Performance of a pilot-scale constructed wetland for stormwater runoff and domestic sewage treatment on the banks of a polluted urban river

Weijie Guo, Zhu Li, Shuiping Cheng, Wei Liang, Feng He and Zhenbin Wu

ABSTRACT

To examine the performance of a constructed wetland system on stormwater runoff and domestic sewage (SRS) treatment in central east China, two parallel pilot-scale integrated constructed wetland (ICW) systems were operated for one year. Each ICW consisted of a down-flow bed, an up-flow bed and a horizontal subsurface flow bed. The average removal rates of chemical oxygen demand (CODCr), total suspended solids (TSS), ammonia (NH4+-N), total nitrogen (TN) and total phosphorus (TP) were 63.6, 91.9, 38.7, 43.0 and 70.0%, respectively, and the corresponding amounts of pollutant retention were approximately 368.3, 284.9, 23.2, 44.6 and 5.9 g m⁻² yr⁻¹, respectively. High hydraulic loading rate (HLR) of 200 mm/d and low water temperatures (<15°C) resulted in significant decrease in removals for TP and NH4+-N, but had no significant effects on removals of COD and TSS. These results indicated that the operation of this ICW at higher HLR (200 mm/d) might be effective and feasible for TSS and COD removal, but for acceptable removal efficiencies of nitrogen and phosphorus it should be operated at lower HLR (100 mm/d). This kind of ICW could be employed as an effective technique for SRS treatment.

Key words | constructed wetland, hydraulic loading rate, nutrient removal, polluted urban river, stormwater runoff and domestic sewage

INTRODUCTION

More and more sewer systems and municipal wastewater treatment plants are being built or upgraded in China and the final effluent of these plants should satisfy the ‘Discharge standard of pollutants for municipal wastewater treatment plant (GB 18918–2002, PR China)’ that was issued in 2002 and enforced in 2003 (Environmental Protection Administration 2002). However, drainage systems are still open channels or scour channels in many areas including small towns, unrenovated urban-villages and most rural areas. Consequently, the amount of stormwater runoff and domestic sewage (SRS) discharged into water bodies increases greatly in the rainy season, because a large quantity of SRS is far beyond the capacity of the sewer system. These diffuse or non-point sources of pollution from impervious or pervious surfaces, usually including organic and inorganic contaminants, total suspended solids (TSS), and phosphorus and nitrogen, contributes significant quantities of pollutants and high nutrient loads to the surrounding surface waters (Boving & Neary 2007), and is identified as a significant contributor to water quality impairment in urban and suburban areas (Malaviya & Singh 2012).

Restoration engineering for polluted surface waters is being designed and employed in increasing numbers across the world. Many corresponding restoration techniques are being employed that vary from ‘hard’ structural approaches to ‘soft’ bioengineering approaches (Brown 2000). Some typical measures described as river restoration projects include stormwater management, bank stabilization, channel reconfiguration and riparian replanting (Bernhardt & Palmer 2007). Furthermore, other acceptable ecological engineering approaches such as artificial floating islands, macrophytes restoration, river aeration and biomanipulation methods are being implemented to improve damaged rivers. Constructed wetlands (CWs) have been accepted as a common treatment alternative all over the world, especially suitable for some...
developing countries, where sewage collection and treatment facilities are far from being sufficient (Hadad et al. 2006). CWs are used not only to treat domestic sewage or industrial wastewater, but also to remove heavy metals and xenobiotics from polluted surface water (Cheng et al. 2002a, b). Furthermore, CWs could be used for non-point source pollution control (Lai & Lam 2009; Wu et al. 2010; Maniquiz et al. 2011). These studies have supplied lots of valuable experience about the establishment, operation and performance of CWs, while the application on an urban river bank to treat SRS (stormwater management) has rarely been reported. To test the performance and feasibility of CWs for stormwater management, two sets of pilot-scale integrated constructed wetlands (ICWs) were demonstrated for SRS treatment on the banks of a polluted urban river, in which SRS discharged through two open channels. The experimental results were compared with the maximum allowed values enacted by Chinese legislation to verify whether the effluent of these ICWs could achieve the discharge standard of GB 18918–2002 (Environmental Protection Administration 2002).

