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Impact of Hydraulic Loading Rate and Season on Water Contaminant Reductions Within Integrated Constructed Wetlands

Yu Dong · Piotr R. Wiliński · Mawuli Dzakpasu · Miklas Scholz

Abstract The impact of hydraulic loading rate (HLR) and seasonal temperature on contaminant removal efficiencies within an integrated constructed wetland (ICW) system of 3.25 ha was assessed. The ICW system was designed to treat domestic wastewater from Glaslough (Ireland). The current loading rate is 800 population equivalents. The system has shown good removal performances (2008 to 2010). Mean concentration removal efficiencies were high: 92% for chemical oxygen demand (COD), 98% for the 5 days at 20°C N-allylthiourea biochemical oxygen demand (BOD), 94% for total suspended solids (TSS), 97% for ammonia-nitrogen (NH_3-N), 90% for nitrate-nitrogen (NO_3-N), 96% for total nitrogen (TN), and 96% for molybdate reactive phosphate (MRP). The mean mass removal efficiencies were 92% for COD, 98% for BOD, 96% for TSS, 92% for NH_3-N, 83% for NO_3-N, 90% for TN, and 91% for MRP. Loading rate fluctuations were mainly due to high variation in rainfall (0.4 to 400 m³ day⁻¹) and in evapotranspiration rate (0 to 262 m³ day⁻¹). The influence on the removal efficiencies of the hydraulic loading rate (−0.7 to 15.3 cm day⁻¹), which was based on overall water balance, was negligible. This implies that the large footprint of the system provides a high hydraulic retention time (92 days).

Introduction

Domestic wastewater contains a large amount of nutrients (particularly nitrogen and phosphorus), TSS and organic matter that can lead to persistent eutrophication and a loss of recreational and commercial value for the receiving waterways (Cloern 2001). Therefore, effective and cost-saving domestic wastewater treatment is increasingly required for disease prevention around the world. Constructed wetland systems, which are characterized by their high capacity to remove nutrients from effluent streams and by their low operational and maintenance costs, have become known as an increasingly popular alternative to conventional treatment systems (EPA 1995). Since the 1990s, the application of wetland systems to treat numerous types of domestic and industrial wastewaters has been expanding radically. This low energy-consuming 'eco-friendly' technique provides a high financial incentive backed by public support to address concerns regarding the rising price of fossil fuels and negative impacts of climate change (Lee et al. 2009). However, the treatment efficiencies vary considerably depending on variables such as system type and design, retention time, hydraulic and nutrient mass loading rates, climate, vegetation, and microbial communities (EPA 1995).

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The ICW concept is based upon the free surface flow of water through a series of sequential shallow ponds vegetated with a range of emergent plant species (Scholz et al. 2007a,b). No artificial liners such as plastic or concrete are used in the construction of ICW systems. Usually, ICW systems comprise many plant species to facilitate microbial and animal diversity (Nygaard and Ejrnæs 2009; Jurado et al. 2010), and are generally aesthetically appealing, which leads to an enhancement of recreation and amenity values. Using physical, chemical, and microbiological parameters, Mustafa et al. (2009) assessed the long-term performance of a representative ICW treating farmyard runoff. The system demonstrated a relatively good reduction of contaminant concentrations: 20°C N-allylthiourea biochemical oxygen demand (BOD, 97.6%), chemical oxygen demand (COD, 94.9%), total suspended solids (TSS, 93.7%), ammonia-nitrogen (NH_4-N, 99.0%), nitrate-nitrogen (NO_3-N, 74.0%) and molybdate reactive phosphate (MRP, 91.8%). On the back of this success with ICW treating agricultural runoff, a new industrial-scale ICW system was constructed in 2006/2007 to treat combined sewage from the village of Glaslough near Monaghan, Ireland.

The polluted effluent is treated in an ICW system through complex physical, chemical, and biological processes which involve emergent plants, microorganisms, soil, sediments, water, and sunlight (Kadlec and Knight 1996; Scholz 2006). The nitrogen removal in wetland systems is accomplished primarily by physical settlement, ammonia stripping, ion exchange, ammonification, nitrification/denitrification, and plant and microbial uptake (Healy and Cawley 2002). While the combination of nitrification and denitrification is usually considered as the most significant nitrogen removal process, a recent study of nitrogen transformations has shown that anammox (anaerobic ammonium oxidation), one of the possible biodegradation processes, could provide a potential mechanism for improving TN removal. In this process, ammonium is autotrophically oxidized to nitrogen gas, while nitrite plays the role of an electron acceptor under anaerobic conditions (Lee et al. 2009). The major environmental factors influencing the removal of nitrogen from constructed wetlands include temperature, hydraulic retention time (HRT), type and density of vegetation, the amount of biomass and characteristics of microbial communities, climate, and influent quality. Of these, temperature and HRT are the most important (Kuschk et al. 2003).

