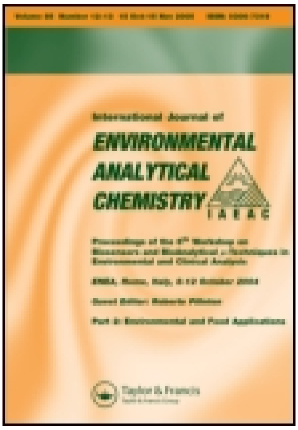

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Nutrient removal in vertical subsurface flow constructed wetlands treating eutrophic river water

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Nutrient removal in vertical subsurface flow constructed wetlands treating eutrophic river water

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Four planted (*Typha latifolia* L.) pilot-scale vertical subsurface flow constructed wetlands were constructed to purify the eutrophic water of the Jinhe River in Tianjin (China) and to determine the feasibility of constructing a full-scale system in the future. The effects of intermittent artificial aeration and the use of polyhedron hollow polypropylene balls (PHPB) as part of the wetland substrate on the nutrient removal potential were also evaluated. During the entire running period, supplementary aeration enhanced the chemical oxygen demand, ammonia-nitrogen, total nitrogen, soluble reactive phosphorus and total phosphorus first order mean removal constants by 0.28 m/d, 3.05 m/d, 0.92 m/d, 0.74 m/d and 0.60 m/d, respectively, but reduced the nitrate-nitrogen removal constant by 1.72 m/d in contrast to non-aerated wetlands. A significantly positive contribution of PHPB to nutrient removal was obtained. The combination of artificial aeration and PHPB resulted in the augmentation of the first order mean removal constants by 0.29 m/d, 3.12 m/d, 1.15 m/d, 0.65 m/d and 0.54 m/d for chemical oxygen demand, ammonia-nitrogen, total nitrogen, soluble reactive phosphorus and total phosphorus, respectively. Findings from a brief cost-benefit analysis suggest that both artificial aeration and the presence of PHPB would result in enhanced nutrient removal that is cost efficient for future projects, particularly if electricity costs are low.

Keywords: constructed treatment wetland; polyhedron hollow polypropylene balls; *Typha latifolia*; intermittent artificial aeration; chemical oxygen demand; nitrogen; phosphorus; cost-benefit analysis

1. Introduction

1.1 *Background and motivation*

Constructed wetlands are an emerging ecotechnology with optimised hydraulic control and management of vegetation [1,2]. Compared with conventional activated sludge and biofilm processes, low cost, easily operated and maintained constructed wetlands can be applied in developing countries with serious water pollution problems [3,4]. Wetlands are

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most frequently used in river systems to manage flow [5,6]. Only few studies report on the potential of constructed wetlands in treating eutrophic river water [7,8]. In China, constructed wetlands succeeded in treating eutrophic lake waters of Taihu [4]. However, there are currently only few full-scale constructed wetland applications for eutrophic river water treatment in China.

Constructed wetlands could be an effective ecotechnology to remove excess nutrients. Previous studies have demonstrated that the removal of biochemical oxygen demand (BOD), chemical oxygen demand (COD) and suspended solids in constructed wetlands can be satisfactory [9–11], although the removal of nitrogen and phosphorus tends to be variable and is frequently low [12,13]. Therefore, constructed wetlands should be designed with some innovation to reach much higher nutrient removal rates. The role of oxygen availability in nutrient removal has frequently been discussed [10,14]. Low oxygen content results in low aerobic organic matter decomposition [12]. Moreover, nitrification may be the main limiting process for nitrogen removal in constructed wetlands, if the oxygen availability is low [15].

Phosphorus removal is also indirectly affected by oxygen availability. Under aerobic conditions, oxidisation of Fe^{2+} to Fe^{3+} (Fe, iron) may enhance the chemical precipitation of phosphorus [16]. Furthermore, between 10 and 12% of phosphorus removal could be obtained by microbial assimilation with good aeration [17]. In planted constructed wetlands, the oxygen availability is enhanced by the presence of macrophytes through diffusion of oxygen in the sediment via the aerenchyma to the rhizomes [18]. However, plant contribution to oxygen supply is still debated [11,19,20].

In the past decade, some design advances have been proposed and applied to promote oxygen availability in constructed wetlands. For example, fluctuations of wetland water level, tidal flow or reciprocation systems are frequently used to enhance the oxygen availability [10,21,22]. An alternative solution is the injection of compressed air into the bed matrix, which greatly increases the nutrient removal efficiency of constructed wetlands in cold climates [14,23]. Artificial aeration requires energy input at additional costs, but in some instances and in most developing countries, it may still be profitable. For eutrophic river water treatment in China, artificial aeration is commonly used as an inexpensive option to increase the oxygen content within the water body to prevent odour development [24].

