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Constructed Wetlands Treating Runoff Contaminated with Nutrients

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Abstract The aim was to assess the role of *Phragmites australis* (Cav.) Trin. ex Steud. in experimental, mature, and temporarily flooded vertical flow wetland filters treating urban runoff rich in organic matter. During the experiment, ammonium chloride was added to sieved concentrated road runoff to simulate primary treated urban runoff contaminated with nitrogen. Five days at 20°C N-allylthiourea biochemical oxygen demand (BOD) and chemical oxygen demand removal efficiencies were relatively lower for planted than unplanted filters. Moreover, there was no significant difference for BOD removal for all filters under fluctuating inflow concentrations of sulfate. The nitrogen removal performances of planted filters were more efficient and stable throughout the seasons compared to those of unplanted filters. A substantial load of nitrogen (approximately 500 mg per filter) was removed by harvesting *P. australis*. Plant uptake was the main removal mechanism for nitrogen during high concentrations (10 mg/L) of ammonia-nitrogen in the urban runoff.

Keywords Constructed treatment wetland · *Phragmites australis* · Ammonia-nitrogen · Biochemical oxygen demand · Suspended solids · Macrophyte harvesting

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1 Introduction

1.1 Nutrient Removal

Despite positive reports on wetland treatment performances, nitrogen removal remains a challenge (Scholz 2006a, b). In Europe, typical removal efficiencies of ammonia-nitrogen (NH_4^+-N) in long-term, engineered wetland systems only range between 35% and 50% (Verhoeven and Meuleman 1999; Vymazal 2002). Furthermore, a recent U.S. Environmental Protection Agency (EPA) publication suggests that constructed wetlands (CW) are not successful in treating nitrogen, and earlier studies, claiming very efficient nitrogen removal by CW, could not be successfully repeated (U.S. Environmental Protection Agency (EPA) 2000).

Nutrient loads from non-point sources of pollution, i.e., diffuse pollution from urban (Birch et al. 2004; Scholz 2006a, b) and agricultural (Poe et al. 2003) runoff cause adverse impacts on mankind and the environment. Concerns over worldwide lake, river, and coastal eutrophication have triggered public pressure to reduce nitrogen inputs to aquatic ecosystems (Reinhardt et al. 2006). Advanced technologies incorporating biological nitrification and denitrification can remove nitrogen efficiently. However, these technologies are often very costly (Bezbaruah and Zhang 2003).

Various literature reviews have reported on the transformation and removal processes of nutrients in subsurface flow CW (Scholz 2006a). Findings show that macrophytes significantly affect the removal of pollutants in horizontal subsurface flow CW with long hydraulic retention times. In comparison, their role is minor for nitrogen removal by surface flow systems, which frequently have shorter retention times (Karathanasis et al. 2003). Furthermore, it has been reported that between 6% and 48% of nitrogen is retained by macrophytes planted in gravel bed subsurface flow wetlands (Kadlec et al. 2005). Typical nutrient uptake capacities of macrophytes were as follows: 30 to 150 kg P/ha a, and 200 to 2,500 kg N/ha a.

1.2 Rationale, Aim, and Objectives

The contribution of macrophytes to the removal of nitrogen for temporarily flooded vertical flow systems treating concentrated urban runoff has not been researched as intensively as for horizontal flow systems. Moreover, the role of macrophytes in CW treating urban runoff contaminated with high concentrations of nitrogen was not examined previously. Therefore, the aim of this study is to assess nitrogen removal, especially by *Phragmites australis* (Cav.) Trin. ex Steud., in experimental vertical flow wetland filters treating urban runoff artificially contaminated with ammonia. The objectives are as follows:

- To analyze the pollutant removal performances for various filter designs
- To evaluate the impact of different concentrations of sulfate on the 5 days at 20°C N-allylthiourea biochemical oxygen demand (BOD) removal processes
- To assess the impact of *P. australis* on the nitrogen removal efficiencies under high concentrations of ammonia-nitrogen

2 Experiments, Materials, and Methods

2.1 Study Site

Since 9 September 2002, six mature wetland filters treating pretreated gully pot liquor have been located outdoors at The King's Buildings campus (The University of Edinburgh, Scotland, UK), as described previously (Lee and Scholz 2006a, b; Scholz 2006a, b). These filters were used to assess the filtration performance under high concentrations of ammonia-nitrogen in a linked subsequent experiment between 1 May and 4 December 2006.

