Article

Nitrogen Removal in a Horizontal Subsurface Flow Constructed Wetland Estimated Using the First-Order Kinetic Model

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Abstract: We monitored the water quality and hydrological conditions of a horizontal subsurface constructed wetland (HSSF-CW) in Beijing, China, for 2 years. We simulated the area rate constant and the temperature coefficient with the first-order kinetic model. We examined the relationships between the nitrogen (N) removal rate, N load, seasonal variations in the N removal rate, and environmental factors, such as the area rate constant, temperature, and dissolved oxygen (DO). The effluent ammonia (NH_4^+-N) and nitrate (NO_3^--N) concentrations were significantly lower than the influent concentrations ($p<0.01$, n=38). The NO_3^--N load was significantly correlated with the removal rate ($R^2=0.9566$, $p<0.01$), but the NH_4^+-N load was not correlated with the removal rate ($R^2=0.0187$, $p>0.01$). The area rate constants of NO_3^--N and NH_4^+-N at 20 °C were 27.01±26.49 and 16.63±10.58 m·yr^{-1}, respectively. The temperature coefficients for NO_3^--N and NH_4^+-N were estimated at 1.0042 and 0.9604, respectively. The area rate constants for NO_3^--N and NH_4^+-N were not correlated with temperature ($p>0.01$). The NO_3^--N area rate constant was correlated with the corresponding load ($R^2=0.9625$, $p<0.01$). The NH_4^+-N area rate was correlated with DO ($R^2=0.6922$, $p<0.01$), suggesting that the factors that influenced the N removal rate in this wetland met Liebig's law of the minimum.

Keywords: horizontal subsurface flow wetland; N removal; first-order kinetics; model

Introduction

The rapid economic development witnessed in recent years in China has led to vast increases in the production of human sewage and other waste effluents, most of which are rich in nitrogen (N) compounds. Nitrogen not only affects the water quality, but also contributes to water eutrophication [1], especially in lakes and rivers [2]. Therefore, N removal from wastewater has become a topic of growing concern, both in China and worldwide [3-4]. Many processes influence N removal from wetlands, which play an increasingly important role in removing N from aquatic systems, and can achieve total nitrogen removals of between 40% and 55% [5]. The two most common forms of N in wetlands are organic and inorganic. Organic N is made up of a variety of compounds, including amino acids, urea and uric acid, and purines and pyrimidines. Inorganic forms of N in wetlands are dominated by ammonia (NH_4^+-N), nitrite (NO_2^--N), nitrate (NO_3^--N), nitrous oxide (N_2O), dissolved elemental N, or dinitrogen gas (N_2) [6].

Constructed wetlands (CWs) are rapidly becoming a viable method for wastewater treatment worldwide [7], because of their similarity to natural wetlands and the fact that they cost less to construct, implement, and maintain than other types of treatments [8-9]. Constructed wetlands can be used to prevent water eutrophication [10]. For example, it was reported that, over a period of seven

consecutive years, CWs were able to maintain a higher removal rate of N than traditional N recovery processes [11]. In HSSF-CWs, N removal is achieved by a combination of physical, chemical, and biological processes, which are affected by a range of factors, such as pH, dissolved oxygen (DO), and temperature [12]. Mathematical models can be used to simulate N removal and the relationships between various parameters and the N removal rates in constructed wetlands to reflect the removal mechanisms in wetlands; this information can then be used to provide decision support and the scientific basis for improving ecosystem service functions of wetlands [13]. Previous studies suggested that the first-order kinetics model can describe N removal in wetland ecosystems [14-15]. However, recent studies that have used the first-order kinetics model for N removal have mainly focused on the application of the area rate constant and the volume rate constant in HSSF-CWs, and, to date, applications of the area rate constant to HSSF-CWs and assessments of the influence of water pollution loads, temperature, and DO on the area rate constant have been largely neglected [16].

Based on the wetland water balance principle, the aim of this study was to examine the removal and seasonal variations of two N forms, namely NO_3^--N and NH_4^+-N, in a HSSF-CW. The first-order kinetic model is a commonly used hydrological model, and has been frequently used to predict, for example, variation in water chemistry, sudden changes in influent organic loads, NO_3^--N removal, and pharmaceuticals removal [17-20]. We also used the first-order kinetic model to estimate the area rate constant and temperature coefficient of N removal in this study. Further, to improve the accuracy and consistency of the model parameters, we also examined the relationships between the area rate constant and factors that influenced it, such as temperature, pollution load and DO.

Materials and methods

Study site description

Our study was focused on a HSSF-CW in the Beijing Wildlife Rescue and Rehabilitation Center, Beijing, China (Latitude: 40°6'14.40" N, Longitude: 116°42'35.71" E) (Fig. 1). The annual average temperature at the study site is 11.5 °C. The average temperatures in January and July are 4.9 and 25.7 °C, respectively, and the minimum temperature in July is 19.1 °C from 2012 to 2013. The average annual rainfall is 625 mm from 2012 to 2013. The wetland consists of three treatment cells (Table 1), and its purpose is to improve the water quality of the artificial lake in the site. Gravel is the principal media; there are two layers, one 20 cm deep that comprises gravels with a diameter of between 15 and 30 mm and the other 50 cm deep comprised of finer gravels with diameters of between 5 and 15 mm from bottom to top. The average water depths in the three treatment cells are 0.3, 0.1, and 0.1 m. Plants in the upper media, planted at a density of two plants per centiare, show vigorous growth. The plant species are mainly *tectorum Lythrum*, *salicaria Iris*, and *congesta Eleocharis*, with densities of 30, 50, and 60 stem/m², respectively. Birds, such as *Ardea alba*, *Anas platyrhynchos,* and Anatidae, are frequently observed at the lake. Oxygenation in the wetland was enhanced by adding oxygen via ventilation pipes (10 cm in diameter and 0.50 m long).