Other ecotechnological research areas are, for example, focusing on the assessment of the potential of novel material as constructed wetland substrate. Aggregates with large surface areas and high void spaces are prone to be rapidly colonised with biofilms. For example, polyhedron hollow polypropylene balls (PHPB) are useful in improving the nutrient removal rates in aerated biofilters [25,26]. Similar plastic material is also used as solid substrate and biofilm carrier in recirculated aquaculture systems for enhanced nutrient removal [27,28]. However, there are only a few reports on the application of plastic biofilm carriers including PHPB in constructed wetlands [29]. A potential drawback could be that PHPB are likely to increase the overall capital costs. However, the extent of cost increase would depend on the amount of PHPB used.

1.2 *Rationale, aim and objectives*

Before the start of most industrial-scale field projects, it is important to assess novel ecotechnologies in laboratories and pilot-scale trials. A sound experimental set-up is therefore required to statistically compare aerated and non-aerated constructed wetlands with and without PHPB. Eutrophic river water rather than synthetic wastewater should be used for pilot-scale experiments to simulate 'real' operational conditions. As constructed wetlands require a long period to reach maturity, any pilot-scale work can only deliver initial results to support decision-making.

The project aim was to assess different experimental systems to help decision-makers in selecting the best design option for subsequent industrial-scale applications. In light of the above rationale, the specific objectives were:

- to assess the potential of vertical subsurface flow (VSSF) wetlands to treat eutrophic Jinhe River water;
- to examine the main and interactive effect of intermittent artificial aeration and PHPB application on nutrient removal within VSSF constructed wetlands;
- to evaluate the contribution of plant biomass uptake to nutrient removal; and
- to conduct a preliminary cost-benefit analysis to determine the economic feasibility of introducing artificial aeration and PHPB in a future full-scale application.

2. Experimental

2.1 *Design of wetland systems*

Four pilot-scale VSSF constructed wetland systems A, B, C and D were set up in the yard of the Tianjin Hydraulic Scientific Institute, Tianjin, China. Each VSSF wetland system consisted of a combined down-flow wetland and an up-flow wetland, which were both made of polyethylene columns (diameter, 0.5 m; height, 1.3 m; surface area, 0.196 m^2). Three different aggregates were used as substrate: coarse predominantly granite-based gravel (15.74 ± 2.97 mm in diameter; 48% porosity), shale dominated by quartz and feldspar (10.78 ± 1.47 mm in diameter; 46% porosity) and PHPB (25.00 mm in diameter; 81% porosity).

The packing order of the constructed wetlands A and D (without PHPB) were the same. Each down-flow wetland unit was filled with 0.3 m of gravel representing the bottom layer, and 0.6 m of shale as the main filter layer, followed by a gravel layer of 0.1 m thickness at the top to reduce the risk of clogging. The up-flow wetland was used for further secondary purification after the treatment of the influent within the down-flow wetland. The only difference in substrate packing between the down-flow and up-flow wetlands was the shale layer. The depth of the shale layer in the up-flow wetlands was 0.5 m. Thus, the water level of the up-flow wetland was 0.1 m lower than the one of the down-flow unit. This design allowed for the wastewater to flow naturally by gravity. Compared to wetlands without PHPB, regardless of the flow direction, a shale layer of 0.2 m thickness located above the bottom gravel layer was replaced by PHPB in the constructed wetlands B and C to examine the effect of PHPB on nutrient removal.

Artificial aeration of constructed wetlands A and B was performed to assess the effect of oxygen availability on nutrient removal. A perforated horizontal 0.3 m diameter circular tube was installed at a distance of 0.05 m from the bottom of both the down-flow and up-flow wetlands. Wetlands C and D functioned as the non-aerated wetlands with no air diffuser present. All of the VSSF constructed wetlands were planted with cattail (*Typha latifolia* L.) at a density of eight rhizome cuttings per wetland (i.e. 16 per wetland system) on 1 May 2006. After one month, cattail was well established in each wetland with a mean individual height of approximately 0.4 m. New shoots were also detected. In August, mean plant heights achieved maximum values between 2.38 and 2.50 m.