METHODS AND MATERIALS

Site description and characteristics of influent

Two parallel pilot-scale ICW systems were established on the bank of the upper reaches of the Nanfeihe River in Heifei, China (31°52′40.89″ N, 117°13′22.89″ E; elevation, 15 m). The Nanfeihe River, as a tributary and the biggest pollution source of Chaohu Lake, contributes 21.3% of chemical oxygen demand (COD), 24.8% of total phosphorus (TP), and 8.8% of total nitrogen (TN) annual input into the whole lake, and it is also a typically polluted urban river in the Yangtze River Basin (Wu et al. 2013). Nearby, there was a sewage pumping station, which mainly received raw SRS from one open channel. The open channel is formed by two converging tributaries, one mainly receiving domestic sewage in the dry season and both of them receiving stormwater runoff in the rainy season, together draining approximately a 750 ha catchment area (Figure 1).

During the dry season, SRS was mainly composed of domestic sewage from one channel serving an unrenovated urban-village and most sewage was transmitted to Wangtang Treatment Plant through sewer lines after a simple preliminary handling by bar screens and a sedimentation tank. Wangtang Treatment Plant located in northwest Hefei city was built in 2003 with a daily disposal capacity of 180 thousand tons. It serves a 66 square kilometer area and receives mostly domestic sewage and some industrial sewage. A carrousel oxidation ditch and V filtering tank progress are adopted for sewage treatment and the effluent quality conforms to first-grade criteria of GB 18918–2002 (Environmental Protection Administration 2002). However, in the rainy season, even with moderate rainfall intensity, stormwater runoff from the two channels exceeds the transport capacity of the sewer lines. Thus, a great quantity of raw mixed wastewater, mainly in the form of stormwater runoff with a small amount of domestic sewage, discharged directly into the Nanfeihe River. The influent of this ICW was pumped from the aforementioned sedimentation tank and the characteristics are given in Table 1.

Design and operation of ICW

Two parallel pilot-scale constructed wetland sets were constructed in early May 2011 and each was designed
as a combination system of a down-flow bed (50 m², planted with *Canna indica*), an up-flow bed (50 m², planted with *Iris pseudacorus*) and a horizontal subsurface-flow bed (50 m², planted with *Acorus calamus*) in series (Figure 2(a)). Slags and zeolites were filled in the upper level of the down-flow bed and the up-flow bed, respectively, and gravels filled in both their sub-layers and the whole horizontal subsurface-flow bed. The average porosity of the ICW substrates was approximately 0.34 (Figure 3). Plants, which are commonly used species in CWs, were transplanted in mid-April 2011 with an original density of 16 ind./m². The plants grew vigorously from July (Figure 2(b)–(d)). The two sets were operated at a hydraulic loading rate (HLR) of 100 mm/d from mid-June 2011 to the beginning of September 2011, then 200 mm/d from mid-September 2011 to mid-October 2011, and 100 mm/d from the beginning of November 2011 to mid-June 2012. The influent was induced intermittently in every ICW set per day by a lift pump during the whole experiment.

Table 1 | Characteristic values of quality parameters for the influent from each ICW in the dry season (*n*, sample number = 9) and the rainy season (*n* = 9)

<table>
<thead>
<tr>
<th>Parameters</th>
<th>In dry season*</th>
<th>In rainy season</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>Range</td>
<td>Mean</td>
</tr>
<tr>
<td>COD&lt;sub&gt;Cr&lt;/sub&gt; (mg/L)</td>
<td>121–335</td>
<td>208</td>
</tr>
<tr>
<td>TSS (mg/L)</td>
<td>2.1–126.0</td>
<td>62.7</td>
</tr>
<tr>
<td>NH&lt;sub&gt;4&lt;/sub&gt;-N (mg/L)</td>
<td>11.1–49.4</td>
<td>26.9</td>
</tr>
<tr>
<td>TN (mg/L)</td>
<td>19.1–73.1</td>
<td>42.4</td>
</tr>
<tr>
<td>TP (mg/L)</td>
<td>0.9–5.5</td>
<td>2.8</td>
</tr>
<tr>
<td>C/N&lt;sub&gt;c&lt;/sub&gt; (mg/L)</td>
<td>5.9–7.2</td>
<td>5.1</td>
</tr>
</tbody>
</table>

*The dry season is during the period October 2011–April 2012 and the rainy season from two periods: from June to September 2011 and from May to June 2012.

bSD, standard deviation.