The water quality and hydrology monitoring

The HSSF-CW was operated with continuous flow from April 2012 to November 2013. The water quality was monitored from April to November in 2012 and 2013. Between December 2012 and March 2013 the low temperatures and frozen water prevented sampling. The sampling points were located in the inflow, middle, and outflow of each cell. Flow-weighted composite water samples were collected from the inlet and outlet of each wetland cell using water samplers. Samples were collected from each point in triplicate. The water samples were collected in 500-mL plastic bottles and stored at 4 °C until analysis. The flow was recorded by a flow meter (SonTek, YSI, USA), while meteorological variables, such as rainfall and evaporation, were recorded hourly by a portable weather station (WeatherHawk, CampSci, USA). The concentrations of NH_4^+-N and NO_3^--N were determined using the standard methods for the examination of water and wastewater [21] with a

water quality analyzer (SmartChem 200 Chemical Analyzer, West Co, USA). A portable water quality probe (EXO, YSI, USA) was used to measure DO, temperature, and other *in situ* variables.

Data analysis and modeling

Equation 1 represents the wetland water dynamic equilibrium model of the HSSF-CW:

$$dV/dt = Q_{in} - Q_{out} + Q_c + (P - ET - I)A,$$

(1)

where Q_{in} and Q_{ou} are the daily volumetric water inflow and outflow rates (m³·d⁻¹), respectively, Q_c is the catchment runoff rate (m³·d⁻¹), P is the daily precipitation rate (mm·d⁻¹), ET is the daily evapotranspiration rate (mm·d⁻¹), I is the daily infiltration rate (mm·d⁻¹), dV/dt is the net change in volume (m³·d⁻¹), and A is the total area of the wetland (m²).

The first-order kinetic model of N removal applied to the HSSF-CW

The areal removal rate constant and the temperature coefficient were calculated from the monitoring data and the following first-order model [14]:

$$\ln\left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) = -\frac{K_{(t)}}{q},$$

(2)

where C_{in} and C_{out} are the influent and effluent concentrations, respectively, in mg·L⁻¹, C^* is the background concentration (mg·L⁻¹, 0 mg/L), q is the hydraulic loading rate (m·d⁻¹), t is the temperature, and $K_{(t)}$ is the area rate constant of a first-order removal rate on temperature (m·yr⁻¹). Equation 3 was used to calculate q (m yr⁻¹):

$$q = \frac{Q}{A},$$

(3)

where Q is total water inflow rate (m³·d⁻¹).

The effect of temperature on the areal removal rate constant was modelled using the following equation:

$$K_{(t)} = K_{(20)} \theta^{(t-20)},$$

(4)

$$\ln(K_{(t)}) = \ln\theta(t-20) + \ln(K_{(20)}),$$

(5)

where $K_{(20)}$ is the area rate constant of a first-order removal rate for a temperature of 20 °C (m·yr⁻¹), θ is the temperature coefficient. The relationship between $\ln(K_{(t)})$ and $(t-20)$ is linear [22]. The resulting slope and intercept are $\ln\theta$ and $\ln(K_{(20)})$, respectively.

The selected variables in this study included concentrations of NH₄⁺-N and NO₃⁻-N, DO, and temperature. The means of actual measured values are presented. Data fitting was carried out with Origin 8.5 and statistical analyses were performed using SPSS version 18.0. Statistically significant differences were determined at the $\alpha = 0.01$ significance level.

Results

Water balance

The daily volumetric water inflow and outflow rates of the HSSF-CW were 33.59±19.07 and 33.27±18.99 $m^3 \cdot d^{-1}$, respectively (Table 2). There were no significant differences between the inflow and outflow ($p>0.01$, n=42), which indicates that rainfall had little influence on the HSSF-CW. The main water supply was via injection. The large standard deviations in the daily volumetric water inflow and outflow rates indicate that there were considerable changes in the water volume in the HSSF-CW each month.

Water injection to the HSSF-CW began in April. As time progressed, the inflow gradually increased. Drainage began in October and the flow decreased with drainage, so the annual flow showed an initial increase and then decreased (Fig. 2). The maximum rainfall in the experimental period occurred in July 2012, when 337.2 mm was recorded; the minimum rainfall (0.5 mm) occurred in November 2013. The outflow was higher than the inflow in July because of the more intensive rainfall in this month. Also, the higher summer temperatures meant that the rates of water surface evaporation and plant transpiration were higher in summer than in the other seasons, which would also have had some impact on the water quantity in the system. Over the long-term, leakages through the impervious layer caused by aging would lead to a decline in the quantity of water in the HSSF-CW. In October and November, the water quantity changed slightly because of lower rainfall, and also resulted in reduced evaporation and infiltration capacity.

Influent and effluent nitrogen concentrations

The average annual NO_3^--N concentration was 3.40±3.44 $mg \cdot L^{-1}$, while the annual average NH_4^+-N concentration was 0.36±0.35 $mg \cdot L^{-1}$ (Table 3). Over the monitoring period, the NO_3^--N concentrations were higher than the NH_4^+-N concentrations. The standard deviations of the concentrations of the two N species were very large, which indicates that the concentrations of the two N species changed considerably in different months. Two mechanisms mainly controlled the migration and transformation of NO_3^--N and NH_4^+-N in constructed wetlands, namely nitrification and denitrification [23]. When compared to the influent concentrations, we found that the concentrations of NO_3^--N and NH_4^+-N in the effluent had decreased significantly ($p<0.01$, n=38), and were, on average, 1.22±0.92 and 0.27±0.27 $mg \cdot L^{-1}$, respectively. The lower concentrations indicate that N was removed from the HSSF-CW in the 2 years of monitoring. Further, the concentrations of these two N species met Class III of the Surface Water Standards in China [24].

There were seasonal variations in the N concentrations of the inflow and outflow of the HSSF-CW (Table 4). The inflow concentrations of NH_4^+-N (0.60±0.33$mg \cdot L^{-1}$) and NO_3^--N (4.52±3.94 $mg \cdot L^{-1}$) were highest in spring and summer respectively, while the NO_3^--N (0.21±0.20 $mg \cdot L^{-1}$) inflow concentration was lowest in spring.

Nitrogen loading and removal rate

There is a generally a good relationship between the inflow and outflow concentrations and loadings of TN and NH_4^+-N. Also, N removal efficiency in constructed wetlands generally falls within a certain range, regardless of whether the N load is large or small[25]. In this study, we found that the NO_3^--N pollution load and removal rate were strongly correlated ($R^2=0.9566$, $p<0.01$) (Fig. 3), but that the NH_4^+-N pollution load and removal rate were not correlated ($R^2=0.0187$, $p>0.01$) (Fig. 4).