2.2 *Inflow water and wetland operation*

The Jinhe River, which was chosen as the source of the eutrophic influent, flows through downtown of Tianjin City (China). The approximate length of the landscaped river is 18.5 km. Relatively low heavy metal and toxic organic compound pollution has been recorded. However, the ammonia-nitrogen (NH_4^+-N), total nitrogen (TN) and total phosphorus (TP) concentrations were in the ranges between 1.15 and 1.85 mg L^{-1}, 4.11 and 6.18 mg L^{-1}, and 0.04 and 0.22 mg L^{-1}, respectively [30,31]. These previously measured (2004 and 2006) nutrient data are guide values but indicate that the Jinhe River is eutrophic [9].

River water was directly pumped into the storage wells of the experimental rig, and then continuously discharged onto the wetland systems as influent. The inflow rates were adjusted manually and checked regularly to achieve a relatively high mean hydraulic loading rate of 0.8 m/d for each constructed wetland. The corresponding influent flow rate was 0.16 L min^{-1}. High hydraulic loading rates are preferable in China for eutrophic river or lake water treatment [4,32].

Concerning the aerated constructed wetlands A and B, compressed air was slowly and continuously introduced via a perforated pipe into the wetland substrate for 8 h between 8:30 and 16:30 at a corresponding ratio of air to water of 5 : 1. The purpose was to achieve full oxygen saturation of the water as far as practically possible. After then, aeration was stopped for 16 h until the next aeration cycle was started.

2.3 *Water sampling and analysis*

From June to November 2006, influent and effluent of the pilot-scale constructed wetlands were sampled once per week ($n = 24$) under normal conditions to evaluate their treatment performances. All samples were analysed on the same day for the following parameters: COD, NH_4^+-N, nitrate-nitrogen (NO_3^--N), soluble reactive phosphorus (SRP), TP, dissolved oxygen (DO), water temperature (T) and pH. Water quality parameters including COD, NH_4^+-N, NO_3^--N, SRP and TP were determined according to Standard Methods [33], if not stated otherwise. A YSI 52 dissolved oxygen meter and a HANNA portable pH meter were used for DO, T and pH analysis, respectively.

2.4 *Plant harvesting and analysis*

All aboveground cattail biomass was harvested in November 2006 to estimate the contribution of plant harvesting to overall nutrient removal. After dividing the biomass into stems and leaves, they were subsequently oven-dried for approximately 48 h at 80°C [34]. Dry weight of the harvested aboveground biomass was expressed in weight (g) per stems or leaves. Sub-samples of dried stems and leaves were powdered, wet digested and analysed for TN and TP content according to a method provided previously [35]. The results were expressed in weight (mg) of nutrient per weight (g) of dry stems or leaves.

2.5 *Statistical analysis and modelling*

All statistical tests were performed using the Statistical Package for the Social Sciences (SPSS). Significances were defined as $p < 0.05$, if not stated otherwise. One-way analyses of variance (ANOVA) and the Tukey's honestly significant difference multiple range tests [34] were carried out to assess the differences between means of nutrient removal efficiency and effluent water quality variables in different constructed wetlands. For all ANOVA, the tested variables were normally distributed.

The removal rate constant k obtained from a first order model (Equation (1)) can be used to further test the contributions of wetland plants, substrate and novel operations including introduction of artificial aeration and the presence of PHPB on the nutrient removal efficiency in the experimental treatment wetlands. The background concentration C^* is often depleted [16], which simplifies Equation (1), leading to Equation (2).

$$C_e = (C_i - C^*)\exp\left(-\frac{kHRT}{h}\right) + C^* \tag{1}$$

where C_e and C_i are concentrations of nutrients ($\mathrm{mg\,L^{-1}}$) in the effluent and influent, respectively, C^* is the background concentration ($\mathrm{mg\,L^{-1}}$), k is the removal rate constant (m/d), HRT is the hydraulic residence time (d) and h is the effective depth of the wetland.

$$C_e = C_i \exp\left(-\frac{kHRT}{h}\right) \tag{2}$$

where C_e and C_i are concentrations of nutrients ($\mathrm{mg\,L^{-1}}$) in the effluent and influent, respectively, k is the removal rate constant (m/d), HRT is the hydraulic residence time (d) and h is the effective depth of the wetland.

3. Results

3.1 *Water quality*

During the entire monitoring period, the majority of nitrogen (78%) in the influent occurred as NH_4^+-N and 77% of phosphorus as SRP (Table 1). Suspended solids ($<10\,\mathrm{mg\,L^{-1}}$) and BOD ($<15\,\mathrm{mg\,L^{-1}}$) concentrations were very low, and therefore not measured routinely. Effluent concentrations of COD, NH_4^+-N, NO_3^--N, TN, SRP and TP for wetlands A, B, C and D are summarised in Table 1. The concentrations in the effluent were lower than the corresponding ones in the influent. Effluent COD and NH_4^+-N concentrations were significantly lower in the aerated wetlands B than in the non-aerated wetlands D. However, effluent NO_3^--N concentrations were significantly higher in the aerated wetlands A than in the non-aerated wetlands D. For all wetlands, there were no significant differences in effluent SRP, TN and TP concentrations (Table 1).

