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Potential of constructed wetlands in treating the eutrophic water: Evidence from Taihu Lake of China

Linfeng Li [a,b,d], Yinghao Li [a], Dilip Kumar Biswas [a], Yuegang Nian [c], Gaoming Jiang [a,*]

[a] Key Laboratory of Vegetation and Environmental Change, Institute of Botany, Chinese Academy of Sciences, Beijing 100093, PR China
[b] Agricultural College of Guangdong Ocean University, Zhanjiang 524088, PR China
[c] Chinese Research Academy of Environment Sciences, Beijing 100012, PR China
[d] Graduate University of Chinese Academy of Sciences, Beijing 100049, PR China

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Abstract

Three parallel units of pilot-scale constructed wetlands (CWs), i.e., vertical subsurface flow (VSF), horizontal subsurface flow (HSF) and free water surface flow (FWS) wetland were experimented to assess their capabilities in purifying eutrophic water of Taihu Lake, China. Lake water was continuously pumped into the CWs at a hydraulic loading rate of 0.64 m d^{-1} for each treatment. One year's performance displayed that average removal rates of chemical oxygen demand (COD), ammonia nitrogen (NH_4^+–N), nitrate nitrogen (NO_3^-–N), total nitrogen (TN) and total phosphorous (TP) were 17–40%, 23–46%, 34–65%, 20–52% and 35–66%, respectively. The VSF and HSF showed statistically similar high potential for nutrients removal except NH_4^+–N, with the former being 14% higher than that of the latter. However, the FWS wetland showed the least effect compared to the VSF and HSF at the high hydraulic loading rate. Mean effluent TP concentrations in VSF (0.056 mg L^{-1}) and HSF (0.052 mg L^{-1}) nearly reached Grade III ($\leqslant$0.05 mg L^{-1} for lakes and reserviors) water quality standard of China. Wetland plants (*Typha angustifolia*) grew well in the three CWs. We noted that plant uptake and storage were both important factors responsible for nitrogen and phosphorous removal in the three CWs. However, harvesting of the above ground biomass contributed 20% N and 57% P of the total N and P removed in FWS wetland, whereas it accounted for only 5% and 7% N, and 14% and 17% P of the total N and P removed in VSF and HSF CWs, respectively. Our findings suggest that the constructed wetlands could well treat the eutrophic lake waters in Taihu. If land limiting is considered, VSF and HSF are more appropriate than FWS under higher hydraulic loading rate.

Keywords: Constructed wetlands; Nutrient removal; Pollution; Taihu Lake; *Typha angustifolia*

1. Introduction

The constructed wetlands (CWs) are considered as low-cost alternatives for treating municipal, industrial and agricultural wastewater (Seo et al., 2005). Those naturalized treatment systems also have been demonstrated to have significant potentials for both wastewater treatment and resource recovery (Hofmann, 1996). Therefore, CWs have been frequently used for nutrient removal from polluted rivers (D'Angelo and Reddy, 1994; Michal et al., 1995; Jing et al., 2001) and lakes (William, 1995; Annadotter et al., 1999; Coveney et al., 2002). Wetland filtration systems, for example, could reduce up to 30–67% total phosphorous (TP) and 30–52% total nitrogen (TN) of the hypereutrophic lake water (Coveney et al., 2002). Compared with conventional treatment systems, CWs, which are of low cost, easily operated and maintained, can be potentially applied in developing countries with serious water pollution problems. However, due to lack of awareness, these systems have not been widely used (Kivaisi, 2001). Until 2004, for example, there had not been any full-scale constructed wetland applications for eutrophic lake water treatment in China.

* Corresponding author. Tel.: +86 1062836286; fax: +86 1082590815.
E-mail address: jianggm@126.com (G. Jiang).

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As a result of fast economic growth in the past several decades, China now faces serious lake water pollution problems, with lakes throughout the country being commonly undergoing the process of eutrophication (Qin, 2002). Up to 66% and 22% of all lakes in China, are eutrophic and hypereutrophic, respectively (Jin and Hu, 2003). Taihu Lake is one of the seriously polluted lakes in China. As the most densely populated areas, Taihu Lake region is economically crucial for its fast growth. Although it accounts for only 0.4% of the total area of China and 2.9% of the nation's population, it provides more than 14% of China's gross domestic production (GDP) (Shen et al., 2000). Along with the economic process during the past three decades, Taihu Lake has been treated for multiple purposes: water supply, flood control, shipping, waste disposal, commercial fishery and sightseeing. It plays, therefore, an extremely important role in both economical and social development (Sun and Huang, 1993).

