Assessment of an Integrated Constructed Wetland for Decentralized Wastewater Treatment in a Rural Community in Ireland

Mawuli Dzakpasu1,3,*, Miklas Scholz2, Valerie McCarthy3, Siobhán Jordan3
1 UCD Dooge Centre for Water Resources Research, School of Civil, Structural and Environmental Engineering, Newstead Building, University College Dublin, Belfield, Dublin 4, Ireland
2 Civil Engineering Research Group, School of Computing, Science and Engineering, Newton Building, The University of Salford, Greater Manchester M5 4WT, UK
3 Centre for Freshwater and Environmental Studies, North Building, Dundalk Institute of Technology, Dundalk, Co. Louth, Ireland

ABSTRACT
We examined the efficacy of a full-scale integrated constructed wetland (ICW) to remove bulk organic pollutants (BOD5, COD, TSS) and nitrogen (total N, NH4-N, NO3-N) from wastewater. The ICW system was designed to treat domestic wastewater from Glaslough village in Ireland at a loading rate of 1750 population equivalents. During the four-year period of operation, hydraulic loadings to the ICW ranged between 2.25-14.01 mm/d. Influent concentrations were high and variable for all parameters. The influent nitrogen (N) was mostly in ammonical form (60% of total N). Nitrogen removal was seasonally dependent, with greatest removal rates occurring during warm periods. Although significant removals of the N species were recorded, the ICW released dissolved inorganic N fractions. The release of NH4-N was greatest during cold periods (December-February), which was probably related to decreased water temperature, associated plant senescence and increased decomposition, and increased DO content of wetland waters. Nonetheless, long-term first order rate constants (k20) for removal were low. Annual k20 values ranged between 5.6-9.0 m/year for organic substances and 5.1-15.4 m/year for N species. ICW systems should be managed to maximize NH4-N removal in the long-term, while mitigating the release of inorganic N during cold periods, which requires dynamic wetland management.

Keywords: Domestic wastewater; hydraulic loading rate; constructed wetland; organic pollutant; nitrogen

1. INTRODUCTION
The ever-increasing human population and, therefore, in domestic wastewater generation has created a great challenge to develop and introduce simple, affordable, efficient and sustainable domestic wastewater treatment systems in many countries across the world. The efforts in providing these essential services especially for poorer regions of the world are hindered by financial limitations, and the shortcomings of the current concept of urban water management. Most of the conventional treatment technologies are rather expensive and complex (Grau, 1996; Verhoeven and Meuleman, 1999) for small communities, typically rural or sub-urban areas and scattered settlements.

Among the different wastewater treatment technologies now available worldwide,
engineered and constructed treatment wetland systems are increasingly gaining attention as the market for decentralized wastewater management expands in both developed and developing countries (Babatunde, 2010). This is because they are commonly perceived as cost-effective and environmentally friendly (Babatunde, 2010; Lee et al., 2009), and also, because they are easy to operate, require low maintenance and have low investment costs (Hammer et al., 1993; Scholz, 2011).

In Ireland, the ICW Initiative of the Irish Department of Environment, Heritage and Local Government has developed a specific design concept of constructed treatment wetlands, namely integrated constructed wetlands (ICW), over the last two decades. The concept of ICW was developed with an approach that endeavored to achieve ‘water treatment’, ‘landscape fit’ and ‘biodiversity enhancement’ targets by an innovative wetland design methodology. As described by Scholz et al. (2007), the concept of ICW employs the free water surface flow (FWS) constructed wetlands (CW) model and incorporates the concept of restoration ecology, specifically mimicking the structure and processes of natural wetlands. They are characterized by a multi-celled configuration with sequential through-flow, and are based on the holistic and interdisciplinary use of land to control water quality. Typically, ICW systems have shallow water depths (10-30 cm) and contain many plant species in a mixed culture. This can facilitate microbial and animal diversity (Jurado et al., 2010; Nygaard and Ejrnæs, 2009), and is generally aesthetically appealing, which enhances recreation and amenity values.

ICW systems, like natural wetlands are composed of a unique assemblage of water, vegetation, soils, and microbes. These components contribute to ICW systems typically being net sinks for nitrogen (N), phosphorus (P) and carbon (C) (Kayranli et al., 2010a; Scholz et al., 2007). However, they can also be sources of N, P and C by the release of C and nutrients upon reflooding after water level drawdown (Olila et al., 1997; Pant and Reddy, 2001). The net effect of whether a system is typically a sink or a source of nutrients often depends upon the dominant biogeochemical processes, the environmental conditions that influence process rates, and the associated timescale. If the dominant biogeochemical processes that govern both retention and release are known, ICW systems can be managed to provide suitable conditions for the desired processes. For example, the water level, flow, depth, and duration can be manipulated to provide aerobic conditions for nitrification and anaerobic conditions to facilitate denitrification.