Except for the above-mentioned nutrient variables, changes in values of the online measured parameters T, DO and pH were also presented in Table 1. The mean temperature recordings for the influent and effluent were not significantly different. In contrast, effluent pH values in wetlands with PHPB were considerably lower than in the influent. However, significantly higher effluent pH values in the aerated constructed wetlands compared to the influent were noted. Furthermore, DO concentrations in the effluent were significantly higher in the aerated wetlands than in the influent and effluent

Table 1. Mean concentrations ± SD and pollutant removal efficiencies for chemical oxygen demand (COD), ammonia-nitrogen (NH_4^+-N), nitrate-nitrogen (NO_3^--N), total nitrogen (TN), soluble reactive phosphorus (SRP) and total phosphorus (TP) and other variables including water temperature (T), dissolved oxygen (DO) and pH in the influent and effluent waters of experimental aerated wetlands A (without PHPB) and B (with PHPB) and non-aerated wetlands C (with PHPB) and D (without PHPB).

		Effluent			
Variables	Influent	Wetlands A	Wetlands B	Wetlands C	Wetlands D
COD					
Concentrations ($mg\,L^{-1}$)	106.02 ± 13.70^{c}	$69.45 \pm 24.02^{a,b}$	63.07 ± 23.00^{a}	$69.23 \pm 23.08^{a,b}$	77.75 ± 24.03^{b}
Removal (%)		38 ± 10.2^{b}	43 ± 12.4^{b}	$37 \pm 12.0^{a,b}$	29 ± 10.6^{a}
NH_4^+-N					
Concentrations ($mg\,L^{-1}$)	5.73 ± 3.96^{c}	0.71 ± 0.84^{a}	0.66 ± 0.81^{a}	1.89 ± 1.38^{b}	2.34 ± 1.63^{b}
Removal (%)		89 ± 5.7^{c}	89 ± 6.9^{c}	67 ± 8.7^{b}	56 ± 14.9^{a}
NO_3^--N					
Concentrations ($mg\,L^{-1}$)	1.19 ± 0.23^{c}	0.76 ± 0.23^{b}	$0.63 \pm 0.20^{a,b}$	$0.44 \pm 0.11^{a,b}$	0.35 ± 0.10^{a}
Removal (%)		41 ± 26.7^{a}	50 ± 28.4^{a}	65 ± 9.9^{b}	72 ± 12.1^{b}
TN					
Concentrations ($mg\,L^{-1}$)	7.34 ± 3.61^{b}	1.98 ± 1.26^{a}	1.65 ± 1.15^{a}	2.43 ± 1.41^{a}	2.63 ± 1.38^{a}
Removal (%)		73 ± 9.0^{b}	78 ± 9.0^{c}	67 ± 8.7^{a}	63 ± 8.1^{a}
SRP					
Concentrations ($mg\,L^{-1}$)	0.40 ± 0.25^{b}	0.09 ± 0.05^{a}	0.09 ± 0.04^{a}	0.14 ± 0.10^{a}	0.17 ± 0.12^{a}
Removal (%)		74 ± 7.7^{b}	75 ± 10.9^{b}	65 ± 11.1^{a}	60 ± 14.6^{a}
TP					
Concentrations ($mg\,L^{-1}$)	0.52 ± 0.28^{b}	0.15 ± 0.07^{a}	0.15 ± 0.07^{a}	0.20 ± 0.12^{a}	0.22 ± 0.14^{a}
Removal (%)		68 ± 10.7^{b}	69 ± 9.9^{b}	60 ± 11.5^{a}	57 ± 14.9^{a}
T (°C)	24.56 ± 3.69	23.76 ± 3.65	23.70 ± 3.78	23.76 ± 3.65	23.70 ± 3.78
DO ($mg\,L^{-1}$)	3.42 ± 1.69^{b}	4.50 ± 1.79^{c}	4.53 ± 1.84^{c}	2.11 ± 0.55^{a}	1.85 ± 0.60^{a}
pH (–)	$7.73 \pm 0.39^{a,b}$	$7.91 \pm 0.34^{a,b}$	8.00 ± 0.45^{b}	$7.64 \pm 0.38^{a,b}$	7.57 ± 0.38^{a}

Note: Values with a different superscript letter (i.e. $^{a,\ b}$ and ^{c}) indicate significant difference at $p \leq 0.05$ based on Turkey's HSD.

of the non-aerated wetlands. However, no significant differences in effluent DO concentrations were detected between wetlands with and without PHPB (Table 1).