During the study period, the NO_3^--N removal rate was higher than that of NH_4^+-N, and was vulnerable to seasonal changes. The highest NO_3^--N and NH_4^+-N removal rates of 44.7% and 37.4% occurred in summer and autumn, respectively, and are consistent with the higher NO_3^--N loads in summer and autumn. The NH_4^+-N removal rate was lowest in summer (19.1%) because of the higher

rainfall in summer. An increase in both the hydraulic loading rates and hydraulic retention times resulted in a decrease in the NH_4^+-N removal rate (Table 5).

Estimation of the area rate constants and the temperature coefficients

Using equation (1) and the measured values of C_{in} and C_{out}, we calculated area rate constants (K) for NO_3^--N and NH_4^+-N of 27.22±26.37 and 16.41±11.98 m·yr^{-1} (Table 6) respectively. The corrected values of K for a temperature of 20 °C, obtained from equation (4), were 27.01±26.49 and 16.63±10.58 m·yr^{-1} for NO_3^--N and NH_4^+-N, respectively. When corrected to 20 °C, the $K_{(20)}$ values of both NO_3^--N and NH_4^+-N were slightly lower, indicating that the influence of temperature on N removal was minimal. The $K_{(20)}$ of NO_3^--N and NH_4^+-N were lower and slightly higher, respectively, than those reported by Kadlec and Wallace [14], but were slightly higher than the values obtained by Dzakpasu et al. [22]. The temperature coefficient (θ) values estimated for NO_3^--N and NH_4^+-N were 1.0042 and 0.9604, respectively, and were lower than those reported by Kadlec and Wallace [6], but were very similar to those reported by Dzakpasu et al. [22].

The temperature of the inflow to the HSSF-CW ranged from 5 to 30 °C. Curves can be fitted using equation (4), and are shown in Fig. 5 and Fig. 6. The figures suggest that, for both NO_3^--N and NH_4^+-N ($p > 0.01$), the area rate constant and temperature were not correlated. At the same time, lower θ values demonstrate that temperature had little or no impact on either the area rate constant or the physical removal process. This contrasts with the conclusion of Huang et al.[26], who reported that the NH_4^+-N and total nitrogen removal rates were significantly influenced by temperature. Moreover, the findings suggest that temperature had a role in promoting the growth of wetland plants. Therefore, appropriate external thermal insulation measures need to be adopted to cover the surface of the wetland, to ensure improved growth of hardy plants all year round.

The influence of N loads and DO on the area rate constant

There was a strong linear correlation ($R^2 = 0.9625$, $p < 0.01$) between the area rate constants and loads of NO_3^--N (Fig. 7), which suggests that physical processes were important in NO_3^--N removal [22]. The area rate constants and loads of NH_4^+-N were not correlated ($R^2 = 0.0369$, $p > 0.01$) (Fig. 8), possibly because the NH_4^+-N load over the entire study period were lower. This also indicates that the area rate constant can accurately reflect the removal ability of different pollutants in wetland systems that are in receipt of high pollutant loads [27].

As shown in Fig. 9 and Fig. 10, the inflow DO concentrations had a minimal influence on the NO_3^--N area rate constants, and that the inflow DO concentration and NH_4^+-N area rate constants were positively correlated ($R^2 = 0.6922$, $p < 0.01$). The lower NH_4^+-N removal rate may be the result of the lower DO concentrations; these lower DO concentrations facilitate anaerobic conditions in wetlands, increase denitrification, and decrease nitrification. Besides, the low DO concentrations may be related to the lack of temperature effects. The lower DO levels would cause a decrease in biodegradation, which suggests that a significant portion of the N removal is driven by physical processes that are not influenced by temperature.

Discussion

In this study, we examined the wetland hydrology of the HSSF-CW over a period of 2 years using the wetland water balance principal. The results showed that water injection was the main source of water to the HSSF-CW. The flows into and out of the wetland varied on a monthly basis. In this study, we combined information about the wetland hydrology and N removal, and the results suggest that, when the rainfall is higher, the hydraulic retention time is lower, which would then influence the N removal rate. The water balance model adopted in this study only included hydrological data, and did not consider factors such as soil, topography, and land use [28]; this is a major limitation of the study.

There was obvious removal of both NO_3^--N and NH_4^+-N in this HSSF-CW. The removal rates however varied on a seasonal basis, with higher NH_4^+-N inflow concentrations in spring, because the swimming birds that live in the open water connected to the HSSF-CW were more active in April and May than in other months. The inflow NO_3^--N concentrations were higher in summer, reflecting the weaker denitrification caused by the abundant rainfall (Fig. 2) and shorter hydraulic retention time in summer. The NO_3^--N loads were strongly correlated with the removal rate (R^2=0.9566, p<0.01), but the NH_4^+-N load and removal rate (R^2=0.0187, p>0.01) were not correlated. This may reflect the higher and lower inflow loads of NO_3^--N and NH_4^+-N, respectively. Under certain pollutant loads, increased pollutant concentrations are conducive to biofilm formation on the substrate surface, which can not only provide good conditions for microbial growth but can also adsorb large quantities of organic matter, which contributes to pollutant removal [29].

We examined the relationship between the area rate constant and temperature, pollutant loads, and DO for both NO_3^--N and NH_4^+-N, and also estimated the temperature coefficient θ. The area rate constant and the NO_3^--N load were strongly correlated (R^2=0.9625, p<0.01), but there was no correlation between temperature and DO. The area removal rate constant for NH_4^+-N was correlated with DO (R^2=0.6922, p<0.01), but was not correlated with either the temperature or the NH_4^+-N load. The lower θ value estimated for the two N species illustrates that temperature had little or no impact on nitrification and denitrification. Nonetheless, previous studies have indicated that the rates of all biological processes are temperature-dependent, especially those of nitrification and denitrification [30]. In constructed wetlands, N removal occurs at the interface between the substrate and water, and denitrification is influenced by diffusivity, which is temperature-controlled. Microbial activity associated with nitrification and denitrification will decrease dramatically when the temperature is either below 15 °C or above 30 °C [31]. The water temperature was between 5 °C and 30 °C in this study, and the water samples had temperatures of less than 15 °C. As long as the temperature is within appropriate limits, temperature does not limit N removal.

