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PHOSPHATE AND AMMONIUM DISTRIBUTION IN A PILOT-SCALE CONSTRUCTED WETLAND WITH HORIZONTAL SUBSURFACE FLOW USING SHALE AS A SUBSTRATE

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Abstract—Phosphate (P) and ammonium (N) distributions were investigated in a pilot-scale constructed wetland system (CWS) with horizontal flow. Shale was selected as a substrate, on the basis of its P adsorption capacity as well as its suitability for plant growth, investigated in an earlier study. The system was set up in a greenhouse, and comprised of four tanks with and four tanks without *Phragmites australis* (common reed). Sampling ports were installed along the length of each tank, and at three different depths, in order to facilitate three-dimensional monitoring of nutrient distribution over the period of study (11 months).

In the planted systems, $H_2PO_4^-$-P (ortho-P) and NH_4^+-N concentrations were low (0.5–1.0 g m^{-3}) at all depths throughout their length. Generally, ortho-P and NH_4^+-N concentrations decreased exponentially along the transect from the tank inlet to the outlet. In the unplanted systems, higher values (0.5–16.5 g m^{-3}) of ortho-P were observed. NH_4^+-N in the unplanted systems was relatively high (10–30 g m^{-3}) throughout the period of investigation. In both planted and unplanted tanks, NO_3^--N concentrations were very low at the inlets (0.02–0.05 g m^{-3}), and increased only slightly towards the outlets. Although the presence of *P. australis* had a significant effect ($p < 0.05$) on P and N concentrations at all depths and along the length of the tanks, the nutrient distribution followed the same trend as in unplanted tanks.

The results were consistent with the theoretical removal model which predicts an exponential decrease in pollutant concentrations to a background value approaching zero along the transect from wetland inlet to the outlet. A modification of the model was suggested where a first order area-based reduction rate constant k (m d^{-1}) would be replaced with ($k_p + k_s$), k_p representing the rate constant for the removal by the plants, and k_s the rate constant for the removal by the substrate, and associated microbial populations they support. Approximate values for $k_p + k_s$ of 0.084 m d^{-1} (P) and 0.065 m d^{-1} (N) and for k_s of 0.069 m d^{-1} (P) and 0.034 m d^{-1} (N) were obtained.

The hydraulic residence time, flow characteristics and permeability of the shale was investigated by a bromide (Br^-) tracer. The tracer breakthrough curves showed a similar pattern in all tanks, with about 66% of the flow occurring through the bottom zone (0.35 m). The actual hydraulic residence time (6 days) was slightly higher than the nominal one.

The tracer study also enabled the calculation of the volumetric rate constants, k_v and approximate values of 0.075 d^{-1}(P) and 0.060 d^{-1}(N) were obtained for planted and 0.061 d^{-1}(P) and 0.020 d^{-1}(N) for unplanted tanks.

The results obtained from this pilot scale study, provide a better understanding of nutrient distribution and flow patterns that take place within CWS, and should enable more rational design parameters to be developed that help to optimise future large-scale systems. © 2000 Elsevier Science Ltd. All rights reserved

Key words—phosphate, ammonium, *phragmites australis*, shale, area rate constants, hydraulic residence time, constructed wetland systems

INTRODUCTION

Because of the need for data and information on the performance of constructed wetland systems (CWS), most of the studies to date have focused on the measurements of what was coming into and out of the systems. Theoretical removal models based on such data have been developed principally for

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biological oxygen demand (BOD) and suspended solids (SS) removal, with the suggestion that the same kinetics could apply to the removal of ammonium (Bavor *et al.*, 1995; Reed *et al.*, 1995; Kadlec and Knight, 1996). The form of the *first-order areal model* used by all these workers involves an exponential decrease in pollutant concentrations to a background value approaching zero along the transect from wetland inlet to wetland outlet. The model can be expressed as follows:

$$q\frac{\mathrm{d}C}{\mathrm{d}x} = -k(C - C^*) \quad (1)$$

where q is the hydraulic loading rate defined as the ratio of the inflow (Q, m^3 d^{-1}) and the surface area (A, m^2); x is the fraction of distance from inlet to outlet; C is the pollutant concentration of the system (g m^{-3}); C^* is the background pollutant concentration (g m^{-3}); and k the first order area removal rate constant (m d^{-1}).