There is no gaseous loss or valence change in phosphorus removal from wetland systems (Prithviraj and Keith 2008). While the main removal mechanisms of phosphorus are short-term or long-term storage in soil/sediments and (to a lesser extent) plant uptake, factors such as influent phosphorus concentration, form of phosphorus, rate of internal biomass cycling, and wetland age significantly influence the phosphorus assimilation in constructed wetlands (Kadlec 1999a,b; Wallace and Knight 2006; Mustafa et al. 2009). Previous studies have shown that at phosphorus loading rates of less than 5 g P m^{-2} year^{-1}, the young wetland sediment can absorb more than 90% of the total incoming phosphorus (Faulkner and Richardson 1989). Scholz et al. (2007b) also revealed the positive correlation between phosphorus removal efficiency and wetland area based on the database of free surface flow ICW published by Dunne et al. (2005) and Harrington et al. (2005).

Constructed wetlands have demonstrated a consistent capacity to decompose organic compounds, which are measured as biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total dissolved solids (TSS). The main removal processes include volatilization, photochemical oxidation, sedimentation, sorption, and biodegradation (Kadlec 1992). Organic molecules are broken down by the microbiota through fermentation and/or respiration, and mineralized as a source of energy or assimilated into biomass (Moshiri 1993; Kadlec and Knight 1996).

Most ICW are composed of several different types of vegetation. The presence of plants can strongly affect the seasonal patterns of wastewater treatment performance. Plant rhizosphere stimulates microbial communities to grow. Therefore, any modification of plant diversity and habitat is likely to lead to changes within the microbial community and subsequently to influent contaminant removal efficiency (Boven et al. 2008). In addition, the diversity of vegetation is another important factor with respect to contaminant uptake capacity. Hook et al. (2003) studied the removal efficiencies of various plant types during the winter period. The results show that the COD removal efficiency varies between 60% for *Typha latifolia* and more than 90% for *Carex rostrata*. Furthermore, oxygen concentration within the water body and associated sediment has a great impact on microorganisms, and therefore also on treatment efficiency. The internal gas transport mechanism within wetland plants was classically believed to be passive diffusion along the concentration gradients of the individual gases. Previous studies reported that oxygen concentration in *Phragmites australis* decreased from 20.7% in the upper parts of plants to 3.7% in the growing rhizomes (Sutula and Stein 2003; Vymazal and Kröpfelová 2008). Anaerobic conditions usually appear below the sediment-water interface. This vertical distribution of oxygen influences the oxidation and reduction reactions which are critical for nutrient and organic matter transformation (Sutula and Stein 2003).

Previous studies indicate that hydrologic characteristics such as retention time, water depth, and HLR are vitally important to determine the treatment performance of a wetland system (Richardson and Nichols 1985; Mitsch and Gosselink 1993; Kadlec and Knight 1996). Lowering

hydraulic loading rates and increasing hydraulic retention times usually leads to greater nutrient removal efficiencies (Sakadevan and Bavor 1999). However, most recent research has focused on the system removal efficiency by comparing only inlet and outlet concentrations of contaminants. There is a lack of information on wetland hydrology and corresponding mass balance calculations for real industry-sized ICW systems. Therefore, the objectives of this study are the following:

- to quantify the effect of hydrologic characteristics on contaminant reduction efficiency of a surface flow ICW system;
- to assess annual and seasonal variations in hydraulic loading rate and nutrient removal efficiencies;
- to compare contaminant removal efficiencies calculated on the basis of both concentration and mass budgets; and
- to assess optimization options based on treatment results obtained.

Methods

Study Site

The researched ICW treatment system is located within the Castle Leslie Estate walls in County Monaghan (Ireland) at a longitude of 06°53'37.94" W and a latitude of 54°19'6.01" N. The site is surrounded by woodland and required sensitive development in terms of landscape fit, and biodiversity, amenity, and habitat enhancement. Mean seasonal temperatures for Monaghan in 2009 were as follows: spring (10.7°C), summer (14.9°C), fall (7.9°C), and winter (2.9°C). The mean annual rainfall is approximately 970 mm (Met Éireann 2010).

The ICW system was commissioned in October 2007 to treat combined sewage from the village of Glaslough. The aim was to improve the water quality of the Mountain Water River, which flows through the site. The design capacity of the ICW system is 1,750 population equivalents and covers a total area of 6.74 ha. The surface area of the constructed wetland cells is 3.25 ha in total. The maximum wastewater inflow rate was 613.06 m^3 day^{-1} (19 November 2009). The corresponding outflow was approximately between 0.02 m^3 day^{-1} (10 August 2009) and 475 m^3 day^{-1} (20 November 2009). These values are relatively low due to high evapotranspiration (between 0 and 262 m^3 day^{-1}) and infiltration (approximately 11 m^3 day^{-1}) rates. Untreated influent wastewater is pumped directly into a receiving sludge pond. The system contains two receiving sludge ponds that can be used alternately to allow for desludging without interrupting the process operation. The wastewater subsequently flows by gravity through five sequential unlined ponds, and the effluent of the last pond discharges to the adjacent Mountain Water River. The wetland cells were densely vegetated with *Carex riparia*, *Phragmites australis*, *Typha latifolia*, *Iris pseudacorus*, *Glyceria maxima*, *Glyceria fluitans*, *Juncus effusus*, *Sparganium erectum*, *Elisma natans*, and *Scirpus pendulus*.