To treat the water pollutions happened in lakes like Taihu, it is practically urgent to consider new, cheap and environmentally friendly approaches to solving these serious problems. To this end, constructed wetlands might be effective in treating nutrient pollution as well as in restoration of lake ecosystems. Nevertheless, before real action should be undertaken, stimulated wetland systems should be carefully designed and run to test the capabilities in purifying eutrophically polluted waters.

For this purpose, we conducted the present study in Taihu Lake, South China. Three parallel units of pilot-scale CWs (30 m^2 each) with different water-flow formats were implemented to purify the eutrophic lake water in 2004. The primary goal of this study was to compare the nutrients removal efficiency of different CWs and evaluate the possibility of using such large full-scale constructed wetlands to purify China's most natural water-bodies with eutrophication problems. Simultaneously, plant biomass contributions to nutrient removal processes in the constructed wetlands were evaluated.

2. Methods

2.1. Study site and wetland design

Taihu Lake (31°30′N, 120°30′E), the third largest shallow freshwater lake in China, sits in the south of the Yangtze River delta. It covers a water surface area of 2428 km^2, with a mean depth 1.9 m. Taihu Lake Plain is dominated by monsoon climate, with annual mean air temperature varying from 14.9 to 16.2 °C. The annual mean precipitation and evaporation are 1000–1400 mm and 941 mm, respectively. The water temperature ranges from 0 to 38 °C, with the minimum temperature happening in January and the maximum in August (Sun and Huang, 1993).

The constructed wetlands were built on the northeast bank of Wuli Lake, which is a lagoon to the north of Taihu Lake. As the hypereutrophic water body of Taihu Lake, it has an area of 5.15 km^2 and a distance only 2 km to the City of Wuxi. The constructed wetlands were built into vertical subsurface flow (VSF), horizontal subsurface flow (HSF) and free water surface flow (FWS) wetland, with dimensions of $20\ m \times 1.5\ m \times 1.0\ m$, $20\ m \times 1.5\ m \times 1.0\ m$ and $20\ m \times 1.5\ m \times 0.8\ m$ ($L \times W \times D$), respectively. All the units were lined with polyethylene with thickness of 0.5 cm. From bottom to top, the VSF and HSF CWs were filled with gravels (size) of 25–35 mm, 16–26 mm, 6–16 mm, forming a depth of 25 cm, 25 cm and 30 cm deposition layer, respectively. The FWS CW was filled with local soil forming a depth of 40 cm. Several types of PVC pipes were used to distribute the lake water flow evenly into the CWs. Afterwards, polyethylene drainflex pipes were used to collect the treated lake water.

The CWs were planted with the same emergent species of *Typha angustifolia* Linn. In July 2004, seedlings of *T. angustifolia* from the local natural field were transplanted initially at a density of eight plants (approximately 50 cm in height) per square meter in each unit. After planting, water levels were kept constant at 10 cm below the soil surface for VSF and HSF, whilst 20 cm above the soil surface level for FWS wetland. In August, 2004 when the plants were well established, the investigation commenced.

2.2. Operation and monitoring

Water of Wuli Lake was continuously pumped into the above-mentioned CWs through a submersible pump placed at a distance of 10 m apart from the bank and at a depth of 40 cm beneath the water surface. To ensure a uniform water flow to each of the wetland units, a variable area flow meter and 24 cm gate valves were fitted to each flow distribution PVC pipes. The inflow rates were adjusted manually and checked regularly to achieve a mean hydraulic loading rate of 0.64 $m\ d^{-1}$ for each unit. Because of land limiting, hydraulic loading rate of 0.64 $m\ d^{-1}$ in this study was higher than values reported by Wu et al. (2002) and Liu et al. (2003). The operation and monitoring of the wetlands were conducted between August 2004 and July 2005.