The transformation and translocation of N in ICW systems can occur via various processes, which include sedimentation of particulates, mineralization, nitrification and denitrification, dissimilatory nitrate reduction to ammonium, microbial assimilation, plant uptake and release, anaerobic ammonium oxidation, resuspension, diffusion, sorption, and volatilization (Kadlec and Wallace, 2009). Environmental factors that influence process rates include but are not limited to temperature, pH, soil organic matter, water depth, water flow rate, alkalinity, dissolved oxygen, concentration gradients, and biota (Kadlec, 2010; Picard et al., 2005, Toth, 2010). Burgeoning studies have reported nitrification/denitrification dynamics and N budgets within wetlands from around the world. However, many previous studies assessing the performance and nutrient dynamics of ICW systems for domestic wastewater treatment have been limited in time (less than 2 years), and, therefore, have not provided insight into their long-term nutrients treatment capabilities. For example, Kayranli et al. (2010b) evaluated the performance of an ICW system treating domestic wastewater after only 1 year of operation. However, Kadlec and Knight (1996) noted that nutrients treatment in
wetland systems during the initial operating period (1-2 years) is likely enhanced due to higher availability of sorption sites and increased biological activity during the rapid expansion of biomass.

This paper evaluates a successful case of decentralized domestic wastewater treatment with ICW systems in a rural community in Ireland. The specific objectives were to (a) examine how the different N species and organic matter components in domestic wastewater change via ICW treatment over long-term system operation; and (b) evaluate the effect of operational and environmental factors like hydraulic and contaminant loadings, temperature, and seasonality on N and organic matter transformations and removal.

2. MATERIALS AND METHODS

2.1 Study site description

The ICW treatment system at the center of the study is located within the walls of Castle Leslie Estate at Glaslough in County Monaghan, Ireland (06°53’37.94” W, 54°19’6.01” N). The ICW (Fig. 1) comprises a small pumping station, two sedimentation ponds, and a sequence of five shallow and vegetated wetland cells. Hydraulic characteristics of the wetland cells are presented in Table 1. It was commissioned in October 2007 to treat combined sewage from Glaslough village and to improve the water quality of the Mountain Water River, which flows through the site. The sewerage collection system serving the village is a combined foul sewer, which transport excess storm water and sewage from households and commercial building within the village to the ICW system.

The design capacity of the ICW system is 1,750 p.e. The functional water area of the ICW cells is 3.3 ha, within acurtilage area of 6.7 ha. The wetland cells have no artificial lining. Excavated local soil material was used to construct the base of the wetland cells and compacted to a thickness of 500 mm to form a low permeability liner. A site investigation conducted in September 2005 indicated a soil coefficient of permeability of $9 \times 10^{-11}$ m/s (IGSL Ltd., Business Park, Naas, Co., Kildare, Ireland).

Influent primary domestic wastewater from the village is pumped directly into a receiving sedimentation pond. The system contains two sedimentation ponds that can be used alternately so that one can be desludged without interrupting the whole treatment process. The purpose of the sedimentation pond is to retain the suspended solids contained in the influent wastewater. In this way, the build-up of sludge in the wetland cells, which could otherwise decrease the capacity of the cells, is prevented. From the sedimentation pond, the wastewater subsequently flows by gravity sequentially through the five earthen-lined wetland cells, and the effluent of the last cell discharges directly into the adjacent Mountain Water River.

The wetland cells were planted in a club-pattern (Fig. 1) and the original ones were Carex riparia Curtis, Phragmites australis (Cav.) Trin. ex Steud., Typha latifolia L., Iris pseudacorus L., and Glyceria maxima (Hartm.) Holm. After four years operation the plant distribution has changed to include a complex mixture of Glyceria fluitans (L.) R. Br., Juncus effusus L., Sparganium erectum L. emend Rchb, Elisma natans (L.) Raf., and Scirpus pendulus Muhl.

2.2 Water sampling

Sampling points were fixed at the inlet of each ICW cell (Fig. 1). Composite influent water samples were collected during the monitoring period (February 2008-March 2012) by using a suite of automated sampling and monitoring
instrumentation, namely, the ISCO 4700 refrigerated automatic wastewater sampler (Teledyne Isco, Inc., NE, USA). Sampling was carried out at a 12-hour interval frequency, normally at 12:00 and 24:00, daily. A fixed sampling procedure was followed, which essentially involved taking duplicate water samples of ≈500 mL per each sampling event. All collected samples were stored in the refrigerated chambers of the auto-samplers at 2°C and then transported to the Monaghan County Council wastewater laboratory in Ireland for analysis.