Water and Hydrological Monitoring

A substantial suite of automatic sampling and monitoring instrumentation such as the ISCO 4700 Refrigerated Automatic Wastewater Sampler (Teledyne Isco, Inc., NE, USA) was applied for weekly surface water sampling. Moreover, all flows into, within, and out of the ICW system were measured and recorded with the computer-linked Siemens Electromagnetic Flow Meters F M MAGFLO and MAG5000 (Siemens Flow Instruments A/S, Nordborgrej, Nordborg, Demark). Mean flows were recorded at 1 min interval frequency. The locations of the electromagnetic meter are shown in Fig. 1. Eight lysimeters were installed before construction of the wetland ponds to give an 'upper-limit' indication of infiltration through the subsoil. A site investigation by the Geological Survey of Ireland (IGSL Ltd., Business Park, Naas, Country Kildare, Ireland) in September 2005 indicated a soil coefficient of permeability of 9×10^{-11} m s^{-1}. However, the mean infiltration obtained by lysimeters over a period of more than 2 years was 4×10^{-9} m s^{-1}, which was the value used for mass balance calculations. A weather station is located beside the inlet pump sump to measure air temperature, precipitation, and evaporation.

The net change in volume was calculated with Eq. 1.

$$D_s = Q_{in} + (P - ET - I) \times A - Q_{out} + Q_{runoff} + GW_{in}$$

(1)

where D_s = net change in volume (m^3 day^{-1}), Q_{in} = daily inflow to system (m^3 day^{-1}), P = daily precipitation rate (m day^{-1}), ET = evapotranspiration rate (m day^{-1}), I = infiltration rate (m day^{-1}), A = total surface area of five ponds (m^2), Q_{out} = daily outflow from system (m^3 day^{-1}), Q_{runoff} = catchments runoff (m^3 day^{-1}), and GW_{in} = groundwater inlet (m^3 day^{-1}).

Laboratory Sample Analysis

Water samples were analyzed for variables including temperature, pH, conductivity, redox potential, BOD, COD, TSS, pH, NH$_3$-N, NO$_3$-N, and MRP (equivalent to soluble reactive phosphorus) at the Monaghan County Council water laboratory using American Public Health Association (APHA 1998) standard methods, unless stated otherwise.

Fig. 1 Sketch of the integrated constructed wetland in Glaslough, Ireland. The sampling and monitoring points for various functions are indicated
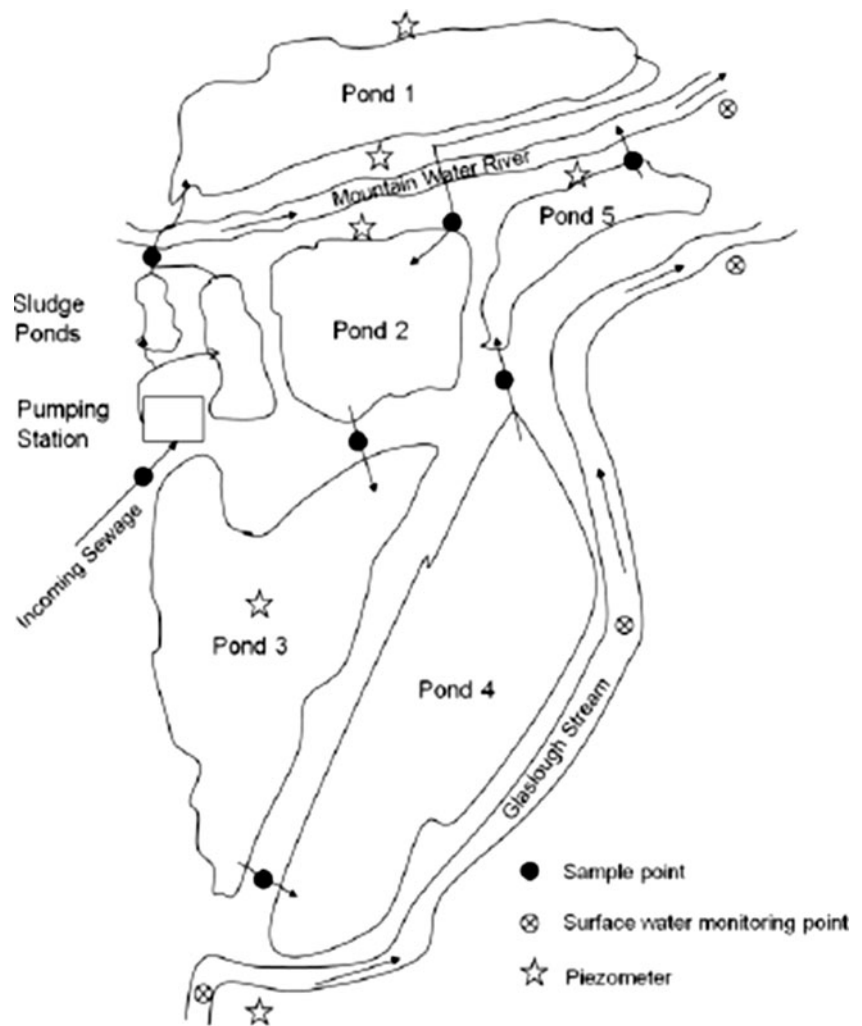
Mass Balance Calculations

The contaminant mass balance and the ICW hydrological cycle were calculated by using the dynamic water budget approach (Kadlec and Knight 1996; Kadlec and Reddy 2001) outlined in Eqs. 2 to 4. The evapotranspiration (ET) rate was calculated using Eq. 5. The HLR (cm day^{-1}) was calculated via Eq. 6.