Gully pot liquor is a concentrated surface runoff, which is detained in wet gully pots until it overflows into the sewer due to incoming surface runoff from subsequent rainfall events. Gully pots were randomly selected from an area comprising the campus, near housing estates and two major roads. After mixing both the sediment and the liquid phases within each gully pot, a sample was collected by manual abstraction with a 2-L beaker.

2.2 Filter Design and Media Composition

Round drainage pipes were used to construct the experimental filters in 2002 as described by Lee and Scholz (2006a, b). All six vertical flow wetland filters, which were operated in batch flow mode, were designed with the following dimensions: height= 83 cm and diameter=10 cm. The filters were sufficiently large to avoid edge effects (Scholz 2006a, b). The outlet of each CW filter comprised a valve at the bottom of each filter. The valve was only open during the water exchange periods lasting up to 15 min a few times per week (i.e., the water was still most of the time). Different packing order arrangements of filter media and plant roots were used in the wetland filters (Table 1). *P. australis* pot plants were collected from a local grower and supplier. Four established plants with healthy rhizomes were planted densely in each filter in fall 2002, approximately 4 years before the start of this additional experiment. This was deemed sufficient time for the plants and biomass to reach maturity and relative.

No replicate filters were used in this study. A linked previous study by Lee and Scholz (2006a, b), using the same filters, showed that there were no

Table 1 Packing order of vertical flow wetland filters

Height (cm)	F1	F2	F3	F4	F5	F6
61–83	W	W	W	W	W	W
56–60	W	6	6+P	7	7+P	7+P
51–55	W	6	6+P	6	6+P	6+P
36–50	W	4	4+P	5	5+P	5+P
31–35	W	3	3	5	5	5
26–30	W	3	3	4	4	4
21–25	W	2	2	3	3	3
16–20	W	2	2	2	2	2
11–15	2	2	2	2	2	2
0–10	1	1	1	1	1	1

Filter 1 is the control

W water, *P Phragmites australis* (Cav.) Trin. ex Steud., *1* stones, *2* large gravel, *3* medium gravel, *4* small gravel, *5* Filtralite (light expanded clay product), *6* sand (0.6–1.2 mm), *7* Frogmat (barley straw)

significant differences between the different filters for most water quality variables after the systems have matured and the outflow water quality has stabilized.

The wetland filters were similar to various other engineered treatment processes. For example, filter 1 (control) was similar to a wastewater stabilization pond or gully pot (extended storage) without a significant amount of aggregates. In comparison, filters 2 and 4 were similar to gravel filters, and filters 3, 5, and 6 were typical reed bed filters (Table 1).

The reed bed filters contained gravel and *P. australis*, all of similar total biomass weight of approximately 0.5 kg during planting. After 4 years of growth, the plant biomass was assumed to be similar in the planted filters. In comparison, filters 4, 5, and 6 also contained adsorption media (Filtralite and Frogmat). Filtralite pellets, which contain 3% of calcium oxide (CaO) and have diameters between 1.5 and 2.5 mm, are associated with enhanced metal and nutrient reduction (Scholz and Xu 2002a, b). Furthermore, Frogmat (a natural product comprising raw barely straw) has a high adsorption area and is therefore likely to be linked to a high nutrient reduction potential. The use of other filter media with high adsorption capacities such as activated carbon (Scholz and Martin 1998; Scholz and Xu 2002a, b) and oxide-coated sand (Sansalone 1999) was not considered because of unpromising findings reported elsewhere (Scholz 2006a).

The aggregates within filters 5 and 6 were the same. However, filter 6 has been used to treat gully pot liquor contaminated with heavy metals (copper and nickel) since 2002, as described previously (Lee and Scholz 2006a, b). The purpose of investigating this additional filter was to assess the recovery of wetlands that were previously polluted by heavy metals over a long period, because these high concentrations of heavy metals are not particularly bio-available and are likely to accumulate within the CW filters (Scholz and Xu 2002a, b). As metals build up, metal toxicity subsequently increases (Scholz 2006a). This may change the composition and behavior of bacteria within the CW (Vymazal and Krasa 2003).