3.2 *Nutrient removal performance*

Significantly higher COD, NH_4^+-N, TN, SRP and TP removal efficiencies were recorded for the aerated compared to the non-aerated wetlands (Table 1). However, the NO_3^--N removal efficiency was significantly lower in the aerated wetlands than in the non-aerated wetlands. Wetlands C containing PHPB showed significantly higher COD and NH_4^+-N removal efficiencies in contrast to wetlands D without PHPB ($p < 0.05$). Without artificial aeration, however, no significant difference in NO_3^--N, TN, SRP and TP removal efficiency was observed between wetlands with and without PHPB (Table 1).

The first-order model (Equations (1) and (2)) can estimate the actual k value, since the nutrient concentrations of the influent and effluent are known (in addition to the HRT). As shown in Table 2, the mean removal rate constant k (m/d) values for COD, NH_4^+-N, NO_3^--N, TN, SRP and TP removal in wetlands A, B, C and D were identified. According to the first-order model, COD, NH_4^+-N, TN, SRP and TP removals were higher in the aerated wetlands A and B compared to the corresponding non-aerated wetlands D and C. Irrespectively of the presence or absence of aeration, COD, NH_4^+-N, TN, SRP and TP removal was higher in wetlands with PHPB than in wetlands without PHPB (Table 2). For NO_3^--N, however, the removal constants k were lower in the aerated wetlands than in the non-aerated wetlands. Moreover, among all the tested wetlands, the lowest NO_3^--N removal was noted for the aerated wetlands without PHPB (Table 2).

A linear regression analysis for wetlands D (representative example) was performed to test the relationships between each nutrient variable and other water quality parameters such as DO and T. The coefficients of the corresponding correlation matrix for all variables are shown in Table 3. The effluent COD concentrations were significantly and positively correlated with the effluent NH_4^+-N, SRP and TP concentrations ($p < 0.05$). Effluent NH_4^+-N concentrations were significantly and positively correlated with the effluent COD, TN, SRP and TP concentrations ($p < 0.05$). The effluent NO_3^--N

Table 2. Mean removal rate constant k (m/d) values for chemical oxygen demand (COD), ammonia-nitrogen (NH_4^+-N), nitrate-nitrogen (NO_3^--N), total nitrogen (TN), soluble reactive phosphorus (SRP) and total phosphorus (TP) removal in experimental aerated wetlands A (without PHPB) and B (with PHPB) and non-aerated wetlands C (with PHPB) and D (without PHPB).

	Wetlands			
Variable	A ($k_a + k_p + k_s$)	B ($k_a + k_b + k_p + k_s$)	C ($k_b + k_p + k_s$)	D ($k_P + k_s$)
COD	1.01 ± 0.44	1.02 ± 0.39	1.02 ± 0.47	0.73 ± 0.36
NH_4^+-N	4.75 ± 0.88	4.82 ± 1.54	2.37 ± 0.68	1.70 ± 0.77
NO_3^--N	0.96 ± 0.81	1.26 ± 1.03	2.38 ± 0.78	2.68 ± 1.13
TN	2.87 ± 0.76	3.10 ± 0.96	2.43 ± 0.73	1.95 ± 0.43
SRP	2.69 ± 0.55	2.60 ± 0.97	2.36 ± 0.76	1.95 ± 0.88
TP	2.26 ± 0.71	2.20 ± 0.86	2.02 ± 0.55	1.66 ± 0.67

Note: k_a, k_p, k_b and k_s represent contributions of aeration, plant, biofilm carrier (PHPB) and traditional substrate (shale) to the mean removal constant values, respectively.

concentrations were positively correlated with the DO concentrations and the effluent SRP. The TP concentrations were both significantly and positively correlated with effluent COD, NH_4^+-N and TN concentrations.

3.3 *Plant biomass production and nutrient removal*

Aboveground biomass (stems and leafs) harvesting contributed to TN and TP removal (Figure 1a,b). However, the contribution of <10% for total nitrogen

Table 3. Correlation matrix for chemical oxygen demand (COD), ammonia nitrogen (NH_4^+-N), nitrite nitrogen (NO_3^--N), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), dissolved oxygen (DO) and water temperature (T) for the representative wetlands D. The corresponding *p* values are shown in parentheses.