An alternative approach, based on the concept of water retention time has been formulated by Kadlec and Knight (1996):

$$\frac{Cx - C^*}{Ci - C^*} = \mathrm{e}^{-k_v T/\xi h} \quad (2)$$

where k_v is the volumetric rate constant (d^{-1}); T the wetland retention time (d); ξ the wetland porosity; and h the depth of the system (m).

Due to the lack of available data on internal transect profiles, it is not possible to adequately verify the theoretical removal and, therefore, it is not clear whether the model applies to phosphorus (P) in subsurface flow wetlands (Kadlec and Knight, 1996). Green and Upton (1993) did observe decreasing P profiles through a series of subsurface flow units at Little Stretton, UK, but their sampling was conducted over only two 1-week periods. Cooper and Green (1995) emphasised the need for more information on pollutant distribution within CWS, especially during cold and potentially freezing weather. In addition, Bavor *et al.* (1995) also stated that because of the lack of data and understanding of the key mechanisms involved in removal processes, theoretical removal models tend to be system-specific.

Kadlec and Knight (1996) stated that the design of CWS based on predictable or assumed sets of conditions can lead to an overestimation of the wetland treatment performance. In addition, they pointed out that the effects of a flow regime on P removal should be taken into account in terms of rate constants derived from experimental data. Determination of the residence time distribution (RTD) model parameters such as the volumetric rate constants k_v and actual residence times T (equation (2)), enables the quantification of these effects to be made. Currently, there is a lack of such data due to the fact that only a small proportion of the existing wetlands have been tracer tested (Kadlec and Knight, 1996; Vymazal *et al.*, 1998). Therefore, it is necessary to establish whether exponentially decreasing concentration profiles exist in constructed wetland systems.

In this paper, we present experimental data on spatial distribution of ortho-P and ammonium (NH_4^+-N) within the liquid phase in a CWS, collected on four occasions over a period of 11 months. Flow characteristics of the systems are also discussed. It is believed that this kind of information on nutrients distribution and flow patterns of the systems will lead to improved design in the future (Drizo, 1998). The data were used to test the theoretical models of removal (see Eqs (1) and (2)). In addition, the contribution of *P. australis* to these processes was assessed.

METHODS

The full description of the greenhouse set-up is given in Drizo *et al.* (1997) and Drizo (1998) and is summarised as follows. Eight 250 l tanks, 1 m long × 0.5 m wide × 0.5 m deep, were each filled with 200 kg of shale to the depth of 0.35 m. The volume of the material in each tank was 165 l with a pore space of approximately 50 l. Four tanks were planted with *P. australis*, four were unplanted. Each tank was connected to a reservoir containing 250 l of synthetic sewage, via a peristaltic pump, to distribute the wastewater through the tank. The pump was set to operate four times each day for 50 min, at a rate of 3 l h^{-1}. With this arrangement, 10 l day^{-1} of waste water passed through each of the tanks, giving a residence time of 5 days.

The synthetic sewage (made up by mixing seven different components: bacteriological peptone, meat extract, urea, sodium chloride, calcium chloride, magnesium sulphate and dipotassium hydrogen phosphate) after dilution contained approximately 106 g m^{-3} organic C, 46 g m^{-3} N and 5 g m^{-3} P (DoE, 1981). In order to investigate the capacity of the systems for ortho-P and NH_4^+-N removal over a wider range of nutrient input concentrations, the contents of N and P in the synthetic sewage were modified (Drizo, 1998). The average inlet concentrations were 20–40 g ortho-P m^{-3} (0.4–0.8 g m^{-2} day^{-1}), 16-24 g NH_4^+-N m^{-3} (0.32–0.48 g m^{-2} day^{-1}) and 0.2–0.4 g NO_3^--N m^{-3} (0.004–0.008 g m^{-2} day^{-1}). Biological oxygen demand (BOD) values ranged between 30 and 60 g m^{-3} day^{-1} (Drizo, 1998).