2.3 Operation

The filters were designed to operate in batch and tidal flow mode to reduce pumping and computer control costs and to encourage biodegradation by allowing air into the wetlands during periods of drainage. All filters were periodically inundated (100%; ponding depth above the sediment of approximately 22 cm as shown in Table 1) with contaminated inflow gully pot liquor and partially drained (50%) or entirely drained (0%) to encourage air penetration through aggregates as discussed in detail by Lee and Scholz (2006a, b), and previously in a similar context by Scholz and Xu (2002a, b). However, the raw gully pot liquor was sieved (pore size of 2.5 mm) to simulate preliminary and primary treated effluent. This procedure complies with common wastewater treatment practice (Scholz 2006a; Tchobanoglous et al. 2003).

On 1 May 2006, the ammonia-nitrogen concentration of the inflow (approximately 1.5 mg/L) was artificially raised by the addition of ammonium chloride (NH_4Cl) to the sieved gully pot liquor. As a consequence, the inflow ammonia-nitrogen concentration increased permanently up to approximately 10 mg/L.

Table 2 summarizes typical inflow water quality characteristics for the contaminated urban water. The empty bed volumes (i.e., sum of voids between particles and pond zone volume) varied between approximately 3.5 and 5.5 L, depending on the filter composition and filter maturity (i.e., indicated by

Table 2 Typical urban run-off inflow characteristics (n=30 samples) after addition of ammonia-nitrogen (01/05/06–04/12/06)

Variable	Unit	Min	Max	Mean	SD[a]
BOD[b]	mg/L	28.0	220.0	90.4	48.56
Chemical oxygen demand	mg/L	45	291	158	63.0
Ammonia-nitrogen	mg/L	4.7	11.5	9.8	1.37
Nitrate-nitrogen	mg/L	0.02	0.47	0.07	0.088
Nitrite-nitrogen	mg/L	0.01	0.46	0.05	0.079
Ortho-phosphate-phosphorus	mg/L	1.12	5.84	3.90	1.212
Sulfate	mg/L	8.0	79.0	15.0	16.29
Suspended solids	mg/L	38	1,640	277	350.9
Dissolved oxygen[c]	mg/L	4.6	10.9	7.8	1.74
pH	–	6.0	7.8	6.6	0.44
Redox potential	mV	70	248	178	48.8
Conductivity	µs	215	2,290	450	403.5
Total dissolved solids	mg/L	107	1,146	227	201.7
Water temperature	°C	5.8	20.7	13.2	3.70

[a] Standard deviation

[b] Five days at 20°C N-allylthiourea biochemical oxygen demand

[c] n=28 samples

accumulation of debris). All filters were operated in batch flow mode as described in detail by Lee and Scholz (2006a, b). The corresponding loads can be easily calculated with Table 2. Moreover, loads and removal rates for selected variables and filters are shown in Table 3.

2.4 Nutrient Harvesting

P. australis was harvested at the end of the growing season in fall, and plant material was taken from points at least 83 cm above the bottom of the planted filters. It follows that the stems were cut approximately 20 cm above the top of the layer of debris (Table 1). Plants in each filter were collected, dried at 80°C for 24 h, and subsequently weighed. Dried plant samples were divided into stems and leaves, digested and analyzed for total nitrogen (TN) and total phosphorus (TP) weights according to Kalra (1998). The data were used to calculate the different nutrient uptakes (Table 4).

2.5 Water Quality Analysis

At least weekly water sampling was undertaken every time when the outflow valves were opened. The BOD was determined in all water samples with the OxiTop IS 12-6 system, a manometric measurement device, supplied by the Wissenschaftlich-Technische Werkstätten (WTW), Weilheim, Germany. Nitrification was suppressed by adding 0.05 mL of 5 g/L N-allylthiourea (WTW chemical solution No. NTH600) solution per 50 mL of sample water (Scholz 2006a).

Nutrients (ammonia-nitrogen, nitrate-nitrogen, nitrite-nitrogen, and ortho-phosphate-phosphorus) were determined for all water samples by the Palintest photometer 7000, using Palintest tablet-based methods (Palintest Ltd., Tyne and Wear, UK). For the chemical oxygen demand (COD), the water samples were oxidized by digestion in a sealed reaction tube with sulfuric acid and potassium dichromate in the presence of a silver sulfate catalyst (Palintest PL450 tubes). The amount of dichromate reduced was proportional to COD. A reagent blank was prepared for each batch of tubes to compensate for the oxygen demand of the reagent itself. The color of heated water samples was indicative of the COD and was measured using the Palintest photometer 7000 (Palintest Ltd., Tyne and Wear, UK). All remaining variables (suspended solids (SS), dissolved oxygen (DO), pH, redox potential, conductivity, total dissolved solids, and temperature) were determined according to standard methods (Allen 1974; APHA 1998).