While many factors influence the N removal efficiency in wetlands, the main controls are nitrification and denitrification [4]. Nitrification needs aerobic conditions and denitrification occurs more easily in anaerobic conditions. Both processes are affected by the DO concentration, meaning that N removal in constructed wetlands is strongly influenced by the DO concentration. There are three main sources of oxygen in constructed wetlands, including release from plants, atmospheric reaeration, and oxygen transported by water, of which release from plants and atmospheric reaeration dominate [32]. Plants can also partly block the oxygen transfer process from water to the substrate [33], so that the low oxygen concentration in the substrate becomes the main limit on NH_4^+-N removal. In this study, the concentrations of DO in the HSSF-CW varied from 0.5 to 4.5 mg·L^{-1}, and the DO% was as high as 87%. These DO concentrations are much lower than the water-saturated dissolved oxygen concentration [34], indicating that the DO concentrations in this HSSF-CW were sufficiently low to satisfy the conditions for denitrification. The DO concentration did not limit NO_3^--N removal in this HSSF-CW. The NO_3^--N removal rate in spring was low because the organic carbon source that provided energy for denitrification was degraded before NO_3^--N denitrification, so the lower organic matter concentration resulted in a lower removal rate. Further, degradation of organic matter would also consume a lot of oxygen, which helps explain the low DO concentrations. The NH_4^+-N area rate constant and DO were correlated (R^2=0.6922, p<0.01), suggesting that the low DO concentration became the limiting factor for NH_4^+-N removal. Nitrification of NH_4^+-N is slower than degradation of organics [35], and the conditions for nitrobacterium are more stringent than those for fungus that consume organic matter. The oxygen was first used to remove organic matter when the concentrations of organics were high; this was followed by nitrification until the organic concentrations dropped to a low level. Studies have shown that nitrification will only happen when the biochemical oxygen demand after 5 days (BOD$_5$) falls to below 100 mg·L^{-1} [36]. The controls on the N removal rate in this wetland met Liebig's law of the minimum, therefore, the main way to improve the pollutant removal in constructed wetlands is to regulate and control the dissolved oxygen content [37].

The flow regime and N migration and transformation processes in constructed wetlands involve

many complex physical, chemical, and biological processes. The N removal models are more complicated than those for other conventional wastewater treatments [38]. Also, the background concentrations of pollutants in the same type of wetland will vary [39]. In this study, we considered only three controls on the area rate constants and did not consider the influence of background pollutant concentrations. These other factors will be explored in future studies.

Conclusions

The main water source of this HSSF-CW in the Beijing Wildlife Rescue and Rehabilitation Center was system injection and the main N forms were NH_4^+-N and NO_3^--N. Over a period of 2 years, the effluent concentration of both N forms were significantly lower than the influent concentrations ($p<0.01$, n=38). The NO_3^--N load was significantly correlated with the removal rate ($R^2=0.9566$, $p<0.01$) and the NO_3^--N removal rate varied seasonally. The area rate constant for NO_3^--N was correlated with the pollution load ($p<0.01$), but was not correlated with either temperature or the DO concentration ($p>0.01$). The NH_4^+-N area rate constant was correlated with the DO concentration ($p<0.01$) but was not correlated with either temperature or the pollutant load ($p>0.01$). The controls on the N removal rate in this wetland meet Liebig's law of the minimum.