Vertical plastic sampling tubes (internal diameter 8 mm) were placed at 0.2, 0.4, 0.6 and 0.8 m from the inlet along the length of each tank and to three different depths: 0.35 m (bottom of the tanks), 0.2 m and 0.1 m below the substrate surface. A closely-fitting fibre-glass rod was placed in each tube in order to prevent it from filling with water between samplings. Waste water samples were taken from the bottom of each tube, using 50 ml syringes to which suitable lengths of small-bore plastic tubing had been attached. The measurements took place on four occasions: November 1995 (5 months after the establishment of the CWS), December 1995, March and May 1996. On every occasion, the syringes were rinsed with distilled water between samplings. In April 1996, the inlet concentration of phosphate was increased three-fold (to 120 g m^{-3}), and that of ammonium two-fold (to 40 g m^{-3}), with the aim of reaching saturation of the substrate.

Ortho-P was determined using a spectrophotometer (model SP6-500 UV, Pye Unicam, Cambridge, England; limits of detection at 0.2 g m^{-3}) according to the standard ammonium molybdate method (DoE, 1980). NH_4^+-N was measured using the method devised by Crooke and Simpson (1971) and NO_3^--N by the method described by Best (1976), using a Chemlab Instruments continuous flow analyser with limits of detection between 0.1 and 0.2 g m^{-3}. Additional samples were taken for measurements of pH, redox potential (E_h) and temperature (t) in order to investigate the effects of these parameters on P and N removal from waste water. pH and E_h values were measured by standard technique described by Rowell (1994), using a conventional glass electrode, a calomel reference electrode and a platinum electrode. Temperature was measured using a combined oxygen electrode and temperature recorder.

A bromide tracer study was conducted in spring 1996, 9 months after establishment of the constructed wetlands. A solution containing 20 g l^{-1} of Br^- (as KBr) was prepared and 2.5 l were added to each tank, during one working cycle (50 min) of the peristaltic pumps. The first set of samples was collected from each of the observation points (plus the outlet) one hour after the tracer application, using 50 ml syringes. The following samples were taken once per day for 6 days. Bromide concentrations were determined using an ion-selective electrode (Adriano and Doner, 1982). Flow was maintained at a constant rate, and mean residence time was obtained from plots of tracer concentration vs time at the sampling points.

RESULTS

Phosphate distribution

Ortho-P concentrations showed a very rapid decrease from inlet to outlet in the planted tanks (Fig. 1). Values were extremely low in planted tanks (0.5–1 g m^{-3}) at all depths and all distances during the first 8 months of investigation, rising to 3 g m^{-3} in May 1996. In the unplanted tanks, higher values (5.0–16.5 g m^{-3}) of ortho-P were observed (especially at 0.2 m and 0.4 m distances), the highest tending to be at the bottom of the tanks (0.35 m) (Fig. 1).

In order to test the effects of depth, distance from the inlet and presence of plants on P distribution, split-plot analyses of variance with tanks treated as main plots were performed (Sokal and Rohlf, 1981), using SAS Stat statistical software (SAS Institute, 1985). The effect of depth on P concentration was significant ($p < 0.01$) in both planted and unplanted tanks throughout the period of investigation, with concentrations being higher at the bottom of the tanks (Table 1, Fig. 1). The effect of distance from the inlet was also significant on all sampling occasions, in both planted and unplanted