Figure 1  Schematic diagram of ICW cells showing diversity of plants and locations of sampling points
Table 1  Dimensions of constructed ICW cells at Glaslough in Co. Monaghan, Ireland

<table>
<thead>
<tr>
<th>ICW section</th>
<th>Area (m²)</th>
<th>Depth (m)</th>
<th>Total volume (m³)</th>
<th>Effective volume (m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sedimentation pond 1</td>
<td>285</td>
<td>0.45</td>
<td>128.3</td>
<td>85.5</td>
</tr>
<tr>
<td>Sedimentation pond 2</td>
<td>365</td>
<td>0.45</td>
<td>164.3</td>
<td>109.5</td>
</tr>
<tr>
<td>Cell 1</td>
<td>4664</td>
<td>0.42</td>
<td>1958.9</td>
<td>1399.2</td>
</tr>
<tr>
<td>Cell 2</td>
<td>4500</td>
<td>0.38</td>
<td>1710.0</td>
<td>1350</td>
</tr>
<tr>
<td>Cell 3</td>
<td>12660</td>
<td>0.32</td>
<td>4051.2</td>
<td>3798</td>
</tr>
<tr>
<td>Cell 4</td>
<td>9170</td>
<td>0.36</td>
<td>3301.2</td>
<td>2751</td>
</tr>
<tr>
<td>Cell 5</td>
<td>1460</td>
<td>0.29</td>
<td>423.4</td>
<td>423.4</td>
</tr>
<tr>
<td>Total wetland</td>
<td>33104</td>
<td>-</td>
<td>11737.3</td>
<td>9916.6</td>
</tr>
</tbody>
</table>

2.3 Hydrological monitoring

2.3.1 ICW inflows and outflows

The volumes of water flowing into and out of each ICW cell were measured and recorded with the Siemens Electromagnetic Flow Meters FM MAGFLO and MAG 5000 (Siemens Flow Instruments A/S, Nordborgrej, Nordborg, Demark) and their allied computer-linked data loggers. Mean flows were recorded at one-minute intervals.

2.3.2 Meteorological data

A weather station (WeatherLink®, Davis Instruments Co., Hayward, CA, USA.) is located beside the inlet pump sump, which measures elements of the local weather including air temperature, precipitation and reference (potential) evapotranspiration ($ET_o$). All meteorological variables were measured and recorded at 15 minutes intervals. The WeatherLink® uses air temperature, relative humidity, average wind speed, and solar radiation data to estimate the $ET_o$.

2.4 Water quality analyses

All analyses were carried out at the Monaghan County Council wastewater laboratory in Ireland, using kits supplied by HACH Lange (HACH Company, Loveland, Co., USA), and by following the standard operating procedures for the HACH DR/2010 portable datalogging spectrophotometer.

2.4.1 Organic matter and suspended solids

Organic matter content of the water samples were analyzed as BOD₅ and COD. BOD₅ was determined by APHA (1998) Method 5210 B, using the respirometric (manometric) BOD OxiTop system (WTW GmbH, Weilheim, Germany). The reactor digestion method, followed by colorimetric analysis (HACH Method 8000), was used to measure COD. Total suspended solids were determined by the gravimetric method (APHA Method 2540 D).

2.4.2 Nitrogen compounds

NH₄-N and NO₃-N were determined by the Nessler method (HACH Method 8038) and the cadmium reduction method (HACH Method 8171), respectively. Low range TN (HACH Method 10071) and high range TN (HACH Method 10072) were determined following persulfate digestion, whereas TP was determined by the acid persulfate digestion Test ‘N Tube Method (HACH Method 8190).

2.4.3 Ancillary water quality

Ancillary water quality parameters such as water pH, temperature, oxidation-reduction potential (ORP), dissolved oxygen (DO), and
electrical conductivity (EC) were measured onsite directly in the ICW cells by using a WTW Multi 1970i portable multi parameter meter and WTW ProfiLine Cond 197i portable conductivity meter (WTW GmbH, Weilheim, Germany).