$$L_{in} = \frac{Q_{in} \times C_{in}}{A} \qquad (2)$$

where L_{in} = inflow loading rate (g day^{-1}m^{-2}), Q_{in} = daily inflow to system (m^3 day^{-1}), C_{in} = inflow concentration of contaminant (g m^{-3}), and A = total surface area of five ponds (m^2).

$$L_{out} = \frac{Q_{out} \times C_{out}}{A} \qquad (3)$$

where L_{out} = outflow loading rate (g day^{-1}m^{-2}), Q_{out} = daily outflow from system (m^3 day^{-1}), C_{out} = outflow concentra-

tion of contaminant (g m^{-3}), and A = total surface area of five ponds (m^2).

$$Q_{actual} = Q_{in} + (P - ET - I) \times A \qquad (4)$$

where Q_{actual} = the actual flow within the wetland taking account of inflow, precipitation, evapotranspiration, and infiltration (m^3 day^{-1}), Q_{in} = daily inflow to system (m^3 day^{-1}), P = daily precipitation rate (m day^{-1}), ET = evapotranspiration rate (m day^{-1}), I = infiltration rate (m day^{-1}), and A = total surface area of five ponds (m^2).

$$ET = \sum_{i=1}^{n} E \times a_i \times r_i \qquad (5)$$

where ET = evapotranspiration rate (m day^{-1}), E = daily evaporation (m^3 day^{-1}), a_i = percentage of specific vegetation (%) as shown in Table 1, and r_i = ET/E ratio.

$$HLR = \frac{Q_{actual} \times 100}{A} \qquad (6)$$

Table 1 Plant evapotranspiration over evaporation (ET/E) ratio and distribution of plants within the integrated constructed wetland ponds

Vegetations	ET/E ratio	Pond 1	Pond 2	Pond 3	Pond 4	Pond 5
Carex riparia Curtis	1.6[a]	75%		15%	20%	
Elisma natans (L.) Raf	-					
Glyceria fluitans (L.) R. Br	1.75[b]				15%	
Glyceria maxima (Hartm.)	1.75[b]	20%	80%	5%	25%	85%
Iris pseudacorus L.	1.75			20%		
Juncus effusus L.	1.52[a]		7%	15%	15%	
Phragmites australis (Cav.)	1.5[b]			10%		
Scirpus pendulus Muhl	1.9[c]		3%			
Sparganium erectum L. emend Rchb	-					
Typha latifolia L	1.75[c]	5%	10%	35%	25%	
Open water	1.0					15%

[a] Vymazal (1999)
[b] Herburst and Kappen (1999)
[c] Snyder and Boyd (1987)

Statistical Analyses

Statistical analyses were conducted using the software package Origin 8.0 and IBM SPSS Statistics 19. Correlations between temperature, evapotranspiration, precipitation and actual flow were analyzed. Removal efficiencies among seasons were compared using a one-way analysis of variance (ANOVA). Post hoc comparisons using Tukey's Honestly Significant Differences Tests were applied to identify significant differences among means at the 5% probability level ($\alpha = 0.05$). A regression analysis between seasonal hydraulic loading rate and mass removal efficiency was also performed.

Results and Discussion

Influent and Effluent Water Quality

The mean influent and effluent concentrations and removal efficiencies of the water quality variables are presented in Table 2. The contaminant concentrations of influent wastewater varied significantly ($p < 0.05$) with respect to

seasons. The ICW demonstrated good contaminant removal performances removing (based on concentrations) approximately 92% COD, 98% BOD, 94% SS, 97% NH_3-N, 90% NO_3-N, 96% TN, and 96% MRP during the study period (May 2008 to April 2010).

Hydraulic Characteristics

The HRT, HLR, and mean daily flow rates for each pond, and the integrated system as a whole are presented in Table 3. In general, surface flows from the sludge ponds and precipitation were assumed to be the input, while evapotranspiration and water infiltration were assumed to be lost water. Precipitation and evapotranspiration were measured as the amount of water entering or evaporating from a wetland cell surface.

Water Budget

The net change in volume for the monitoring period was 62.5 ± 371.3 m^3 day^{-1}. This change in volume was calculated with the dynamic water budget equation (Eq. 1), which includes all possible water sources and sinks in the wetland

Table 2 Wastewater quality variables for the integrated constructed wetland in Glaslough (Ireland) between May 2008 and April 2010

Variable (mg l^{-1})	n[a]	Influent		Effluent		Reduction (%)
		Mean	SD[b]	Mean	SD[b]	
Biochemical oxygen demand	103	373.0	198.74	4.5	4.21	98.4
Chemical oxygen demand	110	610.0	306.63	37.7	25.09	92.3
Total suspended solids	83	236.8	133.51	9.5	19.44	94.2
Molybdate reactive phosphorus	81	4.0	1.86	0.1	0.23	95.8
Total phosphorus	95	7.0	3.16	0.2	0.22	96.5
Ammonia-nitrogen	109	38.1	12.96	0.9	1.92	96.9
Nitrate-nitrogen	96	5.1	3.90	0.3	0.21	89.6
Total nitrogen	102	43.4	14.23	1.2	1.99	96.4