Minitab was used to undertake the statistical analyses of all data. An analysis of variance was carried out to test for the significance (p<0.05) in difference between different data sets.

Table 3 Comparison of the 5 days at 20°C N-allylthiourea biochemical oxygen demand (BOD), chemical oxygen demand (COD), total inorganic nitrogen (TIN), and ortho-phosphate-phosphorus (PO_4^{3-}) reduction rates of unplanted (filters 1, 2, and 4) and planted (filters 3, 5, and 6) filters

Filter	BOD			COD			TIN			PO_4^{3-}		
	Loading (g/m^2 day)	Removal (g/m^2 day)	Reduction rate (%)	Loading (g/m^2 day)	Removal (g/m^2 day)	Reduction rate (%)	Loading (g/m^2 day)	Removal (g/m^2 day)	Reduction rate (%)	Loading (g/m^2 day)	Removal (g/m^2 day)	Reduction rate (%)
1	12.67	6.77	53.4	22.15	8.51	38.4	1.39	0.21	15.1	0.55	0.07	12.7
2	8.06	7.04	87.3	14.10	9.85	69.9	0.88	0.40	45.4	0.35	0.14	40.0
3	8.58	7.48	87.2	15.00	9.97	66.4	0.94	0.89	94.6	0.37	0.29	78.4
4	8.52	7.58	88.9	14.90	10.42	69.9	0.93	0.47	50.5	0.37	0.24	64.8
5	8.81	7.45	84.5	15.41	9.20	59.7	0.96	0.90	93.8	0.38	0.28	73.6
6	8.29	7.70	92.8	14.50	10.35	71.4	0.91	0.89	97.8	0.36	0.30	83.3

3 Results and Discussion

3.1 Effect of *P. australis* on Water Quality

The mean reduction rates for the key water quality variables of the unplanted filters 2 and 4 were relatively higher than those of the corresponding planted filters 3 and 5 (Table 3), but not statistically significantly different from them. These findings suggest that *P. australis* does not significantly improve the removal performance of organic matter as reported elsewhere (Lim et al. 2001). However, the mature root system most likely enhanced the capacity of transporting oxygen to the substrate and providing a large surface area for microorganisms (Scholz 2006a; Scholz and Xu 2002a, b).

In contrast to previous researchers, who reported the worst seasonal performance for BOD removal during cold periods (Karathanasis et al. 2003), all filters with the exceptions of filter 1 (44%; extended storage) and filter 5 (59%) showed high mean BOD removal efficiencies (>85%) since 20 October 2006. The corresponding water temperatures (<10°C) were, however, relatively low. This suggests that microbes have the capacity to effectively decompose organic matter during cold periods.

The effluents of planted filters had a darker yellow color (visual observation) than those of unplanted filters. A possible explanation for the relatively low BOD and COD concentrations in the unplanted filters was the absence of degrading organic matter such as leaves from the vegetation in fall. The vegetation is, however, responsible for additional indirect aeration and subsequent oxidation of the organic load (Scholz 2006a).

Figure 1 indicates that with a rise of the BOD and/or COD loading rate(s), the corresponding removal rates increase as well, regardless if the filters are unplanted or planted. The BOD/COD ratio of the raw wastewater indicates if an organic load is readily biodegradable or not (Scholz 2006a). Figure 2 shows that the influent BOD/COD ratio is mostly over 0.3 and close to 0.9, which indicates that the wastewater is relatively easy biodegradable. As the BOD/COD ratio increases, so does the BOD removal rate, regardless if the corresponding treatment filter is unplanted or planted. For example, the BOD removal ratio of unplanted filter 2 was consistently over 0.8 during most of the operation time.