2.5 Loading and removal rates for bulk organic pollutants and nutrients

The mass loading for bulk organic pollutants and nutrients in the ICW cells were calculated by using the dynamic water budget approach (Kadlec and Knight, 1996; Kadlec and Reddy, 2001; Kadlec and Wallace, 2009):

\[ LRI = q \times C_i \]  

(1)

where

- \( C_i \) = Inflow concentration of contaminant in ICW cell (g/m\(^3\))
- \( LRI \) = Contaminant inflow loading rate (g/m\(^2\)/d)
- \( q \) = Hydraulic loading rate (m/d)

The hydraulic loading rate was calculated as follows:

\[ q = \frac{Q}{A} \]  

(2)

where

- \( A \) = Area of ICW cell surface (m\(^2\))
- \( Q \) = Effective volume flow rate in ICW cell (m\(^3\)/d)

The effective flow rate was calculated as follows:

\[ Q = Q_i + (P - ET - I)A \]  

(3)

where

- \( A \) = Total surface area for 5 ICW cells (m\(^2\))
- \( ET \) = Daily evapotranspiration rate (m/d)
- \( I \) = Daily infiltration rate (m/d)
- \( P \) = Daily precipitation rate (m/d)

Removal rates for bulk organic pollutants and nutrients were quantified using three common approaches for CWs. The first approach estimated the mass removal efficiency (%) as follows:

\[ \text{Removal efficiency} = \left( \frac{Q_i C_i - (Q_o C_o)}{Q_i C_i} \right) \times 100 \]  

(4)

The second approach estimated the areal removal rate (g/m\(^2\)/d) as follows:

\[ \text{Removal rate} = q \times (C_i - C_o) \]  

(5)

The third approach estimated the area-based first-order removal rate constants using the \( k-C^* \) model, assuming plug flow conditions:

\[ \ln \left( \frac{C_o - C^*}{C_i - C^*} \right) = -\frac{k(T)}{q} \]  

(6)

where

- \( C_i \) = Influent contaminant concentration (mg/L)
- \( C_o \) = Effluent contaminant concentration (mg/L)
- \( C^* \) = Background concentration of bulk pollutants and nutrients (mg/L)
- \( k \) = Areal first-order removal rate constant at temperature, T °C (mg/year)
- \( Q_i \) = Daily volumetric water inflow rate (m\(^3\)/d)
- \( Q_o \) = Daily volumetric water outflow rate (m\(^3\)/d)

The \( k_T \) values were normalised to 20°C (\( k_{20} \)) based on Eq. (7). The \( C^* \) used to calibrate the model for various contaminants are shown in Table 2.
Table 2  Approximate background concentrations for FWS wetlands

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Inlet (mg/L)</th>
<th>$C^*(mg/L)$</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOD</td>
<td>100 – 200</td>
<td>10</td>
</tr>
<tr>
<td>BOD</td>
<td>200+</td>
<td>20</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>-</td>
<td>0</td>
</tr>
<tr>
<td>NO$_x$-N</td>
<td>-</td>
<td>0</td>
</tr>
<tr>
<td>TN</td>
<td>-</td>
<td>1.5</td>
</tr>
</tbody>
</table>

Source: Kadlec (2009)

The effect of temperature on the areal first-order removal rate constants for bulk pollutants and nutrients was modeled using the modified Arrhenius relationship:

$$k(T) = k(20) \theta^{(T - 20)}$$  \hspace{1cm} (7)

Where

- $k(T)$ = First-order removal rate coefficient at temperature, T °C (m/year)
- $k(20)$ = First-order removal rate coefficient at 20°C (m/year)
- $T$ = Water temperature (°C)
- $\theta$ = Empirical temperature coefficient (dimensionless)

3. RESULTS AND DISCUSSION

3.1 Influent and effluent concentrations of organic matter and suspended solids

The total suspended solids (TSS) concentrations in the influent wastewater received by the ICW were generally over 1000 mg/L. Overall, the ICW system reduced TSS by 99%, and most of this was due to the sedimentation pond (Fig. 2a). During the period of monitoring, mean TSS was reduced from 1193 ± 267 mg/L, to 164 ± 18.8 mg/L in the sedimentation pond, and to 7 ± 1.11 mg/L in the ICW cells. The concentrations of BOD and COD were highly variable in the ICW as well. Mean concentrations within the influent wastewater received by the ICW were reduced from 799 ± 71.8 mg O$_2$/L and 950 ± 49.2 mg O$_2$/L, respectively, to 308 ± 16.1 mg O$_2$/L and 509 ± 30.8 mg O$_2$/L in the sedimentation pond, and to 5 ± 0.57 mg O$_2$/L and 39 ± 2.40 mg O$_2$/L in the ICW cells, respectively. Overall, the ICW system reduced BOD and COD by 99% and 96%, respectively, and most of the removals were recorded in the sedimentation pond and cell 1 (Fig. 2a).