2.3. Water sampling and chemical analysis

From August 3, 2004 to July 27, 2005, influent and effluent water of the pilot-scale CWs were sampled approximately every 2 weeks ($n = 24$) under normal conditions to evaluate their treatment performances. Water samples were analyzed for chemical oxygen demand (COD), ammonium–nitrogen (NH_4^+–N), nitrate–nitrogen (NO_3^-–N), nitrite–nitrogen (NO_2^-–N), total nitrogen (TN) and total phosphorus (TP). COD was determined by titrimetric method. Determination of NH_4^+–N, NO_3^-–N, NO_2^-–N, TN and TP were performed using a segmented flow analysis (Skalar San^{++} Automated Wet Chemistry Analyzer, the Netherlands). All the parameters mentioned above were determined according to the method as described in the Standard Method for Examination of Water and Wastewater (Standard Method for the Examination of Water and

Wastewater Editorial Board, 1993). The physico-chemical water parameters, such as water temperature, redox-potential (Eh), pH, and dissolved oxygen (DO) were measured in situ. DO was assayed using an Orion Dissolved Oxygen Probe (Model 862Aplus, USA). Water temperature and Eh were recorded with an Orion 250Aplus ORP Field Kit, and water pH with an Orion Portable pH Meter (Model 250Aplus, USA).

2.4. Plant harvesting and nutrient storage

In order to determine nutrient storage in the different parts of plant and estimate the amount of nutrients that could be removed by harvesting plants, all aboveground biomass was harvested in November 2004 and whole plant biomass was sampled in August 2005. Whole plant samples were harvested from three 0.25 m^2 sample plots of each wetland unit, and then separated into leaves and roots/rhizomes. Roots/rhizomes were washed firstly by tap water then by distilled water to remove any adhering sediments. Each sample was cut into small pieces, well-mixed, sun-dried for two days to reduce the moisture content and later oven-dried to constant weight at 80 °C for dry weight determination. The aboveground biomass per m^2 was calculated as an average of the values obtained in November 2004 and August 2005. The concentration of N in plant tissues was determined using the Kjeldahl method, while P was measured by the molybdateascorbic acid method (Institute of Soil Science, Chinese Academy of Sciences, 1978).

2.5. Statistical analysis

The treatment efficiency was calculated as the percent removal R for each parameter, which was calculated by $R = (1 - C_e/C_i) \times 100$, where C_i and C_e are the influent and effluent concentrations in mg L^{-1}. Mean influent values of three sampling sites were used to calculated removal rates for each parameter to eliminate the space difference. All statistical analysis were performed using the SPSS software package (SPSS, 2003), including analysis of variance (ANOVA), Bartlett's and Levine's test for homogeneity of variance and normality, and Duncan's multiple range test for differences between means.

3. Results

3.1. Water quality and removal performance

Concentrations of COD, NH_4^+–N, NO_3^-–N, NO_2^-–N, TN and TP were lower in the effluent water than in the influent, which differed significantly among the CWs (Table 1). However, all parameters mentioned above in the effluent water were not significantly different between VSF and HSF. The differences in removal efficiencies of COD, NH_4^+–N, NO_3^-–N, NO_2^-–N, TN and TP among the different CWs were highly significant ($p < 0.01$ for all variables). Multiple comparisons detected significantly higher COD, NO_3^-–N, NO_2^-–N, TN and TP removal rates in both VSF and HSF than FWS. But no significant difference was observed in removal rates of NH_4^+–N between FWS and HSF ($p = 0.074$). Rather, VSF showed significantly higher NH_4^+–N removal than HSF ($p < 0.05$) and FWS ($p < 0.05$). Nevertheless, no significant differences of COD, NO_3^-–N, NO_2^-–N, TN and TP removal rates were detected between VSF and HSF.

3.2. Physico-chemical variables

Changes in values of the situ measured physico-chemical variables of water temperature, dissolved oxygen (DO), redox-potential (Eh) and pH are presented in Table 1. Mean temperature in the influent and within the effluents of the CWs was not significantly different. In contrast, pH values were considerably lower in the outlet water of the CWs than in the inlet water, however, no significant differences among the three CWs were noted. Furthermore, DO and Eh in effluent were significantly higher in FWS than VSF and HSF. However, no significant difference was detected between VSF and HSF ($p > 0.05$).