	COD	NH_4^+-N	NO_3^--N	TN	SRP	TP	DO	T
COD	1.000	(0.047)	(0.096)	(0.177)	(0.011)	(0.013)	(0.357)	(0.346)
NH_4^+-N	0.737	1.000	(0.454)	(0.007)	(0.002)	(0.001)	(0.187)	(0.076)
NO_3^--N	−0.616	−0.061	1.000	(0.245)	(0.251)	(0.264)	(0.060)	(0.333)
TN	0.465	0.904	0.354	1.000	(0.042)	(0.039)	(0.057)	(0.053)
SRP	0.876	0.951	−0.346	0.752	1.000	(<0.001)	(0.316)	(0.102)
TP	0.864	0.956	−0.326	0.762	0.997	1.000	(0.304)	(0.107)
DO	−0.193	0.447	0.703	0.710	0.251	0.268	1.000	(0.021)
T	0.209	0.662	0.227	0.721	0.604	0.594	0.828	1.000

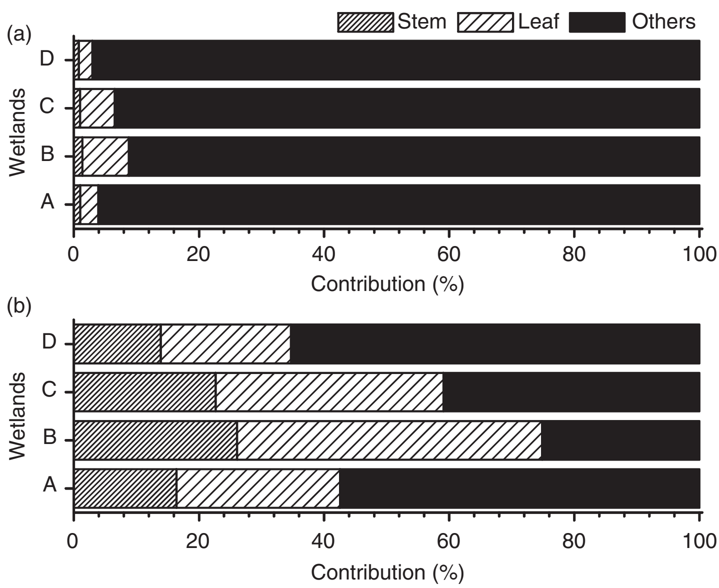

Figure 1. Contribution of the aboveground stems, leaves and others (e.g. substrate and microbes) to the (a) total nitrogen; and (b) total phosphorus removal in wetlands A (aerated without polyhedron hollow polypropylene balls (PHPB)), B (aerated with PHPB), C (non-aerated with PHPB) and D (non-aerated without PHPB) in November 2006.

removal was insignificant. In comparison, aboveground biomass harvesting was between 35 and 75% of the total phosphorus removal. Moreover, leaf harvesting removed more nitrogen and phosphorus than stem harvesting for all experimental wetlands (Figure 1a,b).

Nitrogen and phosphorus removal by aboveground biomass harvesting was much higher in wetlands with PHPB than in wetlands without PHPB. In addition, aerated wetlands showed higher nitrogen and phosphorus uptake and storage by aboveground biomass in contrast to non-aerated wetlands. The highest nutrient removal occurred in wetlands subjected to the interactive effect between intermittent artificial aeration and PHPB (Figure 1a,b).

3.4 *Expenditures*

The capital expenditure for items such as construction, substrates (gravel, shale and PHPB), plants, pumps, pipes, air condenser and diffuser per wetland unit are listed in Table 4. Considering a total investment over a period of 20 years (i.e. estimated treatment plant lifetime) and a sustained treatment capacity of $0.16\,L\,min^{-1}$, the costs to treat eutrophic Jinhe River water are approximately 0.48, 0.50, 0.46 and 0.45 Ren Min Bi (RMB)/m^3 (RMB 1 = \$0.147 = £0.074) for wetland systems A, B, C, and D, respectively.

Concerning wetlands D, artificial aeration and the presence of PHPB resulted in total cost increases of 8% and 3%, respectively. For sub-surface constructed wetlands, operational and maintenance costs, defined predominantly as electricity and labour wages, usually accounted for approximately 10% of the total capital costs in most developed countries [36]. However, this figure might be as low as 5% for China, where energy and labour costs are still relatively low. It follows that the total costs (conservative estimate based on 10% for electricity and labour) are approximately 0.53, 0.54, 0.51 and 0.49 RMB/m^3 for wetland systems A, B, C, and D, respectively.

4. Discussion

The mean COD removal efficiency (< 50%) in this study was lower than typically reported values between 80 and 99% [9,10,37]. Low influent COD concentrations ($106.02 \pm 13.7\,mg\,L^{-1}$, Table 1) may limit the COD removal capacity, as COD

Table 4. Expenditure for the construction of different pilot-scale experimental wetlands.