The influent wastewater had a strong TSS concentration maximum in summer, reaching over 5000 mg/L during this period (Fig. 3a). In contrast, the TSS concentrations in the wastewater inflowing to the ICW from the sedimentation pond were generally stable for the period of monitoring, with no strong seasonal pattern, although slightly lower concentrations were recorded in winter than the other seasons. However, the TSS concen-
trations at the effluent from the ICW system were slightly lower during winter, with higher TSS leaving the system in summer than in winter. Similarly, the influent wastewater had a strong BOD and COD concentration maximum in summer, typically reaching a mean of 982 mg O$_2$/L and 1095 mg O$_2$/L respectively, during the monitoring period (Fig. 3b, c). The lowest average BOD concentration was recorded in spring, whereas that for COD was in autumn, typically reaching a mean of 646 mg O$_2$/L and 861 mg O$_2$/L respectively.

The BOD concentrations in the wastewater inflowing to the ICW from the sedimentation pond were largely stable for the period of monitoring, and had a fairly strong seasonal pattern. Highest concentrations were recorded in summer (typical mean of 408 mg O$_2$/L) and the lowest were in winter (typical mean of 212 mg O$_2$/L). The COD concentrations in the wastewater inflowing to the ICW from the sedimentation pond followed a similar pattern, with a strong summer maximum (typical mean of 646 mg O$_2$/L) and a winter minimum (typical mean of 362 mg O$_2$/L).

The BOD concentrations at the effluent from the ICW system were slightly lower during winter, with higher TSS leaving the system in summer than in winter. Similarly, the influent wastewater had a strong BOD and COD concentration maximum in summer, typically reaching a mean of 982 mg O$_2$/L and 1095 mg O$_2$/L respectively, during the monitoring period (Fig. 3b, c). The lowest average BOD concentration was recorded in spring, whereas that for COD was in autumn, typically reaching a mean of 646 mg O$_2$/L and 861 mg O$_2$/L respectively.

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3.2 Influent and effluent nitrogen concentrations

Overall, NH$_4$-N was recorded as the dominant species of N contained in the influent wastewater received by the ICW. Annual influent concentrations (mean ± SE) of 62 ± 3.54 mg N/L, 37 ± 1.60 mg N/L and 5 ± 0.57 mg N/L were recorded respectively for TN, NH$_4$-N and NO$_3$-N, indicating a slight variability of the influent domestic wastewater. The NH$_4$-N concentrations received by the ICW over the study period were slightly higher than other FWS wetlands receiving primary domestic effluent, reported by Kadlec and Wallace (2009) and Boutilier et al. (2010) or in various types of CWs reported by Vymazal (2007), where concentrations varied depending on climate. Other studies such as Ran et al. (2004) have reported slightly higher influent concentrations as well. Individual values of N concentrations in the ICW effluent were consistently less than 1.0 mg/L and recorded an annual mean of 3.3 ± 0.43 mg/L and 2.2 ± 0.45 mg/L respectively for TN and NH$_4$-N, and 0.5 ± 0.05 mg/L for NO$_3$-N. The effluent concentrations of the N species were significantly lower ($p<0.01$, n = 181) than the influent.

During the period of monitoring, mean NH$_4$-N within the influent wastewater received by the ICW generally increased slightly from 37 ± 1.60 mg N/L to 42 ± 1.94 mg N/L in the sedimentation pond (Fig. 2b), possibly due to mineralization of organic constituents. Overall, the ICW system reduced TN and NH$_4$-N by 95 % and NO$_3$-N by 86 %. Most of the removal was recorded in cell 1 and cell 2 (Fig. 2b). Furthermore, influent concentrations of the N species showed some seasonal variations (Fig. 3d-f). Nevertheless, whereas the variations in concentrations of the influent NO$_3$-N were significant ($p<0.01$), variations of the influent TN and NH$_4$-N were not. The highest seasonal influent concentrations of TN and NH$_4$-N (69 ± 26.3 mg N/L and 42 ± 10.0 mg N/L respectively) and NO$_3$-N (8 ± 5.9 mg N/L) were recorded in summer and spring respectively (Fig. 3d-f). The highest removal rate for TN and NH$_4$-N occurred during summer and that for NO$_3$-N occurred during spring.
Figure 2  Profiles of (a) organic matter, suspended solids (b) nitrogen concentrations within individual cells of the ICW system (Vertical bars are standard errors of the mean, n=47-50 months from February 2008 to March 2012)

Figure 3  Variations in (a) SS (b) BOD₅ (c) COD (d) Total N (e) NH₄-N (f) NO₃-N concentrations within the ICW (Data points are monthly means from Feb. 2008 to Mar. 2012)