Table 4 Dry weight of harvested aboveground *Phragmites australis* (Cav.) Trin. ex Steud. for all planted filters and the corresponding total nitrogen (TN) and total phosphorus (TP) weights (sums and rates) for the annual harvest in fall 2006, where the plants were most mature

Filter numbers		Dry weight (g)	TN		TP	
			Sum (mg)	Rate (mg/g)	Sum (mg)	Rate (mg/g)
3	Stems	53.64	228.02	4.2510	15.29	0.285
	Leaves	19.54	304.14	15.565	20.75	1.062
	Total	73.18	532.16	–	36.04	–
5	Stems	83.10	308.05	3.7070	27.84	0.335
	Leaves	15.82	212.52	13.434	13.57	0.858
	Total	98.92	520.57	–	41.41	–
6	Stems	86.50	263.56	3.0470	16.78	0.194
	Leaves	15.52	209.86	13.522	16.94	1.092
	Total	102.02	473.42	–	33.72	–

The BOD removal rates observed in the planted and unplanted filters in the presence of sulfate (concentrations ranged between 8 and 79 mg/L; mean of 15 mg/L) did not change significantly with different concentrations of sulfate. With respect to filter 3, for example, the mean BOD removal ratio was 0.823 when sulfate concentrations exceeded 60 mg/L, while the mean BOD removal ratio was 0.841 when sulfate concentrations ranged between 8 and 19 mg/L, which was the case during most of the operation time.

This finding contrasts recent research regarding the sulfate reduction achieved with surface flow CW; e.g., if influent concentrations are above 75 mg/L sulfate, the organic matter removal decreases by 20% (Wiessner et al. 2005). This may be related to sulfide toxicity, which has been observed to affect negatively both sulfate-reducing and methanogenic bacteria in anaerobic reactors (Kalyuzhnyi and Fedorovich 1998; Kuo and Shu 2004). This observation was confirmed by similar experiments elsewhere (Caselles-Osorio and Garcia 2006), where the COD removal efficiency was approximately 85% in the presence of sulfate and around 95% in its absence.

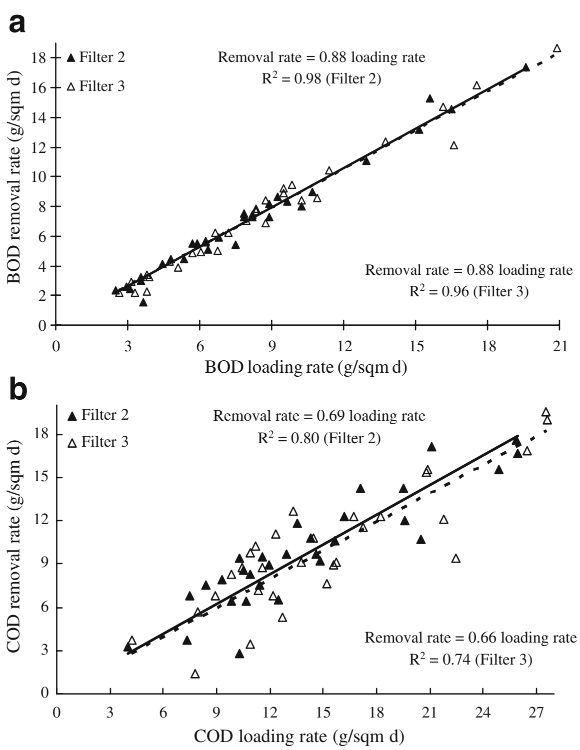

Fig. 1 Regression between the mass loadings and removal rates of (**a**) the 5 days at 20°C N-allylthiourea biochemical oxygen demand (*BOD*) and (**b**) the chemical oxygen demand (*COD*) in the unplanted filter 2 and planted filter 3. R^2, coefficient of determination

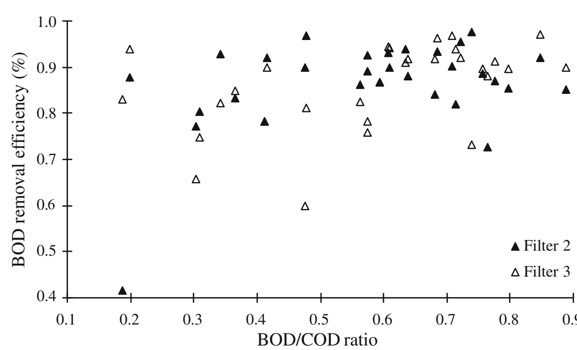

Fig. 2 Regression between the 5 days at 20°C N-allylthiourea biochemical oxygen demand (*BOD*)/chemical oxygen demand (*COD*) ratios and the BOD removal ratios in the unplanted filter 2 and planted filter 3. R^2, coefficient of determination