C/N is the ratio of COD<sub>Cr</sub>/TN.

Figure 2 | (a) Schematic diagram of the ICW; (b) ICW in late April; (c) ICW in late July; (d) ICW in late October.

Figure 3 | Type, distribution and grain diameter of the substrates used in the ICW.
Sampling and analysis

The influent and effluent of the ICWs were sampled once every half month from June to November 2011, and then once a month from November 2011 to June 2012. Parameters of COD by the dichromate method (COD\textsubscript{Cr}), TSS, nitrite (NO\textsubscript{2}-N), nitrate (NO\textsubscript{3}-N), ammonium (NH\textsubscript{4}+-N), TN and TP were analyzed according to the Standard Methods of Environment Monitoring in China (National Bureau of Environment Protection 2002). Turbidity was analyzed using a turbidimeter (2100Q HACH, USA) and other physical and chemical characteristics including pH, dissolved oxygen (DO) and water temperature (T) were measured using a portable Multimeter (YSI Proplus, USA).

Data analysis

The data used were the mean values of the two parallel sets, except from mid-June to mid-July when individual values of each ICW were unable to be determined and so the data used were from their mixed effluents. The concentrations of organic nitrogen (Org-N) in samples were calculated by taking the difference between TN and the sum of NH\textsubscript{4}+-N and NO\textsubscript{x}/C\textsubscript{0}-N including nitrate and nitrite:

\[
\text{Org-N} = \text{TN} - \text{NH}_4^+ \text{-N} - \text{NO}_x^-/C_0\text{-N}
\]

Statistical analyses were executed with SPSS 13.0 (SPSS Inc., Chicago, IL, USA). The data were analyzed through a non-parametric test to detect the effect of HLR (100 mm/d and 200 mm/d) and water temperature (T > 15 °C and T < 15 °C) on removal efficiencies taking \( P < 0.05 \) as a significant difference.

RESULTS

Turbidity, temperature, DO and pH variation

The turbidity values in the effluent remained at a low level of 8.0 NTU, although it oscillated drastically from 28.6 to 395.0 NTU in the influent (Figure 4(a)). A transient increase of effluent turbidity was observed when the ICW operated at 200 mm/d HLR. The water temperatures in the influent varied from 8.3 to 29.4 °C and were all below 15 °C from December 2011 to March 2012. Average temperature in the effluent decreased by 1.2 °C in comparison with that of the influent (Figure 4(b)). The average concentration of DO was 1.6 mg/L within the range of 0.2–4.8 mg/L in the influent, and dropped further to below 0.5 mg/L in the effluent as a result of oxygen consumption by the decomposition of organics (Figure 4(c)). In addition, the pH of the influent was 7.5–8.1 with an average of 7.8, and increased significantly to a mean value of 8.9 ranging from 7.6 to 10.5 in the effluent, especially in the initial operation stage (Figure 4(d)); this was caused by steel slags that tended to generate high levels of alkalinity (10–11) over extended periods in both laboratory and field studies (Ziemkiewicz 1998).
Removal of TSS, COD and TP

TSS removal by the ICW was efficient although it suffered a brief drop at a higher HLR of 200 mm/d (Figure 5(a)). The concentrations of TSS in the influent varied considerably (2.1–481.0 mg/L), but those in the effluent decreased to 0.1–9.5 mg/L with an average removal rate of 91.9%. The 200 mm/d HLR seemed to have no continuing impact on TSS removal. The average removal rate of COD was 63.6%, within a range from 36.5 to 88.7% (Figure 5(b)). While the removal rate underwent a short descent to 62.6% at an HLR of 200 mm/d, it seemed that the increased HLR of 200 mm/d had little effect on COD removal. The average removal rate of TP by the ICW was 70.0%, within a range of 40.7 to 97.2% (Figure 5(c)). When the ICW was operated at 200 mm/d HLR, the removal rates experienced continuous decline and went to 75.3, 66.1 and 42.3%, successively, then rose to 72.1% with the HLR dropping to 100 mm/d, which indicated the obvious effect of HLR on TP removal.