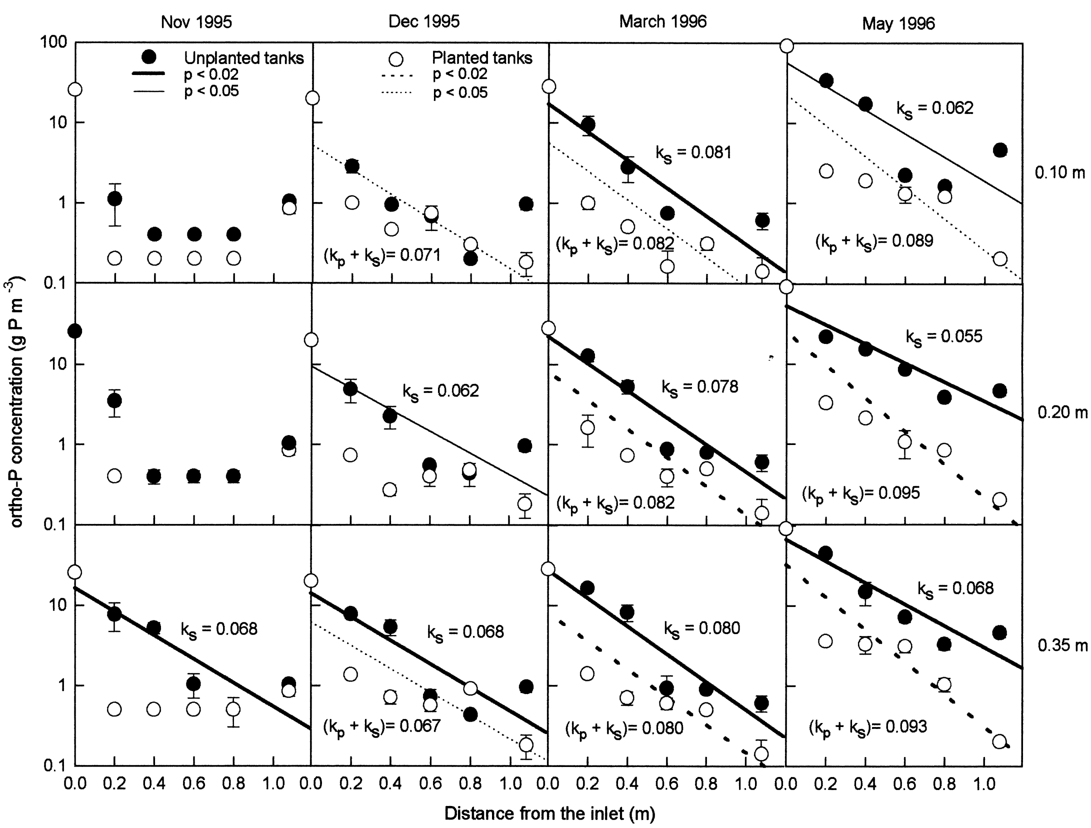

Fig. 1. Spatial distribution of ORTHO-P at three depths in the liquid phase in the unplanted (●) and planted tank (○) for the period November 1995–May 1996. Vertical bars denote standard deviation (SD). Fitted line represents the relationship between ORTHO-P concentration and distance from the inlet.

tanks, with concentrations decreasing as distance increased ($p < 0.01$, Table 1). The presence of plants had a significant effect at all depths and distances throughout the experiment.

Ortho-P concentrations were plotted on a logarithmic scale to test whether the decrease with distance from the inlet was exponential. In planted tanks this was true in eight out of 12 cases, and in unplanted tanks in nine out of 12. Regression lines and associated area rate constants k (equation 1) are presented in Fig. 1. The rate constants k for unplanted and planted tanks were compared for each sampling occasion, using a t-test Sigmastat statistical software (Kuo *et al.*, 1992) (data not shown). A significant difference ($p < 0.02$) occurred only on the last sampling occasion, in May, with rate constants being higher in the planted tanks.

Temperature values ranged between 6.6 and 14.9°C with values remaining above 5°C even in December, when outside temperature values decreased to below 0°C. pH values ranged between 5.5 and 6.5 being fairly stable at all depths and along all lengths of the tanks, while redox potentials (E_h) were between 60 and 120 mV, increasing in values along the transect from the inlets towards the outlets (Drizo, 1998).

Ammonium distribution

NH_4^+-N concentrations followed a similar trend of rapid decrease from inlet to outlet in planted tanks (Fig. 2). Values were very low (below 2.5 g m^{-3}) during the first eight months of wastewater application at all distances and depths, but then gradually increased to almost 5.0 g m^{-3} between months 8 and 10 (which still represented an 87.5% reduction when compared with the input load rates). The levels of NH_4^+-N in unplanted tanks were relatively high at 0.2 m, 0.4 m and 0.6 m distances throughout the experiment, and rose from 10–15 g m^{-3} in November to 20–30 g m^{-3} in May 1996, the latter representing only a 25% reduction.

As for ortho-P, split-plot analyses of variance were employed to test the effects of depth, distance from the inlet and presence of plants on NH_4^+-N distribution, using SAS Stat statistical software (SAS Institute, 1985). The effect of depth was significant on two sampling occasions, in November 1995 and May 1996 (Table 1). The effect of distance on NH_4^+-N distribution was significant throughout the experiment, with lower concentrations at greater distance ($p < 0.01$, Table 1). In addition, the presence of plants had a significant effect ($p < 0.01$) at all depths and distances, planted tanks having the lower concentrations throughout the period of investigation (Table 1).