	Cost for tested constructed wetlands (RMB)			
Item	A	B	C	D
PHPB	–	50	50	–
Gravel	39	27	27	39
Shale	64	48	48	64
Plant	16	16	16	16
Pump, pipe and other facilities	150	150	150	150
Wetlands apparatus fee	480	480	480	480
Air condenser and diffuser	60	60	–	–
Total	809	831	771	749

Note: RMB: Ren Min Bi (currency of the People's Republic of China).

concentrations below $50\,mg\,L^{-1}$ are difficult to reduce any further [38]. Furthermore, the hydraulic loading rate applied in this study is considerably higher than those rates applied in similar systems, and the contact time and potential for biological degradation of organic matters may not be at an optimal level [11,16,39]. Increased oxygen availability could improve organic matter mineralisation and aerobic biodegradation [40]. Our findings confirmed that aerated wetlands performed better in COD removal compared to non-aerated wetlands [14,23].

There are two reasons that wetlands with PHPB outperformed corresponding systems without PHPB. Firstly, due to the high porosity of PHPB (81%) compared to shale (46%), more organic compounds settled out and were retained in the wetland filter media for a long time. This allowed for improved hydrolysis of organic compounds and for rapid biodegradation [41]. Secondly, the presence of PHPB allowed for the accumulation of large numbers of attached bacteria colonisation onto the substrate surface. This greatly improved biodegradation of COD [1,29]. However, a further discussion on bacteria colonisation is beyond the scope of this paper and would be speculation because micro-organisms were not determined in this study.

Nitrogen removal efficiency differed significantly among the constructed wetlands (Table 1). Despite the high hydraulic loading rate in this study, considerable amount of NH_4^+-N was removed and the corresponding removal efficiency was reasonably comparable to the removal obtained for other wetlands [10,23]. Significantly high NH_4^+-N removal efficiency occurred in aerated wetlands confirming the positive effect of aeration on nitrifying bacteria [14]. Artificial aeration allowed sufficient NH_4^+-N-substrate contact and enhanced the transfer of oxygen by drawing air from the atmosphere into the wetland bed media, which subsequently resulted in significantly higher effluent DO concentrations in aerated wetlands (Table 1). NH_4^+-N removal performed significantly better in wetlands with PHPB than in corresponding wetlands without PHPB which can mean one of two things. One of them is that using PHPB as wetland substrate favoured biofilm attachment and thus enhanced bacteria nitrification [25]. The second is that microbial assimilation removal of NH_4^+-N would be encouraged with the presence of PHPB.

There was more nitrate in the effluents of the aerated wetlands than the non-aerated wetlands (Table 1). This could be due to the fact that injection of compressed air into the wetland substrate increased oxygen availability but simultaneously decreased the anaerobic conditions, which are necessary for the denitrification processes [3]. Furthermore, pH affected NO_3^--N removal [4,13], however, the average values for measured pH of our study were within the range optimal (between 6.6 and 8.3) for denitrification [42]. Like supplemental artificial aeration, using PBHB as filter media also failed to improve NO_3^--N removal (Tables 1 and 2). This could be the case considering that microbes prefer the autotrophic uptake of NH_4^+-N over a corresponding uptake of NO_3^--N [43].

The TN removal efficiency was between 63 and 67% for non-aerated wetlands, which are higher than 52% obtained in constructed wetlands treating eutrophic lake water in China [4]. Complete nitrification followed by denitrification was the most important approach for total nitrogen removal [13,23]. In this study, effluent TN concentrations in the tested wetlands were low, and the net accumulation in the concentration of nitrate was rarely observed (Table 1). Therefore, nitrification-denitrification performed not completely but well in TN removal. Furthermore, considering that the majority of TN occurred as NH_4^+-N, microbial assimilation of NH_4^+-N may also contributed greatly to the current TN removal as reported elsewhere [10,43]. In addition, PHPB lead to improved

TN removal (Table 2), as the processes that facilitate TN removal in wetlands are sedimentation, nitrification-denitrification and uptake by plants and microbes [13,18,28,43]. It is reasonable to believe that these processes may be greatly enhanced by the presence of PHPB.