[a] Sample number
[b] Standard deviation

Table 3 Dimensions, mean hydraulic retention time (HRT), hydraulic loading rate (HLR) and daily flow rate for the integrated constructed wetland (ICW) system in Glaslough (Ireland)

ICW section	Area (m^2)	Depth (m)	Volume (m^3)	HRT (d)	HLR (cm day^{-1})	n[a]	Inflow (m^3 day^{-1})		Outflow (m^3 day^{-1})	
							Mean	SD[b]	Mean	SD[b]
Pond 1	4,664	0.42	1,958.9	18	2.44	708	103.8	106.10	111.8	132.35
Pond 2	4,500	0.38	1,710.0	16	2.68	708	111.8	132.35	109.0	134.27
Pond 3	12,660	0.32	4051.2	32	1.07	708	109.0	134.27	128.0	158.48
Pond 4	9,170	0.36	3,301.2	23	1.61	708	128.0	158.48	143.2	191.56
Pond 5	1,460	0.29	423.4	3	10.03	656	143.2	191.56	131.0	179.43
Total system	32,454	n/a[c]	11,444.7	92	0.54	n/a[c]	103.8	106.10	131.0	179.43

[a] Sample number

[b] Standard deviation
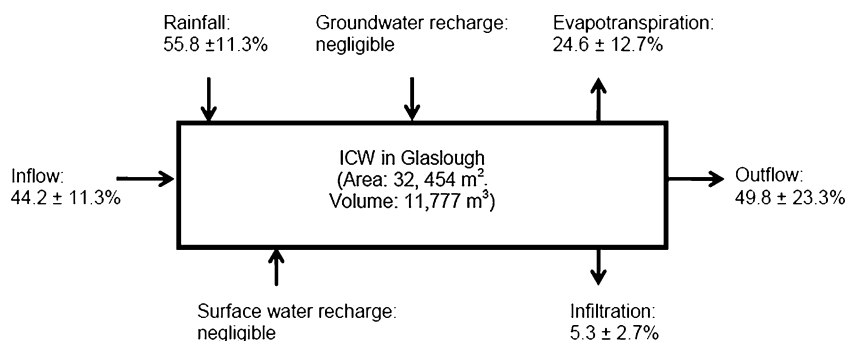
[c] Not applicable

such as rainfall, evapotranspiration, infiltration, wastewater inflow, and the outflow from wetland. The inflow to the ICW system originated mainly from precipitation, which contributed approximately 55.8% of the overall input. The percentages for the outflow of the system were as follows: evapotranspiration of 24.6±12.7%; infiltration of 5.3± 2.7%, and outflow of 49.8±23.3%. The remaining proportion represents the water within the wetland at any one time. Catchment runoff and groundwater inflow were assumed to be negligible (Fig. 2). A strong correlation ($R^2=0.97$) was observed between precipitation and actual flow, which indicates that precipitation is likely to have a considerable influence on the hydraulic loading rate.

Evapotranspiration is a very complex process. Figure 3 shows a low coefficient of determination ($R^2=0.41$) obtained between evapotranspiration and temperature, which suggests that other factors may be affecting evapotranspiration such as humidity, wind speed, dominant plant cover, and the moisture level within plants (Vymazal and Kröpfelová 2008). Seasonal variations in evapotranspiration data were significant during the course of the monitoring period. Higher evapotranspiration rates were observed during spring and summer. The mean humidity values for Glaslough in spring and summer of 2008 were

75.5% and 87.2% respectively. Considering the small differences in temperature (spring, 13.3°C; summer, 15.4°C) and wind speed (spring, 0.55 m s^{-1}; summer, 0.50 m s^{-1}), the decrease in evapotranspiration rate during summer was probably due to higher humidity conditions. In contrast to 2008, only a slight evapotranspiration difference was observed in spring and summer of 2009. Despite obvious temperature differences, humidity data were similar in both seasons (83.4% in spring and 85.6% in summer); the relatively high wind speed in spring (0.71 m s^{-1}) increased the evapotranspiration rate slightly. However, the actual water surface areas (partly covered by plants) were not measured, and their effect on evapotranspiration can therefore not be accurately analyzed.

Loading Rates and Removal Efficiencies

Optimal HLR and HRT are important to achieve good wastewater treatment results within wetland systems. Previous studies have indicated that the removal efficiencies of pollutants decreased significantly with HLR (Tanner et al. 1995a,b; Huang et al. 2000; Chung et al. 2008; Trang et al. 2010). At low HLR, the HRT is relative high. In comparison, at high HLR, the wastewater passes rapidly through the

Fig. 2 Water budget for the integrated constructed wetland system between May 2008 and April 2010

Rainfall: 55.8 ±11.3%

Groundwater recharge: negligible

Evapotranspiration: 24.6 ± 12.7%

Inflow: 44.2 ± 11.3%

ICW in Glaslough (Area: 32, 454 m^2. Volume: 11,777 m^3)