3.3. Time course of removal

It was found that the nutrient removal rates fluctuated in each wetland during the 12-month investigation (Figs. 1–3). Though VSF and HSF showed similarly higher trend COD removal than FWS (Fig. 1), no statistically significant seasonal differences were found ($p > 0.05$ for all). Again, high fluctuations were observed for NH_4^+–N (Fig. 2a), NO_3^-–N (Fig. 2b) and TN (Fig. 2c) removal in the three CWs. Higher NH_4^+–N and TN removal effects appeared in autumn and summer in three CWs. However, in winter (December–February), the elimination of NH_4^+–N and TN remained relatively the lowest in all CWs. Despite apparently lower NO_3^-–N removal efficiencies in winter, no statistically significant seasonal variation was detected for each treatment ($p > 0.05$ for all) (Fig. 2b). Rather, the removal rate of TP was higher in the first several months (August–October) in the three CWs then decreased and/or fluctuated over the rest experimental period (Fig. 3).

3.4. Plant biomass production and nutrient accumulation

All the plants adapted themselves well into the CWs and did not wither until the later autumn (November) in 2004, with the average plant heights being 201.2 ± 18.3, 198.3 ± 20.2, and 211.3 ± 7.9 cm for VSF, HSF and FWS CWs, respectively. In the second growing season (July, 2005), the average heights of the *Typha* plants reached to 235.3 ± 13.1, 235.1 ± 14.2 and 246.9 ± 15.2 cm for VSF, HSF and FWS CWs, respectively. One-way ANOVA detected no statistically significant differences in the height of *Typha* among the CWs ($p = 0.284$).

Table 1
Mean concentrations ± SD and pollutants removal efficiencies for chemical oxygen demand (COD), ammonia nitrogen (NH_4^+–N), nitrate nitrogen (NO_3^-–N), nitrite nitrogen (NO_2^-–N), total nitrogen (TN), total phosphorus (TP) and physico-chemical water parameters of water temperature (T), dissolved oxygen (DO), redox-potential (Eh) and pH in the influent and the effluent waters of the free water surface flow (FWS), horizontal subsurface flow (HSF) and vertical subsurface flow (VSF) constructed wetlands

Parameters	Influent	Effluent		
		FWS	HSF	VSF
COD				
Concentration (mg L^{-1})	$7.37^a \pm 1.2$	$5.93^b \pm 1.2$	$4.23^c \pm 0.8$	$4.25^c \pm 0.7$
Removal (%)		$16.5^b \pm 13.6$	$39.6^a \pm 9.5$	$40.4^a \pm 8.7$
NH_4^+–N				
Concentration (mg L^{-1})	$1.63^a \pm 0.6$	$1.37^{ab} \pm 0.6$	$1.16^{bc} \pm 0.5$	$0.89^c \pm 0.4$
Removal (%)		$22.8^b \pm 17.7$	$32.0^b \pm 16.1$	$45.9^a \pm 26.4$
NO_3^-–N				
Concentration (mg L^{-1})	$1.41^a \pm 0.9$	$0.85^b \pm 0.6$	$0.37^c \pm 0.5$	$0.50^{bc} \pm 0.5$
Removal (%)		$34.2^b \pm 31.4$	$65.3^a \pm 38.4$	$62.9^a \pm 27.5$
NO_2^-–N				
Concentration (mg L^{-1})	$0.27^a \pm 0.1$	$0.19^a \pm 0.21$	$0.03^b \pm 0.02$	$0.05^b \pm 0.03$
Removal (%)		$33.8^b \pm 34.6$	$82.6^a \pm 25.3$	$78.9^a \pm 17.7$
TN				
Concentration (mg L^{-1})	$4.82^a \pm 0.9$	$3.97^b \pm 1.36$	$2.29^c \pm 0.86$	$2.37^c \pm 0.97$
Removal (%)		$19.8^b \pm 17.9$	$52.1^a \pm 14.6$	$51.6^a \pm 14.7$
TP				
Concentration (mg L^{-1})	$0.152^a \pm 0.03$	$0.103^b \pm 0.06$	$0.052^c \pm 0.03$	$0.056^c \pm 0.04$
Removal (%)		$35.1^b \pm 33.9$	$65.7^a \pm 19.1$	$64.3^a \pm 23.4$
T (°C)	$22.3^a \pm 9.5$	$21.8^a \pm 9.9$	$21.8^a \pm 9.9$	$21.7^a \pm 8.9$
DO (mg L^{-1})	$7.76^a \pm 2.3$	$6.26^a \pm 4.0$	$1.80^b \pm 0.8$	$1.86^b \pm 0.53$
Eh (mV)	$284.0^a \pm 9.4$	$283.7^a \pm 91.8$	$222.1^b \pm 96.4$	$221.5^b \pm 37.9$
pH	$7.88^a \pm 0.3$	$7.66^b \pm 0.51$	$7.62^b \pm 0.21$	$7.57^b \pm 0.14$