In relation to phosphorus removal, high average TP removal efficiencies achieved in our study were comparable to 60% and 64% reported previously [4,13] for VSSF constructed wetlands, respectively. SRP and TP removal performed significantly better in aerated wetlands than in non-aerated wetlands (Table 2). Findings showing that artificial aeration can reduce phosphorus loading at a level significantly lower than non-aerated wetlands was also confirmed elsewhere [23]. Furthermore, low oxygen concentrations within constructed wetlands cause usually relatively low TP removal efficiencies [29]. As a matter of fact, aerobic conditions favoured the chemical precipitation of phosphorus [11,16]. Alternatively, adsorption removal of phosphorus would also be encouraged in aerated wetlands with a much better degree of internal mixing than conventional non-aerated wetlands [23].

Under intermittent artificial aeration conditions, wetlands with PHPB performed best in both SRP and TP removal (Table 1). Our findings verified that presence of bioballs including PHPB favoured the microbial process responsible for phosphorus removal [26].

Wetland plants are known to take up nutrients but the amount varies widely within and between constructed wetlands. In the present study, aboveground biomass nitrogen and phosphorus removal was between 21.5 and 79.9 $\mathrm{g\,N\,m^{-2}}$ and between 14.8 and 41.6 $\mathrm{g\,P\,m^{-2}}$ (Figure 1a,b), respectively. These ranges are comparable with those values (20 to 30 $\mathrm{g\,N\,m^{-2}}$ and 3 to 8 $\mathrm{g\,P\,m^{-2}}$) obtained from traditionally designed vertical-flow constructed wetlands treating eutrophic Taihu Lake (China) water [4]. Furthermore, the findings by the authors of this paper compared well with previously reported values of 0.6–88.0 $\mathrm{g\,N\,m^{-2}}$ [44–46] and 0.1–45.0 $\mathrm{g\,P\,m^{-2}}$ [1,44,46].

If the uptake and storage of nitrogen and phosphorus by plants would have only occurred from the water column during the running period, a mass balance based on the nitrogen and phosphorus loading rate would have shown that aboveground biomass harvesting contributed to 4, 9, 7 and 3% N, and 43, 75, 59 and 35% P to the total nitrogen and total phosphorus removal in the tested wetlands A, B, C and D, respectively. Obviously, uptake and incorporation into plant tissues was a major factor responsible for the observed total nitrogen and phosphorus removal in VSSF wetlands treating eutrophic Jinhe River water. The results obtained in our study were comparable to those reported for other eutrophic lake or pond water: e.g. plant removal contributed 5–26% and 41–81% to total nitrogen and total phosphorus removal [4,43]. In light of the above calculations, nutrient uptake into aboveground biomass could be greatly improved by artificial aeration and PHPB, and the improvement of nutrient uptake contributed to the majority increase of total nitrogen and phosphorus removal.

The approximate treatment costs for wastewater in China are 1 RMB/m^3 in large cities [47]. In comparison, the treatment costs for the wetland systems A, B, C and D were by approximately 47, 46, 49 and 51% lower, although artificial aeration and PHPB were introduced. Nevertheless, due to the relatively high construction costs of small wetland units, the purchase of only small quantities of PHPB and polyethylene for the wetland columns, and the lack of electrical appliance optimisation, the total treatment costs were between 113 and 135% higher than those for the Chinese Longdao River constructed wetland, which is, however, very large in comparison; i.e. $>$90,000 times bigger in area than the experimental wetlands studied in this paper [47].

Artificial aeration and the presence of PHPB, however, were only associated with less then 10% of the total construction costs (Table 4). Furthermore, the land occupied by traditional constructed wetlands is much greater than the land required for the novel compact wetland systems containing polyethylene columns. The current costs will be reduced with an increase in wetland size and user demand. Moreover, the provided cost-benefit analysis should be seen as preliminary, considering that the assessed wetland system are relatively small.

5. Conclusions and further research

Both intermittent artificial aeration and using PHPB as part of substrate enhanced the ability of nutrient removal in vertical subsurface flow wetlands treating eutrophic river water. Findings indicate that plant uptake and storage played an important role in eutrophic river water treatment. Moreover, aboveground biomass nutrient uptake was significantly improved by artificial aeration as well as the presence of PHPB, and this improvement accounted for the majority of the total enhancement in nitrogen and phosphorus removal.

Although a preliminary cost-benefit analysis has shown that the overall costs compare very well with traditional wastewater treatment plants for large cities, it is necessary to undertake a more detailed evaluation of investment costs for real scale systems. Due to the scale effect and the short running period, the tested wetlands did not reach their full potential in nutrient removal.

Further research should target the assessment of the microbial development, biofilm attachment and root growth in the proposed wetlands. Furthermore, different operating strategies including hydraulic loading rates should be evaluated, particularly to improve the chemical oxygen demand removal. More information regarding the application of full-scale wetlands to purify eutrophic river water would also be desirable.

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