Acknowledgements

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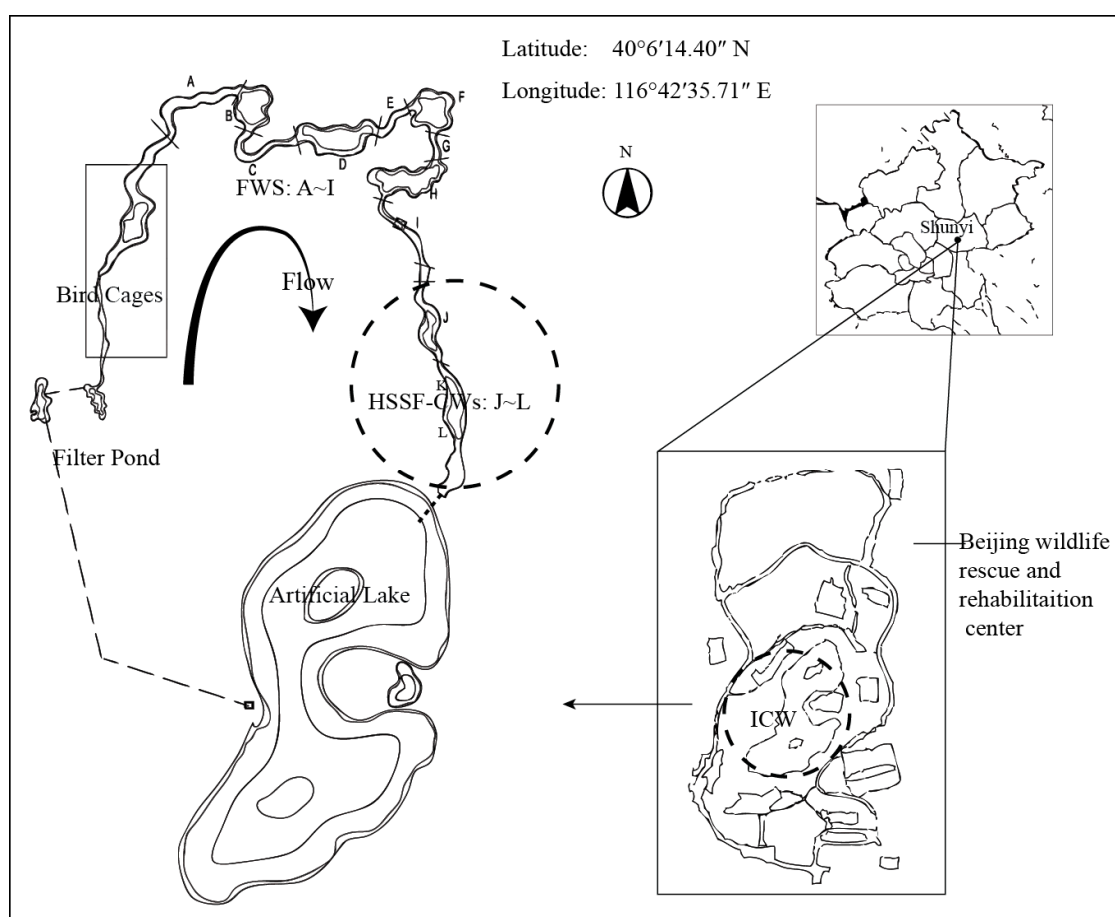
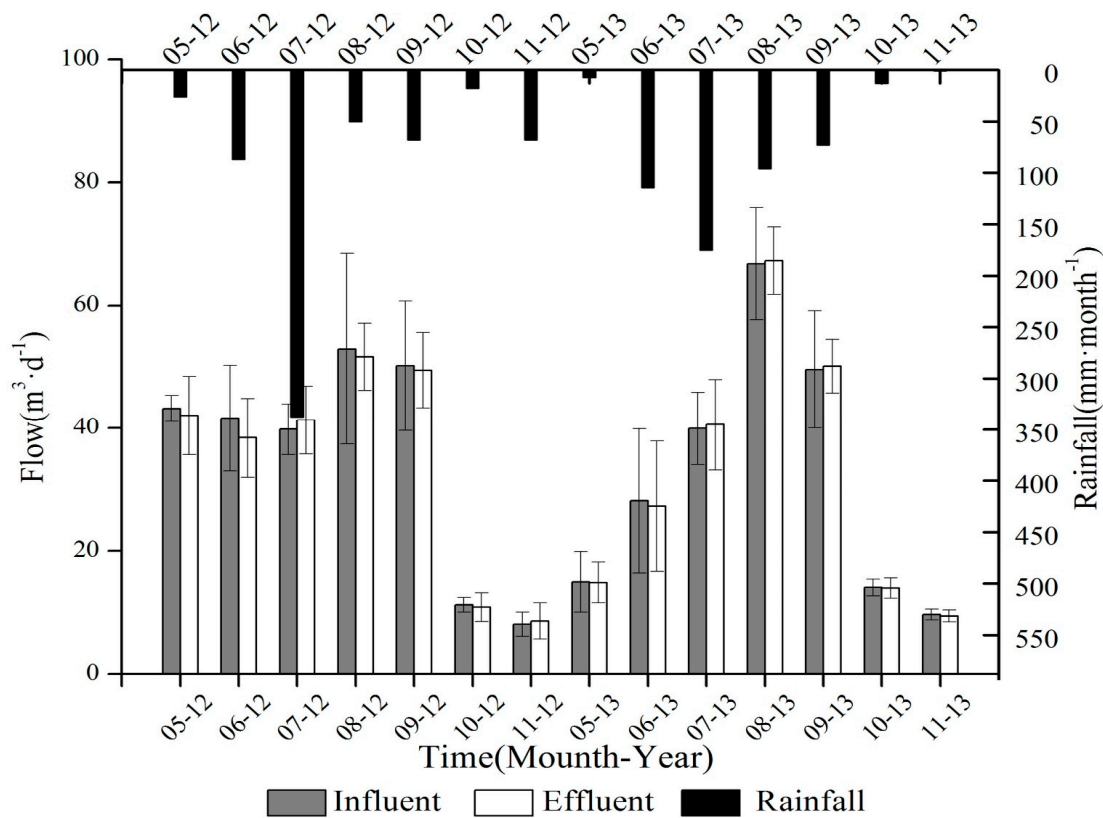Fig.1 Diagram showing the location and layout of the constructed wetland

Fig.2 Average water fluxes, including the influent, effluent, and rainfall, in the HSSF-CW in different months
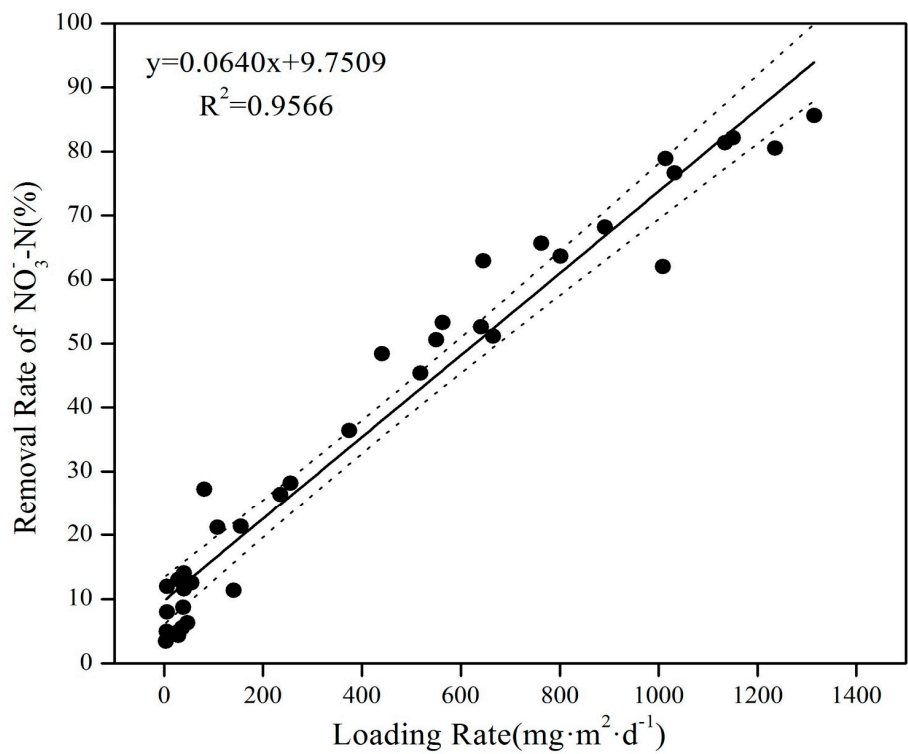