The effluent TN and NH₄-N concentrations were significantly higher during winter compared to the other seasons (p<0.05). No significant seasonal variations in concentrations of the effluent NO₃-N was observed, and was typically in the region of 0.3 mg N/L. The effluent TN and NH₄-N concentrations were highest during winter probably because of
increased surface outflow rates (Dunne et al., 2005a; Dzakpasu et al., 2015; Kadlec and Knight, 1996) caused by increases in precipitation driven hydrological inputs, which subsequently decreased HRT. For example, relatively high effluent NH$_4$-N concentrations were recorded when the outflow rate exceeded 200 m$^3$/d. Other possible explanations for this increase may include vegetation senescence and subsequent nutrient release from vegetation to the overlying wetland water column during this period (Kadlec, 2003). The decay of vegetation during cold months may release organic N back into the system, which are then available for ammonification (Kadlec and Wallace, 2009). Additionally, ice cover during the relatively severe winter in late December 2009 through early January 2010, as well as in mid-November 2010 through early March 2011 may have created anaerobic conditions and decreased biodegradation (Boutilier et al., 2010), and may also partly account for the increased effluent NH$_4$-N and TN concentrations.

3.3 Pollutant loadings and removal rates

The mass removal (based on 5 ICW cells) of the organic constituents (COD, BOD and TSS) and nitrogen (total N, NH$_4$-N and NO$_3$-N) was generally efficient in all cells of the ICW. The mean inlet and outlet COD loading rates were 3.2 ± 0.27, and 0.2 ± 0.03 g/m$^2$/d, respectively. The mean annual mass removal efficiency of the system was 91.9%. The surface inflows carried a total load of 161, 691 kg of COD into the system within the period of monitoring and a retention rate of 35, 652 kg/year was recorded (Table 3). The high COD removal performance may be attributed to good growth of vegetation, resulting in root zone oxygen release, which is used as the electron acceptor for heterotrophic bacteria attached to the rhizomes (Avsar et al., 2007). Previous studies have shown that COD removal within wetland systems largely depends on vegetation type and water level, due to root zone oxygen and carbon release (Su et al., 2009; Stottmeister et al., 2003).

The BOD removal rates ranged from 91.8% to 100%. The mean inflow loading rate was approximately 2.0 ± 0.14 g m$^2$/d. A mean annual reduction by 98.3% due to treatment within the wetland led to an outflow rate of approximately 0.02 ± 0.004 g m$^2$/d. The surface inflows carried a total load of 94, 939 kg BOD into the system within the period of monitoring. About 98% of the inputs were retained; thus yielding an average retention of 22, 387 kg BOD/year (Table 3). The mean annual mass removal rate for TSS was 96.4%. A total load of 49, 393 kg TSS was carried into the system and 11, 423 kg/year were retained.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Loading rates (kg/year)</th>
<th>Total mass retained</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Influent</td>
<td>Effluent</td>
</tr>
<tr>
<td>Biochemical oxygen demand</td>
<td>22,785</td>
<td>398</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>38,806</td>
<td>3154</td>
</tr>
<tr>
<td>Total suspended solids</td>
<td>11,854</td>
<td>432</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>4,205</td>
<td>222</td>
</tr>
<tr>
<td>Ammonia-nitrogen</td>
<td>3,124</td>
<td>165</td>
</tr>
<tr>
<td>Nitrate-nitrogen</td>
<td>285</td>
<td>31</td>
</tr>
</tbody>
</table>
For nitrogen, the average (±SE) areal NH$_4$-N loading rate (0.26 ± 0.02 g/m$^2$/d) was higher compared to that of NO$_3$-N (0.024 ± 0.004 g/m$^2$/d). Nevertheless, the areal removal rates for the two N species were consistently high, with average (±SE) of 0.25 ± 0.02 g/m$^2$/d for NH$_4$-N and 0.021 ± 0.004 g/m$^2$/d for NO$_3$-N. The influent and effluent loading for TN were 0.39 ± 0.02 g/m$^2$/d and 0.36 ± 0.02 g/m$^2$/d, respectively. In general, average annual mass removal efficiencies were relatively high for the ICW. Approximately 95% removal was recorded for both TN and NH$_4$-N, whereas 89.1% was recorded for NO$_3$-N. Over the study period, surface inflows carried a total load of 17,520 kg and 13,017 kg of TN and NH$_4$-N respectively, into the ICW system, and 3,983 kg/year and 2,959 kg/year respectively, were retained (Table 3). Similarly, a total load of 1,191 kg NO$_3$-N had been received by the ICW and about 254 kg/year retention had been recorded. Overall, nitrogen was effectively removed from the influent wastewater throughout the study period.