NH_4^+-N concentrations were also plotted against distance from the inlet on a logarithmic scale and these data for both unplanted and planted tanks are presented in Fig. 2. The regression lines show that NH_4^+-N concentration in unplanted tanks decreased exponentially with respect to distance throughout the experiment, while in planted tanks this was the case in nine out of 12 occasions. The associated rate constants k are presented in Fig. 2.

To test whether there was a significant difference in rate constants between planted and unplanted tanks, a t-test was performed for each sampling occasion, using Sigmastat statistical software (Kuo *et al.*, 1992). There was a significant difference ($p < 0.01$) throughout, with constants being higher in the planted tanks.

Nitrate distribution

The input load of NO_3^--N was very low (0.5–1.0 g m^{-3}) and consequently, the concentrations remained low at all distances and depths of the system throughout the experiment, often falling to 0.1 g m^{-3}, which was at the limit of detection of the analysis.

Similarly to the data for ortho-P and NH_4^+-N, split-plot analysis of variance with tanks treated as main plots was performed (Sokal and Rohlf, 1981). It showed that the effect of depth had a significant influence on nitrate distribution only on one sampling occasion, in December 1995 (Table 1). A tendency for concentrations to be 30% higher in planted than in unplanted tanks at 0.2 m and 0.35 m depth and at 0.5 m distance from the inlet was observed. The effect of distance and the presence of plants were apparent throughout the period of investigation (Table 1).

Flow characteristics

The overall recovery of bromide tracer was good, representing 88% in planted and 84% in unplanted tanks respectively, of which the greatest proportion (66.5 and 65%) was in the bottom layer (Fig. 3). Surface flow was not observed.

Table 1. Results of split plot analysis of variance on the effects of depth, distance from the inlet (length), and presence of *P. australis* on $H_2PO_4^-$, NH_4^+–N, and NO_3^-–N; ($n = 48$)[a]

Parameter	November (5 months)			December (6 months)			March (9 months)			May (11 months)		
	D	L	P/u	D	L	P/u	D	L	P/u	D	L	P/u
$H_2PO_4^-$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.05$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$
NH_4^+-N	$p < 0.05$	$p < 0.01$	$p < 0.01$	ns	$p < 0.01$	$p < 0.01$	ns	$p < 0.01$	$p < 0.01$	$p < 0.05$	$p < 0.01$	$p < 0.01$
NO_3^--N	ns	$p < 0.01$	$p < 0.05$	$p < 0.01$	$p < 0.01$	$p < 0.01$	ns	$p < 0.01$	$p < 0.05$	ns	$p < 0.05$	$p < 0.01$

[a]D = depth; L = length; P/u = planted/unplanted.

In both unplanted and planted tanks, there was almost no flow at 0.10 m depth below the surface, which was expected, given that the inlet pipe was at the depth of 0.20 m. The observed mean hydraulic residence time at the outlets and at all depths, was approximately 6 days, which was slightly higher than the theoretical nominal residence time (5 days) (Fig. 3).

To test the effects of depth and the presence of plants on flow characteristics, two-way ANOVA analyses were performed, using SigmaStat statistical software (Kuo *et al.*, 1982). Each of the treatments had two replicates and tests were performed on absolute values. The results of the analyses showed that there were major hydraulic differences ($0.02 < p < 0.05$) between the bottom of the tanks (0.35 m) and the two upper zones (0.20 and 0.10 m) in both planted and unplanted tanks. Although the tracer breakthrough curves follow a similar pattern (Fig. 3) the difference in Br^- concentrations in planted and unplanted tanks was significant ($p < 0.02$). This implies the presence of different flow characteristics in the unplanted and planted tanks.

DISCUSSION

The decrease in ortho-P concentrations from the inlet towards the outlet conformed well with the exponential profile predicted by equation (1). The pattern was similar in both unplanted and planted tanks, the latter showing a more rapid decrease, in particular at 0.2 m from the inlet.

The tendency for rate constants to be higher in planted tanks (Fig. 1) in May 1996 could be attributed to the enhanced plant uptake of P at the beginning of the growing season. Overall, rate constants were 2.4 times higher than typical values of $k = 12$ m y^{-1} ($k = 0.033$ m d^{-1}) reported by Kadlec and Knight (1996). However, it should be taken into account that the latter values were calculated for full scale systems and greater hydraulic loading rates. The comparison with other pilot scale systems is not possible due to the lack of available data in the literature.