Outflow: 49.8 ± 23.3%

Surface water recharge: negligible
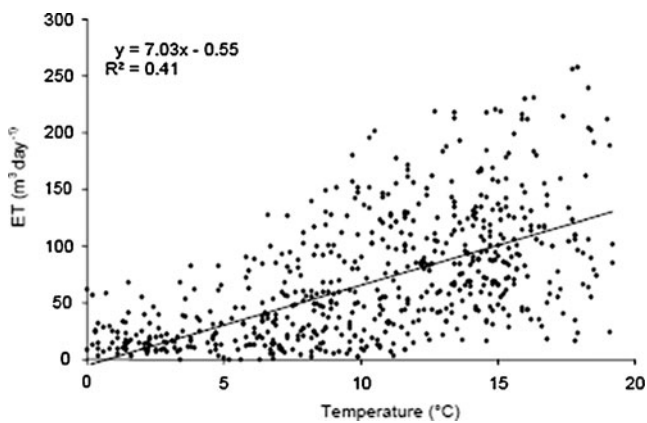
Infiltration: 5.3 ± 2.7%

Fig. 3 Evapotranspiration (ET) as a function of temperature

wetland, reducing the time available for degradation processes to become effective. However, the research study showed that the removal performance of the ICW varied slightly, and decreased only during fall of 2009, when some overland flow was recorded (Table 4). These observations can be explained by the large footprint of the system, which results in a relatively long HRT. Moreover, the system is relatively immature, which means the system has relatively high adsorption and storage capacities.

Over the 2-year period of this study, the mean influent and effluent NH_3-N loading rates were approximately 0.102 and 0.006 g $day^{-1}m^{-2}$ respectively. The mean removal efficiency was 91.7%. The surface inflows brought about a total load of 2349.79 kg NH_3-N received by the ICW system, and 95.5% of inputs were retained (Table 5). The retention rate was approximately 34.64 g NH_3-N m^{-2} $year^{-1}$. Seasonal hydraulic loading and removal efficiencies are presented in Fig. 4a. There was a significant decrease of NH_3-N removal during fall and winter of 2009. However, the overall removal efficiency of NH_3-N was comparatively high. Kayranli et al. (2010) have demonstrated that NH_3-N reduction within the Glaslough system is higher than that of other wetlands treating domestic wastewater, which is probably due to high plant uptake and an increased nitrification process caused by root-zone reaeration. In addition, Boutilier et al. (2010) found that treatment

efficiencies associated with surface-flow domestic wastewater treatment wetlands decreased during winter. This might be caused by ice cover and lack of flow, which may have created anaerobic conditions, thereby decreasing biodegradation.

The mean inflow loading rate for NO_3-N was approximately 0.014 g $day^{-1}m^{-2}$. A reduction by 83.6% due to treatment within the wetland led to an outflow rate of roughly 0.001 g $day^{-1}m^{-2}$. The surface inflows carried a total load of 337.76 kg NO_3-N into the system within the 2-year operation period (Table 5). About 93% of the inputs were retained; thus yielding an average retention of 4.83 g NO_3-N m^{-2} $year^{-1}$. Figure 4a shows the variations of the average seasonal hydraulic loading rates, and the corresponding NO_3-N removal efficiencies. Nitrate was effectively removed from the influent over the study period, except during fall of 2009, when some flooding events were recorded. The increased hydraulic loading rate and reduced retention time contributed to the reduced NO_3-N removal performance during floods.

The major mechanisms of phosphorus removal within ICW systems are adsorption to the aggregates, precipitation, and assimilation into microbial and plant biomass (Trang et al. 2010). The MRP inlet loading rate varied slightly during the study period. Seasonal loading flucuated between 0.010 g $day^{-1}m^{-2}$ (spring 2008) and 0.013 g $day^{-1}m^{-2}$ (summer 2009). A total load of 270.13 kg MRP was carried into the system, and 94.1% of the inputs were retained (Table 5). The mean retention rate over the study period was 3.92 g MRP m^{-2} $year^{-1}$. The MRP mass removal efficiency varied between 62.5% (fall 2009) and 99.5% (spring 2008). The low removal efficiency was likely due to the high hydraulic outflow rate, which consequently decreased the retention time. Despite that, no violation of the threshold value for MRP (1 mg l^{-1}, EU 1991) was observed during the monitoring period.

The removal of COD, BOD and TSS was generally efficient, and decreased only slightly at relatively higher HLR. The mean inlet and outlet COD loading rates were 1.7 ± 1.09 and 0.1 ± 0.11 g $day^{-1}\cdot m^{-2}$ respectively. The mass removal efficiency of the system was 91.8%. Seasonal

Table 4 The regression analysis (R^2) between seasonal hydraulic loading rate and mass removal efficiency

|		Spring 08	Summer 08	Fall 08	Winter 08	Spring 09	Summer 09	Fall 09	Winter09
Molybdate reactive phosphorus	R^2	0.25	0.21	5.01×10^{-5}	0.03	0.20	4.88×10^{-3}	4.97×10^{-4}	0.70
Ammonia-nitrogen	R^2	0.05	0.19	0.05	6.19×10^{-3}	0.38	1.84×10^{-3}	6.79×10^{-4}	0.21
Nitrate-nitrogen	R^2	0.07	0.26	0.09	0.23	8.24×10^{-6}	1.84×10^{-3}	5.35×10^{-3}	0.04
Biochemical oxygen demand	R^2	4.23×10^{-5}	0.25	0.02	0.02	0.02	0.10	0.08	1.43×10^{-3}
Chemical oxygen demand	R^2	3.84×10^{-3}	0.47	0.17	0.03	3.12×10^{-3}	0.01	0.14	0.14
Total suspended solids	R^2	3.49×10^{-3}	0.21	0.51	0.48	1.34×10^{-3}	7.96×10^{-4}	0.28	1.00