Removal of nitrogen

The mean removal rates of NH$_4^+$-N and TN were 50.5 and 49.1%, respectively, when the system operated with 100 mm/d HLR from June to September 2011 and decreased gradually to an average 30.2 and 32.8%, respectively, with the HLR increasing to 200 mm/d (Figure 6(a and b)). A marked decrease of their removal was observed from mid-December 2011 to mid-May 2012. Especially from March to May 2012, nitrogen was even released from the bed and negative removal rates occurred (circled with a dotted in Figure 6). The mean removals of the experimental system were 43.0% and 38.7% for TN and NH$_4^+$-N.

Figure 5 | Concentrations of TSS (a), COD (b) and TP (c) in the influent and effluent, and removal efficiencies of the ICW.

Figure 6 | Concentrations of NH$_4^+$-N (a) and TN (b) in influent and effluent, and the removal efficiencies of the ICW.
respectively, during the whole experiment (not including N-releasing months from March to May 2012).

The composition diagram of different nitrogen (N) forms (for example, Org-N, NO₃⁻N, NO₂⁻N and NH₄⁺-N) shows that the prevalent N form (was NH₄⁺-N with an average proportion of 65.2% in the influent and 65.0% in the effluent, whilst the second was Org-N with 31.6% in the influent and 30.0% in the effluent (Figure 7).

DISCUSSION

Impacts on removal efficiencies of the ICW

It is necessary to find optimal design parameters to enhance the treatment performance of CWs. Many factors, such as HLR, influent concentration, temperature, vegetation type, wetland configuration and porous media could affect the removal efficiencies of CWs. Higher HLR results in higher nutrient loading and shorter hydraulic retention time (HRT). Moreover, the treatment function of CWs is based on natural processes and seasonal temperature change has direct effects on pollutant removal, especially for N removal, which is strongly influenced by the activity of microorganisms and related enzymes from the substrate or the root system (Kuschk et al. 2005). The effects of HLR and water temperature on removal efficiencies in this study are listed in Table 2.

Neither high HLR (200 mm/d) nor low temperatures (<15 °C) had any significant effect on the removal of COD and TSS in our study. Similarly, the removal rates of COD and TSS were generally efficient, and decreased only slightly at relatively higher HLR (Dong et al. 2011). This can be explained by the fact that suspended solids (SS) are mainly removed by physical mechanisms, such as sedimentation and filtration processes, on which the influence of temperature is comparatively low. This indicates that ICWs possess better reliability and flexibility to remove TSS and COD when met with a sudden increase of pollutant concentration in the influent.

HLR and the consequent detention time of the water are important for N treatment processes in CWs, with significant lower removals of TN and NH₄⁺-N at a higher HLR of 200 mm/d in our system. Tunçsiper (2009) reported that the average removal efficiency of nitrogen increased as HLR decreased (from 100 to 30 L/m²/d) in a three-stage constructed wetland which was composed of three connected beds in series, a vertical flow-gravel filter bed, a horizontal-subsurface flow bed and a vertical-subsurface wetland configuration and porous media could affect the removal efficiencies of CWs.

Table 2 | Mean (±SD) pollutant removal rates (%) and results of non-parametric test at different hydraulic loading rates (HLRs) and water temperatures

<table>
<thead>
<tr>
<th>Variable</th>
<th>HLR*</th>
<th>Temperature a</th>
<th>P-value b</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>100 mm/d (n = 6)</td>
<td>200 mm/d (n = 6)</td>
<td></td>
</tr>
<tr>
<td>CODₜₜ</td>
<td>66.0 ± 23.5</td>
<td>57.1 ± 17.8</td>
<td>0.310</td>
</tr>
<tr>
<td>TSS</td>
<td>92.8 ± 2.8</td>
<td>82.8 ± 22.1</td>
<td>0.590</td>
</tr>
<tr>
<td>NH₄⁺-N</td>
<td>62.2 ± 19.6</td>
<td>32.8 ± 3.1</td>
<td>0.002 d</td>
</tr>
<tr>
<td>TN</td>
<td>56.7 ± 11.3</td>
<td>30.2 ± 14.6</td>
<td>0.002 d</td>
</tr>
<tr>
<td>TP</td>
<td>79.9 ± 8.8</td>
<td>60.6 ± 17.0</td>
<td>0.041 d</td>
</tr>
<tr>
<td></td>
<td>&lt;15 °C (n = 6)</td>
<td>&gt;15 °C (n = 12)</td>
<td></td>
</tr>
<tr>
<td>CODₜₜ</td>
<td>71.7 ± 3.3</td>
<td>72.3 ± 14.5</td>
<td>0.120</td>
</tr>
<tr>
<td>TSS</td>
<td>94.0 ± 4.7</td>
<td>94.1 ± 6.9</td>
<td>0.850</td>
</tr>
<tr>
<td>NH₄⁺-N</td>
<td>22.8 ± 28.3</td>
<td>53.4 ± 22.9</td>
<td>0.024 d</td>
</tr>
<tr>
<td>TN</td>
<td>36.9 ± 18.3</td>
<td>52.7 ± 14.1</td>
<td>0.140</td>
</tr>
<tr>
<td>TP</td>
<td>67.2 ± 18.5</td>
<td>85.3 ± 9.7</td>
<td>0.012 d</td>
</tr>
</tbody>
</table>

