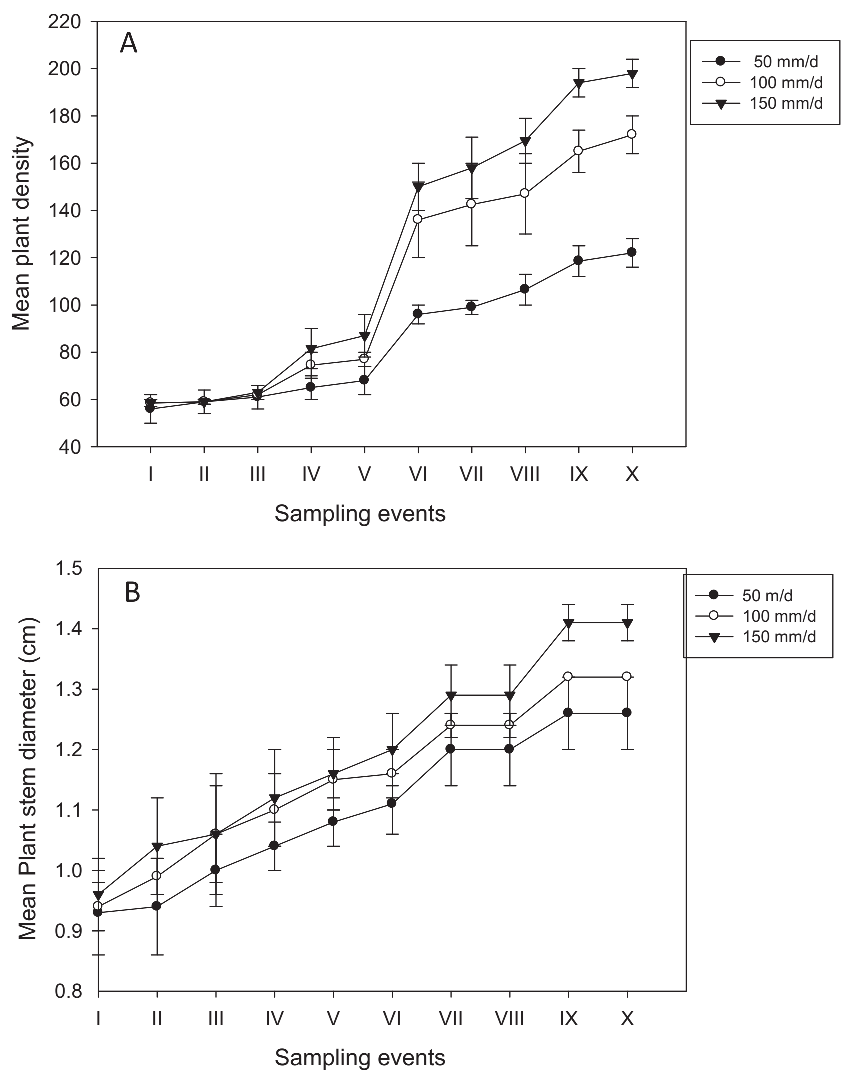

Fig. 8. Metrics of plant growth as a function of hydraulic load for all 10 sampling events. (A) mean plant density; (B) mean stem diameter.

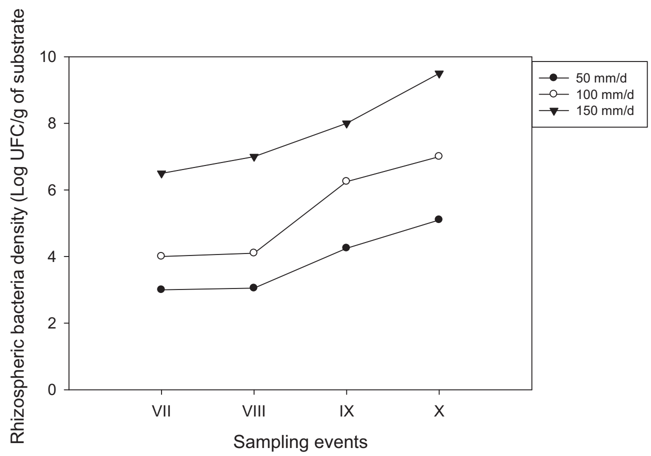

Fig. 9. Concentration of rhizospheric bacteria as a function of hydraulic load for sampling events 7 to 10.

hydraulic loading rates. This could be explained by an increased contact time with rhizospheric microorganisms and filter media (Vymazal, 2001), and/or by a flushing of the filter, where the hydraulic load washes out nutrients that were retained in the filter from the previous application (Molle et al., 2006).

3.3. *Plant growth*

Plant growth responses measured as plant density, plant stem height and stem diameter are reported in Fig. 8A–C, respectively. There was no significant difference ($P > 0.05$) in plant density, plant stem height and plant stem diameter between the different hydraulic loads. After four months of operation, these parameters increased proportionally to the increase in hydraulic loads. The rapid increase in plant density, plant stem diameter and height could be justified by adequate moisture being available for plant growth and by the presence of excess nutrients, which are mineralized and rendered bio-available by the bacteria that are present (Vymazal, 2007; Brix, 1997). This observation was also mentioned by Tripathi et al. (1991) who obtained a close relation between the increase in concentration of nutrients in wastewater and the absorption by plants for growth. Kengne et al. (2008) also reported an increase of biomass with hydraulic load, with a plant production of 165 to 264 t DW/m^2/yr in faecal sludge drying bed. These results are promising for the production of fodder production from faecal sludge treatment. However, this plant growth could be increased with the increase in hydraulic load or number of loading per week to reduce the potential plant stress that is undergone with only a single loading per week.

3.4. *Rhizospheric bacteria*

Rhizospheric bacteria were measured as in indication of their contribution to treatment performance. The mean density of rhizospheric bacteria increased with time and with the increase in hydraulic load as illustrated in Fig. 9. Hatano et al. (1993) reported 2.7×10^6 UFC/g of substrate in subsurface flow constructed wetland treating wastewater in Europe. In this study, the increase in rhizospheric bacteria seems not to influence the treatment performance. This increase could be due to plant growth, as the higher numbers of rhizospehric bacteria are found in beds having higher plant density, and higher plant density also is correlated to greater root biomass (Brix, 1997). It could also be due to the increased water and nutrient availability.

4. Conclusions

This is the first time that treatment performance was assessed for the use of VFCWs in the treatment of faecal sludge leachate in sub-Saharan Africa. The study shows that planted drying beds for faecal sludge dewatering followed by VFCWs for leachate treatment is a very promising technology for comprehensive faecal sludge treatment, and can generate products for end use. This treatment scheme is low-cost, relatively easy to operate and efficient in reducing physico-chemical pollutants. The treatment scheme was able to meet the WHO and MINEP guideline thresholds for discharge or reuse in non-restricted agriculture at loading rates of 50, 100 and 150 mm/d for all the monitored parameters except nitrogen and faecal bioindicators, which were slightly above the guidelines. This could potentially be managed by adding a second bed in series, or by periodically flooding the beds to achieve nitrogen removal. These results are important for the operation of VFCWs in faecal sludge management, as it demonstrates that VFCWs can be operated effectively at any of these loading rates, are adaptable to different size treatment schemes, and are resistant to changes in loading rates. They can therefore be low-cost and effective treatment options for unplanned settlements in sub-Saharan Africa, which have unpredictable and variable rates of sludge production. The results also illustrate that if plant production is a goal of treatment, increased loading rates are preferable, as they still achieve treatment goals and result in greater macrophyte production.

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