3.2 Effect of *P. australis* on Nutrient Removal

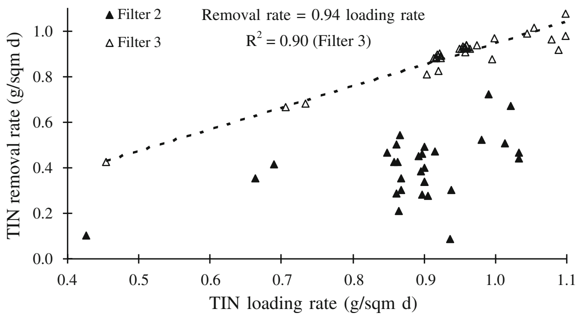

Fig. 3 Regression between the mass loadings and removal rates of total inorganic nitrogen (*TIN*) in the unplanted filter 2 and planted filter 3

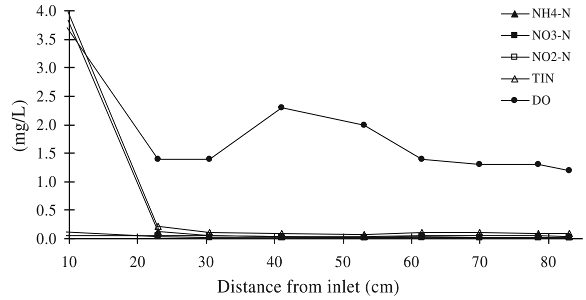

Fig. 5 The vertical concentration profiles of ammonia-nitrogen (NH_4^+*-N*), nitrate-nitrogen (NO_3^-*-N*), nitrite-nitrogen (NO_3^-*-N*), total inorganic nitrogen (*TIN*), and dissolved oxygen (*DO*) with increasing distance from the inlet of the planted filter 3

Total inorganic nitrogen reduction efficiencies for the planted filters 3, 5, and 6 were always higher than those for the corresponding unplanted filters 1, 2, and 4 (Table 3). Findings indicate that the total inorganic nitrogen removal ratio for planted filters did not seasonally fluctuate between May and December 2006.

Figure 3 shows a linear relationship between the loading and removal rates of total inorganic nitrogen in vegetated filters. In comparison to filter 2, higher and more consistent total inorganic nitrogen removal rates were observed in filter 3 due to the uptake of nitrogen by *P. australis*. Moreover, oxygenation by *P. australis* is likely to have improved the reduction of nitrogen within the wetlands. Findings show also that high nitrogen loading rates resulted in high nitrogen removal rates for planted wetlands. This can probably be explained by the enhanced stimulation of bacterial denitrification processes (Birch et al. 2004; Poe et al. 2003).

Detailed discussions on biomass and nutrient mass balances within vertical flow CW have been summarized elsewhere (Meuleman et al. 2002; Scholz 2006a). For example, early (i.e., in October rather than December) harvesting of nutrients in The Netherlands resulted in higher removal rates (Meuleman et al. 2002). Therefore, TN removal by harvesting *P. australis* was assessed on 4 October 2006. The total quantities of accumulated nitrogen in the stems and leaves of *P. australis* are shown in Table 4. The total nitrogen contents were higher within the leaves than within the stems for all planted filters. This result was similar to previous research findings (Kadlec et al. 2000), where nitrogen accumulation within reeds was

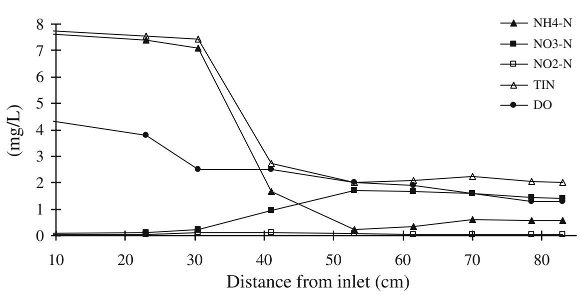

Fig. 4 The vertical concentration profiles of ammonia-nitrogen (NH_4^+*-N*), nitrate-nitrogen (NO_3^-*-N*), nitrite-nitrogen (NO_3^-*-N*), total inorganic nitrogen (*TIN*), and dissolved oxygen (*DO*) with increasing distance from the inlet of the unplanted filter 2