The annual mass removal rate of total N in the ICW system was found to be similar to the average mass removal rate of 126 g N/m$^2$/year reported for 17 FWS wetlands (Kadlec and Knight, 1996; Knight et al., 1993). Moreover, the median net removal rate for 116 FWS wetlands receiving more than 5 mg/L TN was reported as 129 g/m$^2$/year and 2,959 kg/year respectively, were retained (Table 3). Similarly, a total load of 1,191 kg NO$_3$-N had been received by the ICW and about 254 kg/year retention had been recorded. Overall, nitrogen was effectively removed from the influent wastewater throughout the study period.

The influence of the mass loading of organic constituents (BOD, COD and TSS) and nitrogen (total N, NH$_4$-N and NO$_3$-N) on their removal rates in the ICW system were evaluated by using linear regression. Overall, the mass loading rates strongly correlated with the mass removal rates, and showed significant linear relationships for organics and nitrogen constituents ($p < 0.01$). The regression coefficients were high ($R^2 = 0.99$ for all organic matter parameters and $R^2 = 0.97 - 0.99$ for the nitrogen constituents; Fig. 4a-f). This indicated that the mass removal of bulk organic pollutants and N by the ICW was dependent on input loading rates, with significantly higher removal rates at higher loadings. Tao et al. (2006) indicated that higher organic loading rates are desirable for increasing microbial production and bacterial activity. The close fit of the data points to the regression line also indicates a relatively constant areal removal rate. It can therefore, be inferred that surface area of ICW cells plays an important role in the removal of N from the wastewater. Furthermore, OLR had significant influence on the mass removal of N in the ICW. Generally, the N removal rate tended to increase significantly with increasing OLR. Nevertheless, no correlation was found between the N loading rate
and $C_o$. In the broad context, multiple datasets are represented by trends that show decreasing $C_o$ with decreasing LRI (Kadlec, 2009). Therefore, the optimal loadings to achieve the required discharge standards could not be estimated.

It has been noted that loading charts can be used to determine the maximum organic load carrying capacity of a wetland system (Kadlec, 2009; Kadlec and Wallace, 2009). For both batch and continuous flow type systems, the recommended maximum organic loading rates are 112 kg BOD/ha/d (USEPA, 1988), 50-80 kg BOD/ha/d (Knight et al., 1993), 6 g BOD/m²/d (Healy et al., 2007; USEPA, 2000) or 5 g TSS/m²/d (USEPA, 2000). In this study, a linear correlation of loading versus removal rates was sustained at maximum applied organic loadings of 43 kg BOD/ha/d, 81 kg COD/ha/d and 43 kg TSS/ha/d. This suggests that the ICW system can still be operated beyond the maximum ranges applied since relatively high rates of organic load removal was achieved at all times of the year. Removal rates greater than 90% were achieved and the effluent concentrations were consistently less than 5 mg/L for BOD and TSS.

![Graphs of observed mass loading against removal rates for bulk organic pollutants and nutrients within the ICW system](image)

**Figure 4** Plots of observed mass loading against removal rates for bulk organic pollutants and nutrients within the ICW system (All correlations are significant ($p < 0.01$, $n = 46-50$ months))
3.5 Effect of hydraulic loading rate on mass removal rate

Optimal HLR is an important factor for achieving effective wastewater treatment within treatment wetlands. In FWS wetlands, HLR and water velocity can have extreme variations and occasionally be very high (Kadlec and Wallace, 2009). Previous studies have indicated that pollutants removal efficiencies tend to decrease significantly with HLR (Chung et al., 2008; Huang et al., 2000; Tanner et al., 1995a, b; Trang et al., 2010). This is because at low HLR, the HRT is relatively longer. In comparison, at higher HLR, the wastewater passes rapidly through the wetland, and consequently, reducing the time available for microbial degradation processes or plant uptake to occur effectively. Other studies have shown that at occasional high HRLs there can be larger outflows of N than inflows (Spieles and Mitsch, 2000). In this study, the influence of HLR on the mass removal of bulk pollutants and nitrogen was evaluated by plotting a linear regression of mean monthly HLR and the corresponding mass removal rate of BOD, COD, TSS, total N, NH$_4$-N and NO$_3$-N in the ICW system (Table 4).

Linear relationships between the HLR and mass removal efficiency were not significant for all the parameters considered ($p > 0.05$). In addition, the regression coefficients were rather low (Table 4). Nevertheless, the removal efficiencies showed a weak negative response as the HLR increased from 2.25 mm/d to 14.01 mm/d, which generally tended to decrease slightly with increasing HLR. On the other hand, mass removal rates showed a weak positive significant ($p < 0.01$, $R^2 = 0.26-0.60$) linear correlation with the HLR (Table 4). This suggested that the overall removal of bulk pollutants and nitrogen in the ICW could be independent of the hydraulic loading rate. This effect could be explained by the relatively larger footprint of the ICW system, which serves to provide a buffer and maximizes the distance over which the influent must travel, and consequently, allowing for a longer HRT (Harrington and Ryder, 2002).