The fact that the treatment efficiency remained very high throughout (95–98%), with virtually complete P removal by the time the wastewater reached the outlet, can be explained by a very high adsorp-

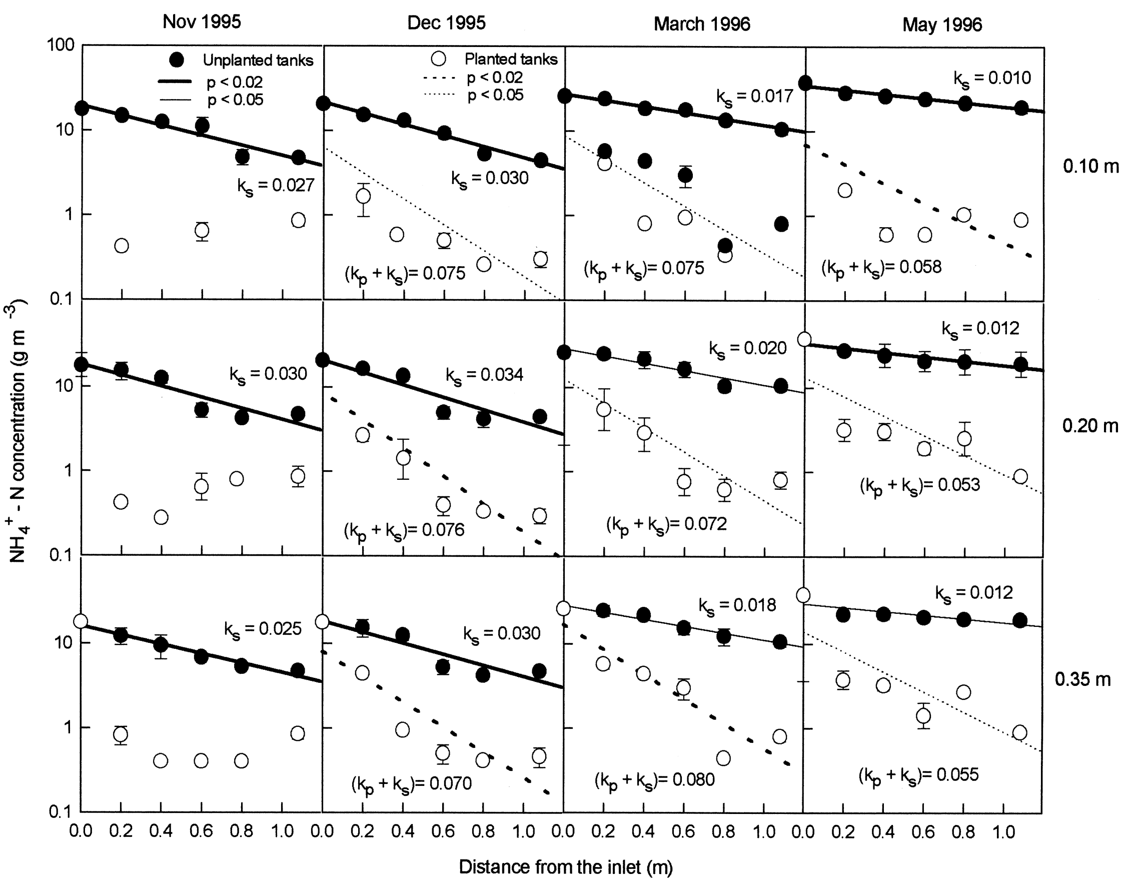

Fig. 2. Spatial distribution of NH_4^+-N at three depths in the liquid phase in the unplanted (●) and planted (○) tank for the period November 1995–May 1996. Vertical bars denote standard deviation (SD). Fitted line represents the relationship between NH_4^+-N concentration and distance from the inlet.

tive capacity of shale for P (0.65–0.7 g P kg^{-1}) determined in laboratory investigations (Drizo *et al.*, 1999).