Values with different superscript letters indicate a significant difference at $p \leqslant 0.05$ according to the Duncan's multiple range tests.

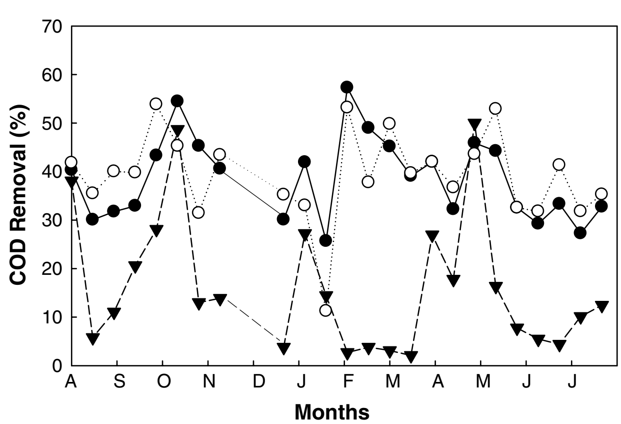

Fig. 1. Temporal course of removal rates of chemical oxygen demand (COD) in vertical subsurface flow (VSF) (closed circles), horizontal subsurface flow (HSF) (open circles) and free water surface flow (FWS) (closed triangles) constructed wetlands of Taihu Lake (2004–2005).

The differences in both above and belowground biomass among the CWs were significantly different ($p < 0.001$ for two variables) (Fig. 4). Biomass of belowground was significantly higher ($p \leqslant 0.001$) than aboveground in VSF and HSF, whereas it was exactly reversed in FWS ($p \leqslant 0.001$). In addition, the highest biomass (9.46 kg m^{-2}) were reached by HSF compared to VSF and FWS ($p \leqslant 0.001$) (Fig. 4).

The concentration of N was significantly higher ($p \leqslant 0.01$) in leaves than in roots/rhizomes for plants in all CWs (Fig. 5b). However, the concentration of P did not show statistically significant differences between leaves and roots/rhizomes ($p > 0.05$) despite high concentration of P in leaves than roots/rhizomes for plants in all CWs (Fig. 5a). One-way ANOVA detected significantly higher ($p \leqslant 0.001$) concentrations of N and P in leaves for plants in FWS than that in VSF and HSF.

4. Discussion

The concentration-based removal efficiency 17–40% for COD in our study was lower than 61–94% reported by Rousseau et al. (2004). This might result from the relatively low influent COD concentrations of our study compared with that in sewage of the literature, since most of external point pollution have been cut off before experiment. So, the low organic loads resulted in the lower concentration-based COD removal efficiencies within the CWs. Korkusuz et al. (2005) stated that it was more difficult to reduce COD concentrations below 50 mg L^{-1} after secondary treatment. And it was even more difficult for our case, since throughout the whole running period, the influent concentration of COD remained constantly below 11 mg L^{-1} (5.4–10.3 mg L^{-1}) (Fig. 1). Similar results were reported by