*Removal rates (%) of experimental system at 100 mm/d are from mid-August, early September and November 2011, and those at 200 mm/d from mid-September, early October and mid-October 2011.

All variables (HLR, n = 12; Temperature, n = 18) performed for the non-parametric test were the respective removal rates of each parallel set rather than averages of the two sets.

All variables (n = 18) are obtained when the ICW operated at 100 mm/d HLR, variables of T > 15 °C were from mid-June to September 2011 and those of T < 15 °C were from December 2011 to February 2012.

P < 0.05 significant.
flow bed. Shorter HRT caused by increasing HLR can directly reduce the reaction time of water and substrates/microbes. In our study, a lower HLR (100 mm/d) could be optimal for removing N for this kind of ICW compared to a higher HLR of 200 mm/d.

Nitrification and denitrification are the main mechanisms for N removal. Kuschk et al. (2003) demonstrated that denitrification was clearly restricted at seasonal temperatures below 15 °C, which is usually considered as the lower limit of the optimal range of the microorganisms responsible for N removal. Nitrogen was present mainly as NH₄⁻-N, then as Org-N, in the influent of this study. The water temperatures in the influent were less than 15 °C in winter (from December 2011 to March 2012), and NH₄⁺-N removal decreased significantly compared to those in summer (from June to September 2011), which agreed with a previous report by Stefanakis & Tsihrintzis (2009). Of course, DO concentration is also an important factor for N transformation in CWs. Zhu & Sikora (1995) pointed out that no obvious nitrification could be observed when DO concentration of the sewage in gravel bed microcosm wetlands (0.4×0.35 m² surface area and 0.5 m depth) is lower than 0.5 mg/L. The DO concentration below 0.5 mg/L in effluent probably caused the lower removal efficiencies of NH₄⁺-N and Org-N in this study. Furthermore, N removal could be affected intensively by different influent C/N ratios. Wu et al. (2009) reported that the system was most effective on N removal at a C/N of 5:1, and high N removal efficiency occurred when the C/N ratio ranged 2.5–5 in another study (Zhao et al. 2010). The mean C/N ratio in the influent was 5.5 which could be appropriate for N removal in this ICW. Limited artificial aeration in CWs presented better performances for organic matter and N removal and is a cost-effective method for treating domestic wastewater (Zhang et al. 2010). So, besides supplying sufficient carbon sources, strengthening oxygen enrichment for influent via aeration to facilitate nitrification or ammonification is essential for improving the removal efficiency of such SRS enriched in NH₄⁺-N and Org-N.

In addition, the withered Canna indica and Acorus Calamus tended to be decomposed by the microbes and N release occurred with the rise of water temperature during this period. In our study, NH₄⁺-N and organic-N were the two predominant N forms in the influent and the ammonification rate of organic-N might proceed more rapidly than nitrification in some cases, leading to N release.

The processes responsible for P removal in CWs are adsorption by sediments and uptake by microbes and plants. During the experiment, TP removal efficiency decreased significantly at high HLR (200 m/d) and low temperature (T < 15 °C). Lu et al. (2009) reported that HRT had a significant correlation with P removal efficiency, whereas water temperature had a slight influence on P removal, in which a free-water surface CW was applied to treat agricultural runoff. Nevertheless, the opposite conclusion that P removal showed a significant dependence on temperature and that the negative removal rates even occurred during winter (at temperatures below 15 °C) has been reported by Akratos & Tsihrintzis (2007), in which five pilot-scale horizontal subsurface flow constructed wetlands were employed and operated continuously for two years in parallel experiments. The water passes rapidly to the effluent at higher HLR reducing the contact time of the pollutants and substrates (slag, zeolite and gravel). Furthermore, the higher HLR could result in an increase of effluent TSS sometimes enriched with particle-bound nutrients (particulate organic P and N), probably leading to lower removals. Slags used in our ICW may also play an important role in P removal due to their superior adsorption performances (Ayaz et al. 2012).