An exponential decrease in NH_4^+-N concentrations along the transect from the inlet to the outlet was in accordance with the theoretical models developed by Bavor *et al.* (1995) and Kadlec and Knight (1996). As for P removal, contact time may play a major role in the distribution within a CWS. The rate constants in planted tanks were on average three times higher than those of unplanted ones (Fig. 2), indicating a more rapid removal of ammonium in the planted tanks. However, the overall area rate constants for NH_4^+-N (0.065 m d^{-1}) in planted tanks were 1.3–1.4 times lower than the value of 0.082 m d^{-1} for a pilot scale wetland systems reported by Zhu (1998) and 0.093 m d^{-1} for full scale subsurface CWS reported by Kadlec and Knight (1996).

Under anoxic conditions, NH_4^+-N is stable and may be adsorbed by the substrate or used by plants and microorganisms (Sikora *et al.*, 1995). The results from this study show that, overall, ammonium removal in unplanted tanks was only 25–45%. This low value suggests that nitrification/denitrification processes may have been limited by inadequate microbial activity in an unplanted mineral medium. The fact that virtually all N was removed in planted tanks, even when loading rates were doubled towards the end of the experiment, confirms the potential of shale-based CWS for N removal from wastewater.

An earlier study showed that variations in temperature, pH and E_h values did not account for the variations in either ortho-P and NH_4^+-N removal rates either in planted or unplanted tanks (Drizo, 1998).

As planted tanks showed more rapid decreases in nutrient concentrations, with the area rate constants being significantly higher in particular in the case of N (p < 0.01), the theoretical removal model (1) could be refined to:

$$q = \frac{dC}{dx} = -(k_p + k_s)(C - C^*) \qquad (3)$$

where k_p represents the rate constant for the removal by the plants and associated microbial populations, and k_s the rate constant for removal by the substrate and its microbial populations.

In the case of unplanted tanks, the equation of both P and N removal is simplified to:

$$q\frac{dC}{dx} = -k_s(C - C^*). \qquad (4)$$

The experimental data from this study are consistent with the exponential model and give mean values for $k_p + k_s$ of 0.084 m d^{-1} (P) and 0.065 m d^{-1} (N) and for k_s of 0.069 m d^{-1} (P) and 0.034 m d^{-1} (N).

Input loading rates of nitrate were very low in this study (0.5–1 g m^{-3}). It is not known how effectively the system would have reduced NO_3^--N concentrations at a higher input loading rate, and this area merits further investigation. A tendency for NO_3^--N concentrations to be 30% higher in planted

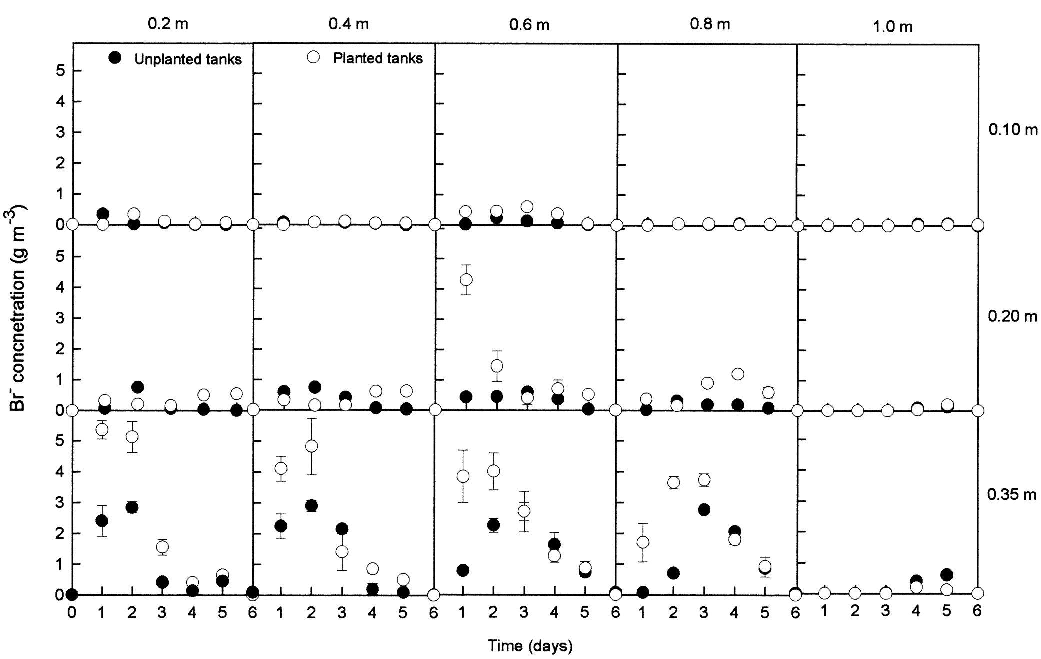

Fig. 3. Tracer concentration vs time in the unplanted (●) and planted (○) tank for the period 29 March–4 April 1996. Vertical bars denote standard deviation (SD).

than in unplanted tanks (at 0.2 and 0.35 m) could be attributed to the roots and rhizomes development, which was more extensive at these depths (Drizo, 1998).