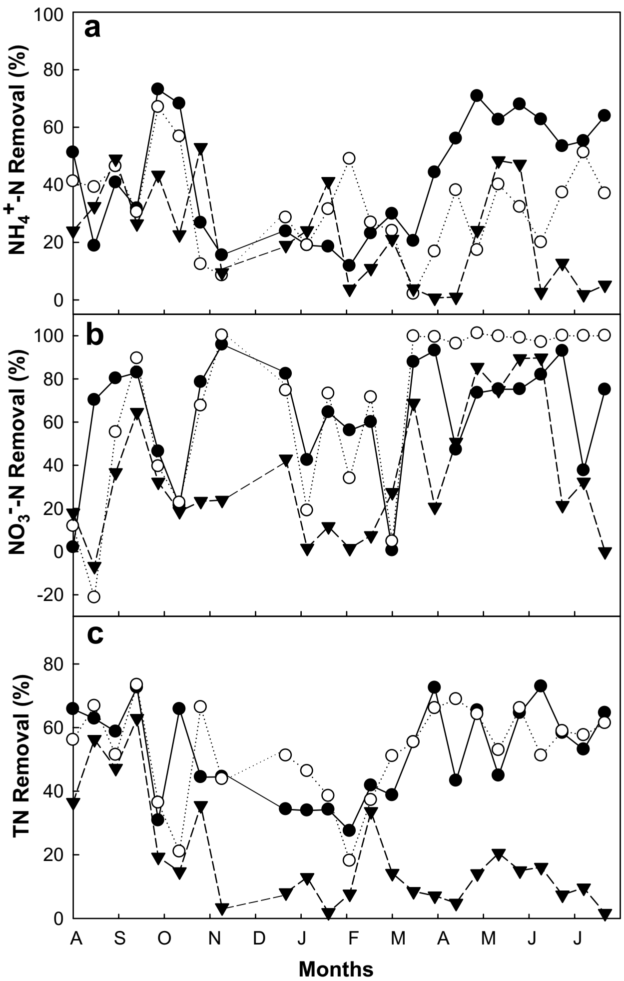

Fig. 2. Temporal course of removal rates of ammonia nitrogen (NH_4^+–N, a), nitrate nitrogen (NO_3^-–N, b) and total nitrogen (TN, c) in vertical subsurface flow (VSF) (closed circles), horizontal subsurface flow (HSF) (open circles) and free water surface flow (FWS) (closed triangles) constructed wetlands of Taihu Lake (2004–2005).

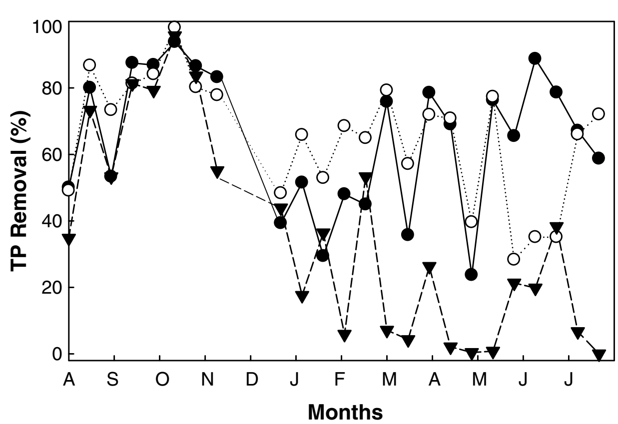

Fig. 3. Temporal course of removal rates of total phosphorus (TP) in vertical subsurface flow (VSF) (closed circles), horizontal subsurface flow (HSF) (open circles) and free water surface flow (FWS) (closed triangles) constructed wetlands of Taihu Lake (2004–2005).

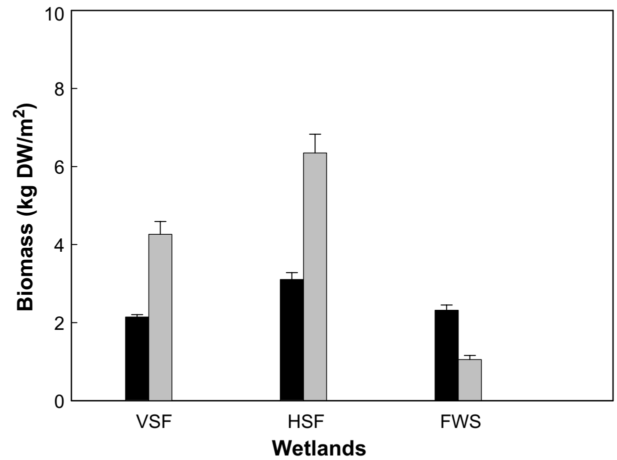

Fig. 4. Biomass productivity (mean ± SE) of above ground (dark bars) and below ground (grey bars) of *Typha* plants in vertical subsurface flow (VSF), horizontal subsurface flow (HSF) and free water surface flow (FWS) constructed wetlands.