Table 5 Total mass loading rates for water quality variables during the monitoring period (May 2008 to April 2010)

Variable (mg l⁻¹)	Loadings (kg)		Mass retained (%)
	Inlet	Outlet	
Biochemical oxygen demand	25,308.31	250.91	99.0
Chemical oxygen demand	42,652.55	2,374.76	94.4
Total suspended solids	15,895.37	432.08	97.3
Molybdate reactive phosphorus	270.13	15.94	94.1
Total phosphorus	452.34	17.48	96.1
Ammonia-nitrogen	2,349.79	101.37	95.7
Nitrate-nitrogen	337.76	24.24	92.8
Total nitrogen	2,712.92	123.85	95.4

hydraulic loading and removal efficiency data are provided in Fig. 4b. The high COD removal performance might be due to good growth of vegetation, resulting in high concentrations of dissolved oxygen, which can be seen as the electron acceptor for heterotrophs attached to the rhizomes (Avsar et al. 2007). Previous studies show that COD removal within wetland systems depends also on vegetation type and water level, because of root oxygen and

Fig. 4 Seasonal means of a) nutrient mass removal efficiencies and the hydraulic loading rate, and b) biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total dissolved solids (TSS) mass removal efficiencies and the hydraulic loading rate
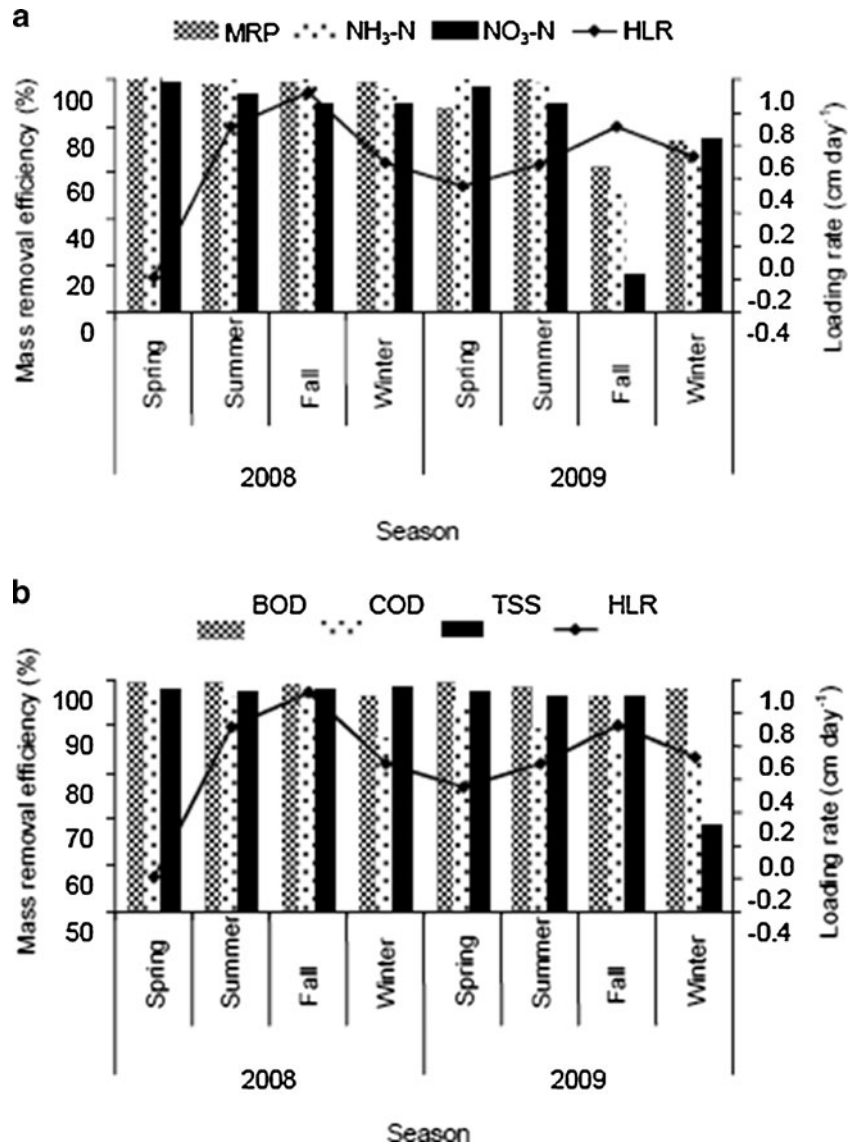
carbon release (Zhu and Sikora 1995; Stottmeister et al. 2003; Sun et al. 2009). The BOD removal rates ranged from 96.3% to 99.4%. The mean mass removal rate for TSS was 97.3%. The relatively lower BOD and TSS removal rates were recorded during fall and winter 2009, which was probably due to the relatively high hydraulic loading rate combined with low temperature.