In general, HLR and temperature, as described above, could be deemed as two key parameters for better treatment performance of CWs. TSS and COD removal were effective and feasible whether the ICW operated at 200 mm/d HLR or 100 mm/d HLR. For nitrogen and phosphorus, operation at 100 mm/d HLR is adequate to achieve acceptable removal efficiencies when the water temperature was above 15 °C, while lower HLR might be adopted when below 15 °C. Furthermore, the influent was characterized by higher concentration of pollutants but less water quantity in the dry season usually accompanied by lower temperatures, but the opposite was the case in the rainy season. Thus, selecting lower HLRs in the dry season and adopting higher HLRs in the rainy season are recommended when employing the ICW to treat SRS in other similar projects.

**Contribution of ICW application on river restoration**

The Nanfeihe River is interrupted by the Dongpu Reservoir in the upstream and goes through Hefei City. It is around 20 meters wide and less than 2 meters deep and has nearly become an encased river. It receives water mainly from tributaries, stormwater runoff and the effluent from wastewater treatment plants. Consequently, its self-purification capacity is weak and the ecosystem is degraded, even a small amount of SRS discharged into the river means a serious overloading and a huge impact on the water quality.
The two parallel systems, achieved a mean of one-year effluent concentrations of 49.9 mg/l for COD, 2.7 mg/l for TSS, 0.6 mg/l for TP, 19.2 mg/l for TN and 14.1 mg/l for NH$_4^+$-N and all of them measured up to the first grade standard of GB 18918–2002 (Environmental Protection Administration 2002). The annual amounts of pollutants removed by such an ICW were approximately 368.3 g COD, 284.9 g TSS, 23.2 g NH$_4^+$-N, 44.6 g TN and 5.9 g TP m$^{-2}$ yr$^{-1}$ in this study. The Kaoping River Rail Bridge Constructed Wetland, designed with a combination of different functions such as water treatment, ecological conservation and wildlife habitat, had a significant effect on water quality improvement and was capable of removing most of the non-point pollution (about 40% of TN and TP) containing domestic, agricultural and industrial wastewaters before they were discharged into the water body (Wu et al. 2010). Mustafa et al. (2009) reported that an integrated CW mainly composed of horizontal surface flow beds with different macrophytes had been deemed as an effective and sustainable wastewater treatment option for farmyard runoff treatment with good removal efficiencies (94.9% for COD, 95.7% for SS, 99% for NH$_4^+$-N and 91.8% for reactive-P), and this had improved the water quality of surrounding streams without degrading the groundwater. After wetland construction, significant average reduction of 43% for NO$_3^-$-N and 72% for NH$_4^+$-N were realized in downstream Hewletts Creek, as indicated by Mallin et al. (2012). Moreover, Birch et al. (2004) investigated the removal efficiency of contaminants in urban stormwater by a 3.1 ha surface flow wetland constructed in the Sydney catchment, indicating that the wetland was moderately efficient in removing contaminants from urban stormwater and could alleviate the degradation of the receiving basin waters and bottom sediments. These findings indicate that integrated CWs are efficient for stormwater management and could be constructed as an available measure on or near a river bank for water quality improvement.

In conclusion, an ICW created on the bank of a polluted urban river could be an effective measure to treat SRS and contribute to the river’s restoration. HLR and water temperature had significant impacts on the removal efficiencies of pollutants and it is necessary to adopt corresponding operation strategies in different seasons. Lower HLRs in the dry season and higher HLRs in the rainy season would be a recommended operation mode in the application of such an ICW to treat SRS. Decaying litterfall in the wetland might increase effluent concentration of pollutants through the decomposition of decayed litter and so plant harvest would be necessary. The ICW showed non-adequate nitrification and ammonification for lower DO concentration, and pre-aeration could be used to improve nitrification.

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