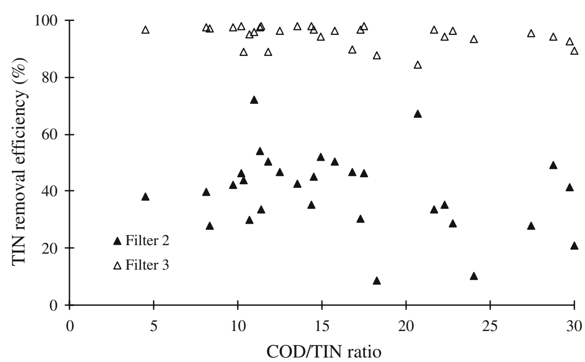

Fig. 6 Correlation between the chemical oxygen demand (*COD*)/total inorganic nitrogen (*TIN*) ratios and the TIN removal efficiencies

based on the following sequence: leaves > roots > flowers > stems. Considering the removal of nitrogen (1,045.9 mg) in filter 3 in 2006, a substantial proportion of nitrogen could be removed by regular harvesting in fall.

The major problem for nitrogen removal in CW is the availability of oxygen for nitrification and subsequent availability of a carbon source for biological denitrification (Poe et al. 2003). In general, nitrification is more efficient in free water surface CW than in subsurface flow CW (U.S. Environmental Protection Agency (EPA) 2000). Figures 4 and 5 indicate that the oxygen concentration within the wetlands slowly decreased with the distance from the inlet. However, the DO concentration at the 40 cm distance mark for filter 3 (Fig. 5) increased slightly, because *P. australis* supplied some oxygen to the rhizosphere (Bezbaruah 2002). In particular, the DO concentration was sufficient for nitrification within all filters, so that the nitrate-nitrogen concentration increased through the treatment process in unplanted filters with the exception of filter 1 (control; extended storage only). However, denitrification was insufficient for planted and unplanted filters, possibly due to relatively high DO concentrations (sometimes even higher than 0.5 mg/L in summer) within these systems (Poe et al. 2003).

Furthermore, Fig. 6 reveals that denitrification did not decrease because of limited carbon availability for neither the unplanted nor the planted filters. Compared to the unplanted filter 2, the total inorganic nitrogen removal ratio was consistently maintained over 0.8 at different C/N ratios for filter 3.

3.3 Nitrogen Removal Mechanisms Regarding Planted Filters

Constructed wetlands can be used as nutrient sinks or transformers; particularly, nitrogen can be removed through wetlands by several pathways including nitrification followed by denitrification, assimilation into biomass, mineralization of organic nitrogen, ammonia volatilization, and adsorption of ammonia onto substrate (Kao et al. 2003; Meuleman et al. 2003; Scholz 2006a). Denitrification is often the main mechanism for nitrate removal in free surface flow CW (Reilly et al. 2000).

Concerning nitrogen assimilation onto biomass within the litter zone, the amount of nitrogen immobilized by biomass was 15% of the added nitrogen during the first year of operation in a case study discussed by Silvan et al. (2003). However, wetlands should achieve nitrogen transfer equilibrium when they are operated in a stable mode after the startup period. Thus, the contribution of belowground biomass assimilation can be negligible. So the nitrogen assimilation onto biomass within the litter zone was not taken into account, because all experimental filters ran since 2002. After long-time operation, filter media will eventually be saturated by nutrients (Scholz 2006a). Concerning the work presented in this paper, the organic nitrogen attributed between 0.12% and 3.44% of the TN for storm water runoff samples. It follows that the main pathway for nitrogen removal for the planted filters was plant uptake, followed by the nitrification and denitrification processes.

4 Conclusions

Experimental vertical flow wetlands (operated in batch and tidal flow mode) were successful in treating urban runoff contaminated with high concentrations of ammonia. Five days at 20°C N-allylthiourea biochemical oxygen demand, chemical oxygen demand, total inorganic nitrogen, and ortho-phosphate-phosphorus removals were not statistically significantly different when comparing unplanted and planted wetland filters with each other. The BOD removal performances for both wetland types were not significantly different in the presence of different concentrations of sulfate. Compared to the unplanted wetlands, *P. australis* (Cav.) Trin. ex Steud. was found to contribute considerably to the nitrogen removal process (if leaves and stems would be harvested in fall) as plant uptake was the main pathway for nitrogen removal. A substantial amount of total nitrogen (between 473 and 532 mg per wetland area (10 cm diameter filters) in fall 2006) was removed by harvesting *P. australis*.

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