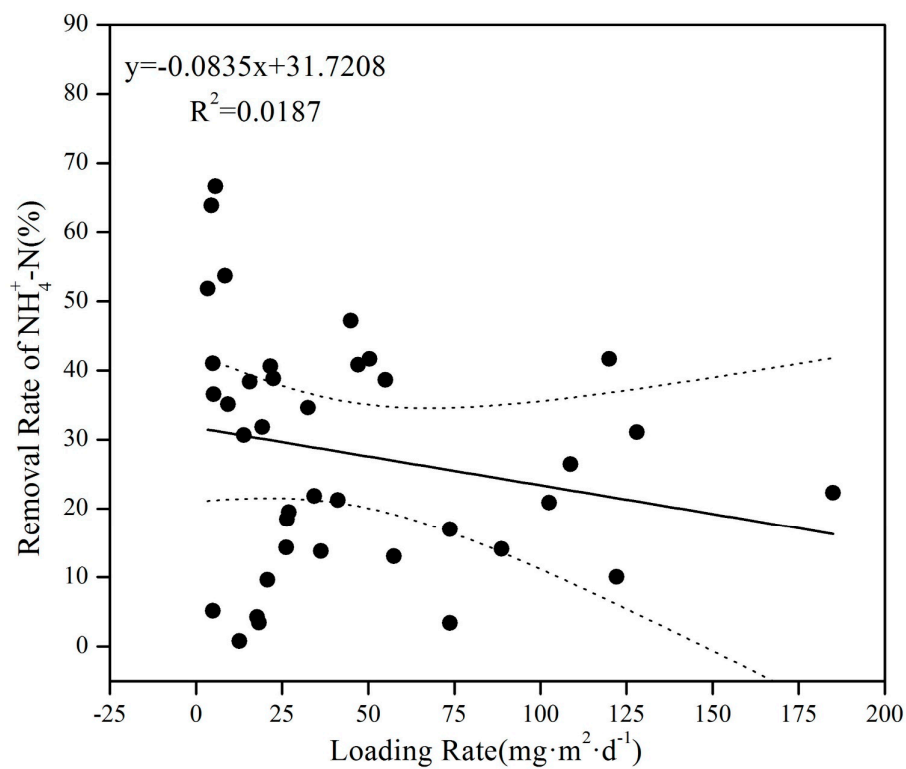Fig.3 Relationship between the areal loading and removal efficiency for NO$_3^-$-N (the dotted line indicates the 99% confidence limit)

Fig.4 Relationship between the areal loading and removal efficiency for NH4+-N (the dotted line indicates the 99% confidence limit)
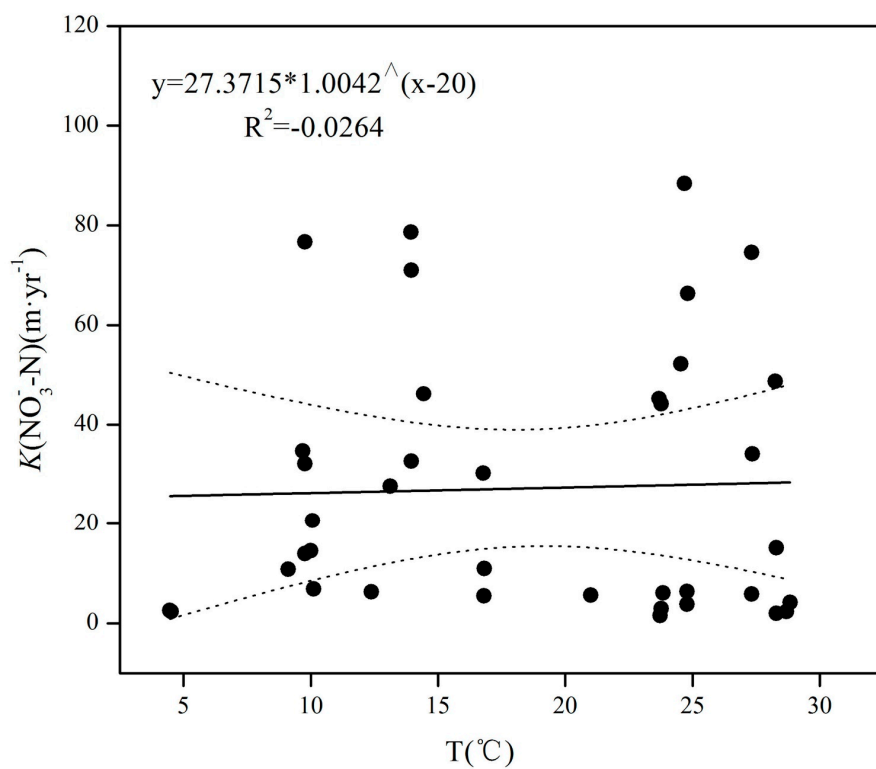

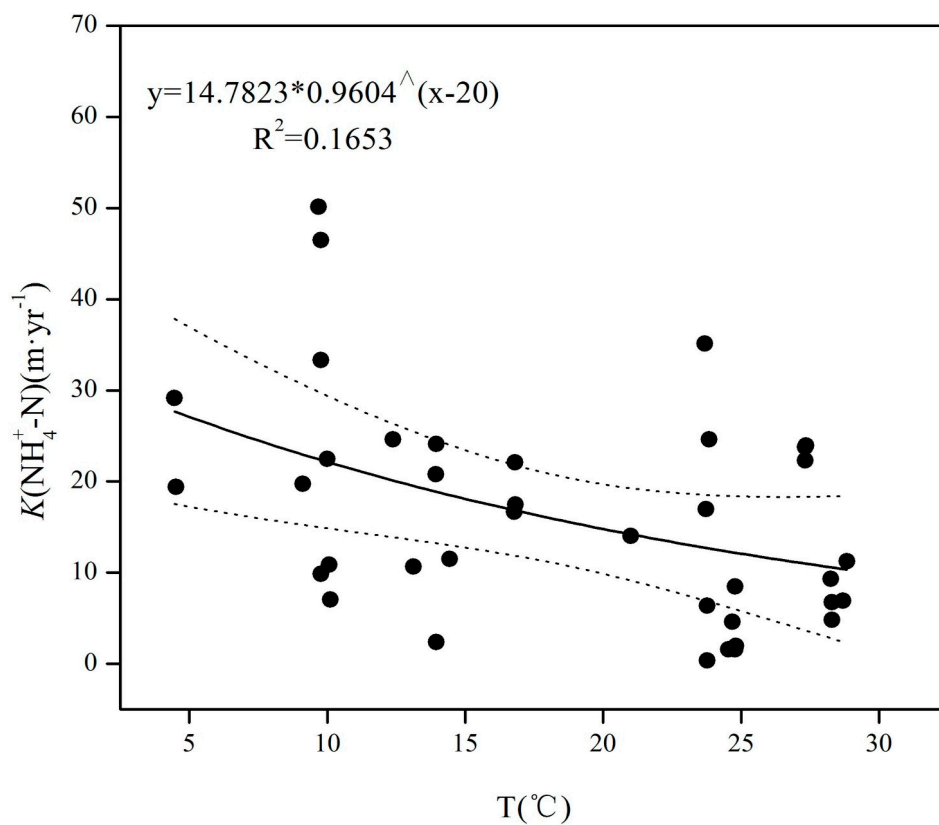Fig.5 Relationship between the temperature and areal rate constants for NO3−-N (the dotted line is the 99% confidence limit)