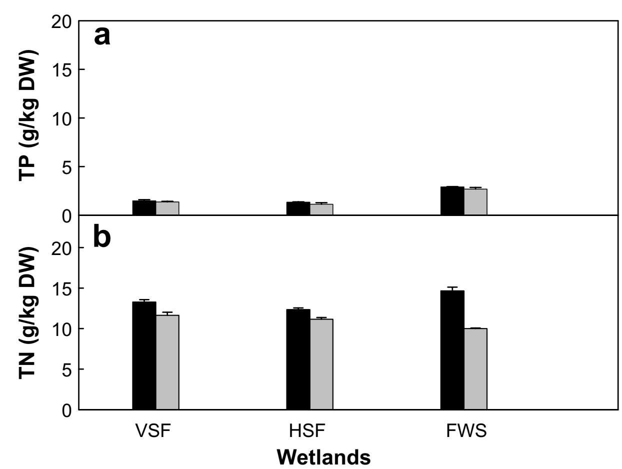

Fig. 5. Nutrient (N&P) content (mean ± SE) in leaves (dark bars) and roots/rhizomes (grey bars) of *Typha* plants in vertical subsurface flow (VSF), horizontal subsurface flow (HSF) and free water surface flow (FWS) constructed wetlands.

Liu et al. (2004), who studied the performance of three parallel subsurface horizontal flow CWs (40 m^2 each) for the purpose of surface water quality improvement. In their case, the influent concentration of COD ranged from 4.5 to 7.5 mg L^{-1} (over 10-month running), with an average COD removal of 15–40% being gained.

Nitrogen removal in CWs is accomplished primarily by physical settlement, denitrification and plant/microbial uptake (Healy et al., 2006). In our study, the mean NH_4^+–N removal in VSF was significantly higher than that in HSF and FWS. This difference was most likely a result of better oxygen transfer from the atmosphere to the wetland bed in VSF than HSF and FWS (Brix, 1997). Even through, the effects obtained in our study were lower than mean values reported by Vymazal (2006). This may be due to the relatively low NH_4^+–N hydraulic loads of our study. Even though, the mean effluent NH_4^+–N concentrations in VSF reached Grade III ($\leqslant$1.0 mg L^{-1}) and that in HSF and

FWS reached Grade IV (⩽1.5 mg L^{-1}) water quality, according to the standard values of surface water in China (SEPA, 2002).

Concentration of nitrate in the effluent of FWS was higher than that of HSF and VSF (Table 1). However, it did not necessarily mean that nitrification in FWS was more efficient than that in HSF and VSF as the denitrification rate was not clearly known. Kozub and Liehr (1999) applied nitrate nitrogen removal rate as an estimate of denitrification rate based on the influent and effluent concentrations in the field. Using their method, we calculated the concentration-based NO_3^-–N removal efficiencies, being 63%, 65% and 34% for VSF, HSF and FWS, respectively. We could, therefore, expect that stronger denitrification occurred in the VSF and HSF while weaker in FWS. Our results were not consistent with the general idea that FWS, similar to HSF, might exhibit more efficient denitrification rates than VSF due to their predominantly anaerobic environments (Vymazal, 2006). Some suggested that denitrification in a constructed wetland depends largely upon the amount of nitrate nitrogen and organic carbon available in the system as well as environment conditions such as pH, temperature, surface area for microbial attachment, and dissolved oxygen concentration (Kozub and Liehr, 1999). Here in our study, the values of influent water pH (7.57–7.66) were within the range optimum for denitrification reported by Paul and Clark (1996). The high underground biomass (particularly in VSF and HSF) (Fig. 4) implied extensive wetland plant root network for microbial attachment (Weisner et al., 1994; van Oostrom and Russell, 1994). According to Reddy and Patrick (1984), when water temperatures lower than 15 °C or greater than 30 °C can drastically reduce the growth rate of nitrifying bacteria, thus limiting the rate of denitrification. Although denitrification rates were found to be very sensitive to the presence of oxygen in pure cultures and stream sediments (Richardson and Ferguson, 1992), anaerobic conditions for denitrification could be found in CWs even with outflowing oxygen concentrations of 4 mg L^{-1} (Schulz et al., 2003). Thus the lower rate of denitrification in FWS may result from the lack of availability of organic carbon or an insufficient supply of NO_3^-–N (Spieles and Mitsch, 2000). This can also explain the nearly equal level of denitrification between VSF and HSF in our study.