Temperature and Removal Efficiencies

The low removal efficiencies recorded were partly a result of the adverse influence of low ambient temperature within the ICW system, reducing microbial activities and diffusion rates (Phipps and Cumpton 1994; Spieles and Mitsch 2000). For BOD and COD, the seasonal removal rates in fall 2009 were relatively low at 96.3% and 84.2% respectively (Table 6). This finding was in contrast to several other studies reporting slight influences of temperature on the removal efficiency of COD within constructed wetlands (Dahab et al. 2001; Vymazal 2001; Mæhlum and Jenssen 2003; Steinmann et al. 2003; Züst and Schönborn 2003). This low COD removal efficiency might be attributed to the absence of sufficient microorganisms attached to the rhizomes at the beginning of the wetland maturation process. Similarly, Table 6 shows that nitrogen losses were rather below expectations during the low temperature period: 50.7% for NH_4-N, and 49.8% for TN (fall, 2009). This shortcoming was probably due to the reductions in nitrification and denitrification rates at lower temperatures. In the winter of 2009/2010, the ambient temperature was below 0°C, and the pond water surface was actually frozen, which may have caused anaerobic conditions, leading to low ammonia degradation. Previous research shows that the biological removal of nitrogen is most efficient at temperatures ranging from 25°C to 30°C (Sutton et al. 1975; Herskowitz et al. 1987; Hammer and

Knight 1994; Vymazal 1999; Mitsch et al. 2000). Furthermore, in November 2009, a severe storm led to river flooding and subsequent discharge of flood water into pond 5. This resulted in a relatively higher HLR and an exceptionally high outflow rate, which might have shortened the HRT, resulting in low ammonia degradation. During colder periods, there is little plant uptake of phosphorus, which leads to a decline in MRP mass removal efficiency, as observed during fall and winter of 2009. However, the system is still relatively inmature, and high phosphorus adsorption and storage capacities are therefore to be expected.

Removal Efficiencies Calculated by Concentration and Mass

The removal efficiencies of water pollutants were calculated as the percent change in concentrations or mass loading rates from influent to effluent. In this study, the concentration removal efficiencies were relatively high (>78%) despite the seasonal variations that were recorded. In fact, clear differences were found between most removal rates based on concentrations compared to mass (winter 2009): BOD was 98.8 versus 97.8%; COD was 91.4 versus 84.3%; MRP was 87.4 versus 72.9%; TP was 88.4 versus 78.1%; NH_3-N was 81.4 versus 65.1%; TN was 82.6 versus 67.3%; TSS was 82.5 versus 68.7%. These figures clearly indicate that the efficiencies based upon only the inflow and outflow concentrations might prove misleading in any attempted assessment of wetland system performance (Vymazal and Kröpfelová 2008).

Conclusions

There was no obvious trend in the treatment performance (except for some seasonal variability) during the 2 years of

Table 6 Seasonal comparison nutrient mass removal efficiencies (%) for the integrated constructed wetland and corresponding results of the one-way analysis of variance (p value)

Variable	2008/2009				2009/2010				p value
	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	
Biochemical oxygen demand	99.37[a]	99.20[a]	98.92[a]	96.29[a]	99.19[a]	98.68[a]	96.29[a]	97.81[a]	1.77×10^{-3}[*]
Chemical oxygen demand	97.87[a]	96.15[a]	94.02[a]	87.12[a]	94.08[a]	89.54[a]	84.24[a]	84.27[a]	3.76×10^{-3}[*]
Total suspended solids	98.01[a]	97.69[a]	97.98[a]	98.46[a]	97.50[a]	96.34[a]	96.26[a]	68.67[b]	5.02×10^{-4}[**]
Ammonia-nitrogen	99.79[a]	99.29[a]	99.51[a]	94.99[a]	99.31[a]	98.71[a]	50.72[b]	65.11[ab]	1.37×10^{-4}[**]
Total nitrogen	99.56[a]	98.42[a]	98.78[a]	96.16[a]	99.07[a]	97.97[a]	49.76[b]	67.33[ab]	3.40×10^{-4}[**]
Molybdate reactive phosphorus	99.47[a]	97.38[a]	98.98[a]	97.30[a]	91.03[a]	99.48[a]	62.49[b]	72.90[ab]	1.58×10^{-4}[**]
Total phosphorus	99.40[a]	97.96[a]	99.06[a]	96.31[a]	98.15[a]	97.30[a]	71.73[b]	78.13[ab]	1.02×10^{-6}[**]

Different superscript letters (a and b) indicate significant differences based on Tukey's Honestly Significant Difference Post Hoc Tests ($\alpha = 0.05$).

[*] $p < 0.01$

[**] $p < 0.001$

observation. However, most previous studies found that increased contaminant load and reduced retention time contribute to reduced removal efficiency. Therefore, the overall hydraulic loading based on a system mass balance should be considered during the planning and early design stage.

Since the studied ICW system was covered by snow and plant debris in winter and subjected to inflow shortage, anaerobic conditions might have led to a decrease in biological activity and treatment efficiency. The removal of contaminants during fall and winter could be optimized further by increasing the hydraulic retention time and by enlarging the wetland system.

Dense stands of wetland plants contributed to relatively high evapotranspiration rates, which led indirectly to additional storage volume required during rainy periods. Further studies on the effects of plant diversity upon ICW removal efficiency and long term performance could be assessed to guide management team at the design stage. Moreover, investigating changes in microbial population influencing treatment performance with temperature would also be beneficial.

Overall, the ICW systems can be seen as an appropriate, robust, reliable and cost-saving technology which can be utilized anywhere in the world.

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