Fig.6 Relationship between the temperature and areal rate constants for NH4+-N (the dotted line is the 99% confidence belt)
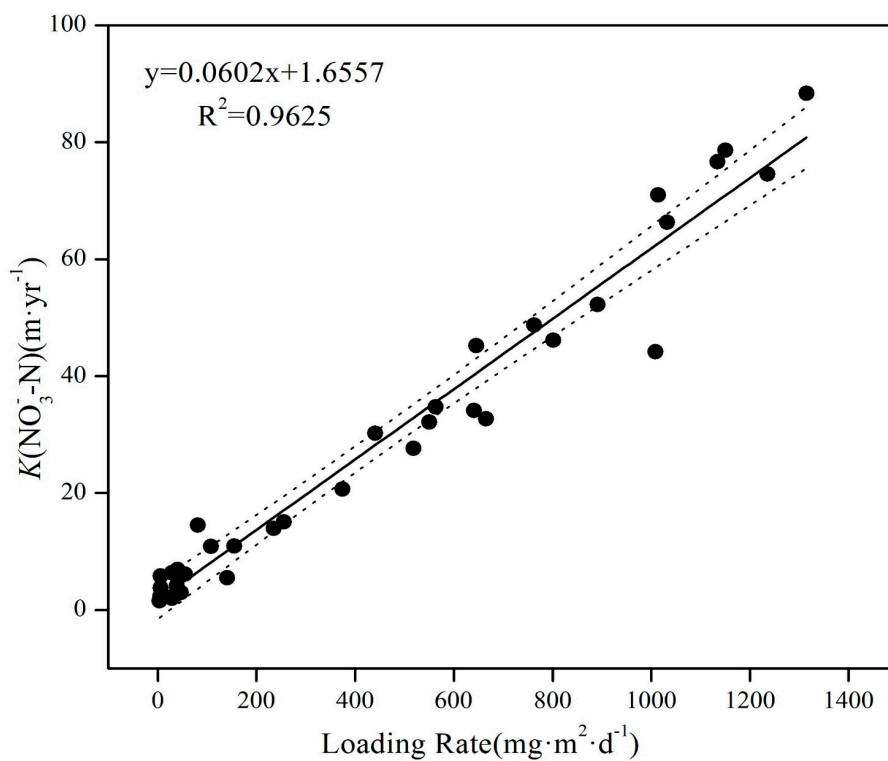

Fig.7 Relationship between the areal loading and areal rate constants for NO3−-N (the dotted line is the 99% confidence limit)
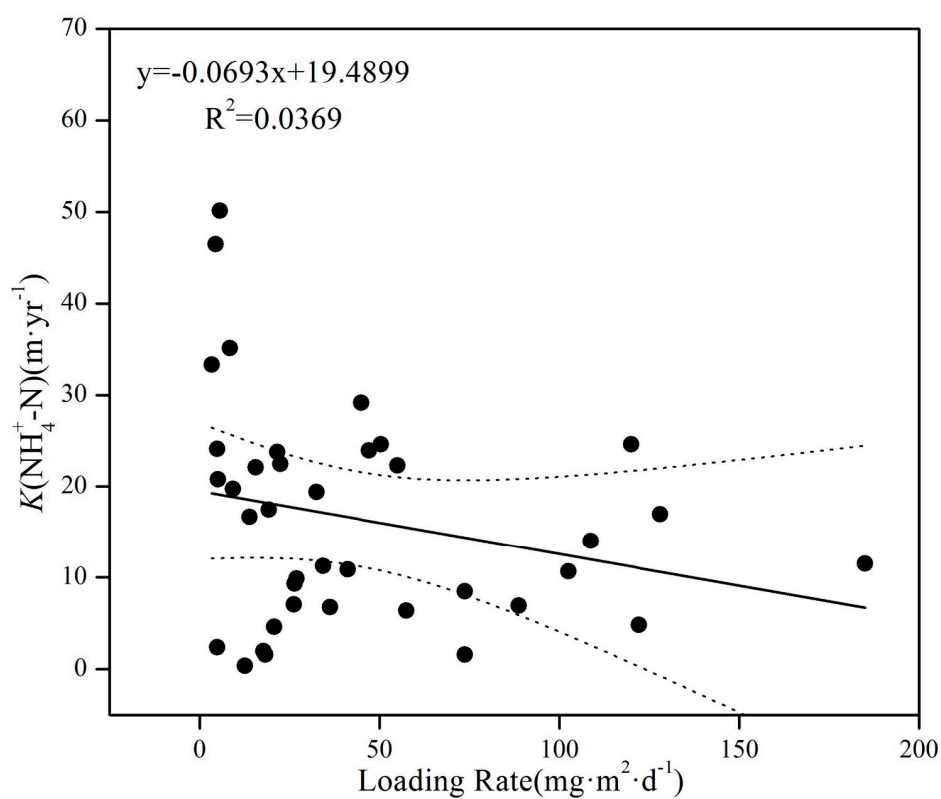
Fig.8 Relationship between the areal loading and areal rate constants for NH4+-N (the dotted line is the 99% confidence limit)