TN removal efficiency differed significantly among the CWs. Both VSF and HSF showed higher removal rates than FWS. This probably due to the short residence and/or contact time happened in FWS for most important processes involved occur in the sediment whereas the wastewater flows over the sediment. Dissolved nutrients thus have to transfer by diffusion, which is a fundamentally slow process (Verhoeven and Meuleman, 1999). Average TN removal rates 52% in VSF and HSF are higher than 44.6% and 43.6% reported by Vymazal (2006) and Wu et al. (2004), respectively. However, an average TN removal of 20% in FWS wetland was lower than 41% obtained by Vymazal (2006), which might probably due to the relatively higher hydraulic loading rate we designed.

In our study, the phosphorus removal capability differed significantly among the wetlands. Both VSF and HSF wetlands showed significantly higher phosphorus removal rates than FWS wetland. This might result from the inadequate contact time between the influent and the soil of the FWS wetland, which the polluted water flows over the soil. However, all the three CWs showed efficient total phosphorus (TP) removal during the start-up period of our study (Fig. 3). This implies that gravel/soil adsorption determined initial efficient phosphrous removal (Lin et al., 2002). However, the ability of adsorption may decrease with time as sorption sites on the gravel/soil become saturated. High average TP removal rates achieved in VSF (64%) and HSF (66%) were higher than 60% and 41% reported by Vymazal (2006) for vertical subsurface flow and horizontal subsurface flow CWs, respectively. Moreover, the results obtained in our study are also higher than those experimented in other water bodies (Liu et al., 2004; Wu et al., 2001) of China. This might be attributed to the high hydraulic loading rate we used than CWs in literature. Rather, the mean effluent TP concentrations (0.056 and 0.052 mg L^{-1} in VSF and HSF, respectively) nearly reached Grade III (⩽0.05 mg L^{-1} for lakes and reserviors) water quality, according to the standard values of surface water in China (SEPA, 2002).

It has once been reported that removal of nitrogen and phosphorus through plant harvesting is negligible since the removed amount was only a small fraction (Brix, 1994). In the present study, if it is assumed that plant N and P uptake occurs only from the water column, mass balance calculations showed plant uptake and storage contribution of 13.8%, 18.6% and 26.2% N and 40.5%, 47.3% and 80.9% P of the total N and P removed by VSF, HSF and FWS CWs, respectively. Obviously, active uptake and incorporation into plant tissue was a major factor responsible for the observed N and P removal in FWS wetland. Other processes such as nitrification/denitrification and adsorption of soluble phosphorus to roots and gravel were more important for N and P removal in VSF and HSF wetlands (Kyambadde et al., 2004). According to the data depicted in Figs. 4 and 5, harvesting of the above ground biomass of *Typha* plants would remove 0.85, 1.16 and 1.02 kg N and 0.09, 0.12 and 0.20 kg P for VSF, HSF and FWS CWs, respectively during our study period, which account for 5.1%, 6.6% and 20.1% of N and 14.2%, 17.4% and 57.0% of P that removed by VSF, HSF and FWS CWs, respectively. These results indicated that harvesting of emergent plants could play a significant removal route for nutrient removal treatment wetlands, especially for the FWS wetland with *Typha* plants of our study.

5. Conclusions

It is clear from our research that the pilot-scale CWs can be effective treatment in treating eutrophic water from a natural water body such as lake. CWs with vertical subsurface flow and horizontal subsurface flow had higher potential for nutrients removal compared to the free water surface flow wetland of our study. Aerobic and anaerobic environments within the wetland cells as well as the differences of water flow formats might decide the removal efficiency of constructed wetlands. However, it is impossible for single-stage CW to achieve high removal of nitrogen since it cannot provide both aerobic and anaerobic conditions at the same time. Therefore, further research is still needed for enhancing the nitrogen removal by combination of the CWs in order to exploit the specific advantages of the individual system. Meanwhile, plant uptake and nutrient storage was also responsible for the removal of N and P in three CWs. Therefore, harvesting of the above ground biomass of emergent plants could play a significant removal route for nutrient removal treatment wetlands, especially for the free water surface flow wetland of our study.

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