The fact that the major portion (66%) of tracer travelled through the bottom layer in both planted and unplanted tanks suggests the occurrence of a preferential flow in the bottom section of the wetland. This could have a two-fold explanation: firstly, the inlet pipe was set at a depth of 0.2 m, contributing to the flow attenuation in the upper layer. Secondly, a difference in the density and the way that shale particles were packed inside the tanks at the time of the setting up the greenhouse system could have favoured channelling and the occurrence of preferential flows. If there had been preferential flow, one would have expected only one third of the wetland pore volume to be available for the waste water flow, resulting in the actual residence time being shorter then the nominal one. This would also greatly reduce the treatment efficiency of the systems (Sanford *et al.*, 1995). However, the tracer breakthrough curves in both planted and unplanted tanks show that the actual residence time (Fig. 3) was slightly higher (6 days) than the nominal one (5 days). This discrepancy remains unexplained.

The negative effect that plants may have on the hydraulic regime of the wetlands is reported by several authors (Bowmer, 1987; Breen and Chick, 1995; Marsteiner *et al.*, 1996). Bowmer (1987) suggested that plants may influence system performance through channelling flow around the root mass thus causing short-circuiting. Breen and Chick (1995) studied the rootzone dynamics in constructed wetlands and concluded that root biomass developed in the upper zone of the wetland system naturally directs flow to the lower zone, making it the path of least resistance. Marsteiner *et al.* (1996) hypothesised that plant roots may significantly reduce the porosity and therefore overall retention time in subsurface flow wetlands. However, the fact that the major portion of the flow travelled through the bottom regions in both planted and unplanted tanks in this study suggests that development of root biomass may not necessarily be the only cause of preferential flows in wetland systems. Other factors such as the location of the inlet zone (Cooper, 1993) or difference in chemical gradients along the wetland (Breen and Chick, 1995) should also be taken into account.

Volumetric rate constants $k_{\dot{V}}$, determined from the RTD model (equation (2)), were lower than area rate constants in both planted and unplanted tanks. Approximate values of 0.075 d^{-1}(P) and 0.060 d^{-1}(N) were obtained for planted and 0.061 d^{-1}(P) and 0.020 d^{-1}(N) for unplanted tanks. If full scale shale based systems were to be designed in future, the effects of non ideal flow patterns on nutrients removal should be accounted for.

If we assume the inlet ortho-P and NH_4^+-N concentration of 20 g m^{-3} and 30 g m^{-3}, respectively, and the hydraulic loading rate of 0.02 m^{-3} m^{-2} d^{-1}, the rate constants derived from this pilot scale study suggest that 95% of ortho-P and 90% of NH_4^+-N removal could be achieved in a full scale system with the shale material.

CONCLUSIONS

Data of this kind from model systems are valuable in providing guidelines for the design of full-scale wetlands. A system with an aspect ratio of 2:1 supported an exponential decrease in pollutants concentrations through the longitudinal profile with up to 95% of the nutrient removal achieved within the first 0.2 m (1/5 of the total length) in planted tanks. It is still difficult to say whether results obtained from this pilot scale study, set up in a greenhouse, are representative of larger scale systems. However, they provide a better understanding of the nutrient distribution and flow patterns that take place within the CWS.

The area rate constants for ortho-P were significantly higher, while for the NH_4^+-N they were lower than those reported by other authors. However, in both cases, high P and N removal rates were achieved throughout the period of investigation (95–98% in both planted and unplanted systems and 85–90% in planted systems, respectively).

The flow distribution was similar in both planted and unplanted tanks, suggesting that plant root development in the shale substrate did not reduce the reactive volume of the systems. In addition, no evidence of clogging was observed. This confirmed the good properties of shale as a substrate found in previous studies (Drizo *et al.*, 1997; Drizo, 1998).

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