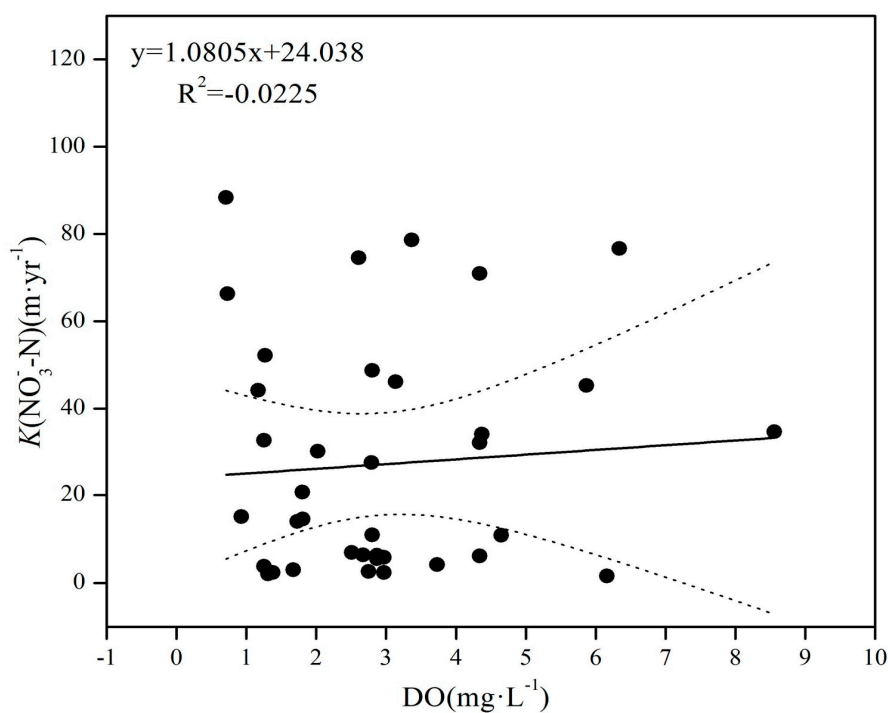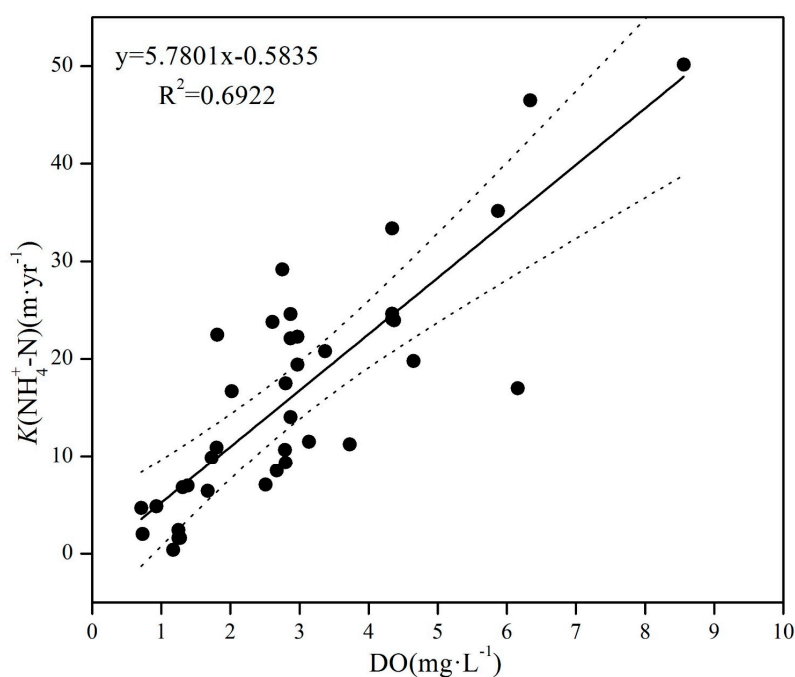Fig.9 Relationship between DO and areal rate constants for NO3−-N (the dotted line is the 99% confidence limit)

Fig.10 Relationship between DO and areal rate constants for NH4+-N (the dotted line is the 99% confidence limit)

Table 1 Configurations of the different treatment cells of the HSSF-CW

Configuration	Treatment cell		
	a	b	c
Length (m)	20	22	25
Area (m^2)	170	160	81
Particle density (g/m^3)	2.43	1.80	1.44
Porosity (%)	91	62	50
Vegetation	*Iris tectorum*		
Plant spacing (m)	2	2	2

Table 2 Flow variations in the HSSF-CW in each treatment cell

Flow (m^3/d)	a		b		c		Total	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Influent	33.59	19.07	33.66	19.13	33.45	19.05	33.59	19.07
Raifall	0.46	0.50	0.43	0.47	0.22	0.24	1.11	1.21
Effluent	33.66	19.13	33.45	19.05	33.27	18.99	33.27	18.99

SD=standard deviation

Table 3 Average influent and effluent nitrogen concentration in the HSSF-CW

Parameter	Unit	n	Influent		Effluent	
			Mean	SD	Mean	SD
NO$_3$-N	mgN L^{-1}	38	3.40	3.44	1.22	0.92
NH$_4^+$-N	mgN L^{-1}	38	0.36	0.35	0.27	0.27

n= sample number, SD=standard deviation

Table 4 Average influent and effluent nitrogen concentrations in the HSSF-CW in different months

Season	Month	n	NO$_3^-$-N (mg L^{-1})				NH$_4^+$-N (mg L^{-1})			
			Influent		Effluent		Influent		Effluent	
			Mean	SD	Mean	SD	Mean	SD	Mean	SD
Spring	4-5	4	0.21	0.20	0.19	0.18	0.60	0.33	0.44	0.25
Summer	6-8	14	4.52	3.94	1.39	1.02	0.34	0.25	0.28	0.23
Autumn	9-11	20	3.25	3.06	1.31	0.82	0.32	0.40	0.23	0.30

n= sample number, SD=standard deviation

Table 5 Areal loading and removal efficiency of the HSSF-CW

Season	Month	n	NO$_3^-$-N			NH$_4^+$-N		
			Loading Rate (mg m^{-2} d^{-1})		Removal (%)	pollution load (mg m^{-2} d^{-1})		Removal (%)
			Mean	SD		Mean	SD	
Spring	4-5	4	25.8	25.6	7.4	75.4	41.9	27.2
Summer	6-8	14	564.6	492.0	44.7	42.2	31.8	19.1
Autumn	9-11	20	406.1	382.0	37.4	39.9	49.7	34.4

n= sample number, SD=standard deviation

Table 6 The areal rate constants and temperature coefficients of the HSSF-CW

Parameter	n	θ	K(m·yr^{-1})		$K_{(20)}$(m·yr^{-1})	
			Mean	SD	Mean	SD
NO$_3^-$-N	38	1.0042	27.22	26.37	27.01	26.49
NH$_4^+$-N	38	0.9604	16.41	11.98	14.63	10.58

n= sample number, SD=standard deviation