Furthermore, inlet and outlet mass balance studies have shown that higher HLRs result in higher area-specific N removals (Kadlec, 2005). In this study, mass removal rates in the ICW system showed a positive significant ($p < 0.01$) but weak correlation with the HLR (Table 4), suggesting that higher area-specific N removals were achieved at higher HLRs. However, Arheimer and Wittgren (2002) found that area-specific NO$_3$-N removal was negligible at very high HLRs.

Table 4 Linear regression analyses showing the effects of observed hydraulic loading rates on the mass removal of bulk pollutants and nutrients within the ICW

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Removal efficiency (%)</th>
<th>Removal rate (g/m$^2$/d)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$R^2$</td>
<td>$p$-value</td>
</tr>
<tr>
<td>Biochemical oxygen demand</td>
<td>1.8 × 10$^{-2}$</td>
<td>0.399</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>2.2 × 10$^{-2}$</td>
<td>0.351</td>
</tr>
<tr>
<td>Total suspended solids</td>
<td>2.7 × 10$^{-3}$</td>
<td>0.755</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>6.7 × 10$^{-3}$</td>
<td>0.636</td>
</tr>
<tr>
<td>Ammonia-nitrogen</td>
<td>2.2 × 10$^{-5}$</td>
<td>0.977</td>
</tr>
<tr>
<td>Nitrate-nitrogen</td>
<td>5.2 × 10$^{-4}$</td>
<td>0.888</td>
</tr>
</tbody>
</table>
3.6 Seasonal variation in ICW performance

The removal of bulk pollutants and nitrogen in the ICW was influenced by seasonality, where slightly lower removal was recorded during winter months (Fig. 5). A one-factor analysis variance indicated that COD mass removal rates were significantly affected by seasonal period ($p < 0.05$). On the other hand, variation of BOD and TSS mass removal rates were not significantly different during the different seasons (Fig. 5a). Total N, NH$_4$-N and NO$_3$-N mass removal rates were significantly affected by seasonal period ($p < 0.05$), whereby the ICW achieved average of approximately 15% higher removal of TN and NO$_3$-N, and 24% higher removal of NH$_4$-N during summer compared with winter periods (Fig. 5b).

The reduction in removal efficiencies recorded could be partly attributed to the adverse influence of low ambient temperature within the ICW system (Fig. 6), which probably reduced microbial activities and oxygen diffusion rates (Dong et al., 2011; Gao and Hu, 2012; Stein et al., 2003; Taylor, 2008). For COD, the seasonal removal rates in autumn and winter were relatively low at 90.9% and 80.9%, respectively (Fig. 5a). This finding was in agreement with several other studies, which reported a slight influence of temperature on the removal efficiency of COD within CWs (Gao and Hu, 2012; Kadlec at al., 2003; Stein et al., 2003; Taylor, 2008). In addition, this reduction in COD removal efficiency might be attributed to the absence of sufficient aerobic bacteria attached to plant rhizomes during this period, when plant senescence was at a peak. There is a marked reduction in oxygen diffusion rates during such periods, where the remaining dissolved oxygen is consumed rapidly by aerobic bacteria and fungi in the process of remineralising plant parts. Rapid decomposition causes further oxygen depletion, which creates a compounded negative effect as aerobic bacteria die out because of lack of oxygen. The ideal conditions for anaerobic bacteria are nonetheless, created albeit, anaerobic degradation of organic matter is much slower.

![Figure 5](image_url)  
**Figure 5** Mean seasonal mass removal efficiency of (a) bulk organic pollutants and (b) nitrogen within the ICW system (Vertical bars are standard errors of the mean. Small letters indicate seasonal differences in mass removal rates. Means that do not share a common letter are significantly different ($p< 0.05$, $n = 11-14$ months from February 2008 to March 2012))
Figure 6  Monthly average temperatures of ICW inflow and outflow water in relation to air temperature (Temperatures are monthly averages for the monitoring period February 2008 to March 2012, n = 51)

Table 5  Average annual area-based first-order rate coefficients for bulk pollutants and nutrients removal within the ICW system

<table>
<thead>
<tr>
<th>Parameter</th>
<th>$k_T \ (\text{m/year})$</th>
<th>$k_{20} \ (\text{m/year})$</th>
<th>$\Theta$ ($\theta$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biochemical oxygen demand</td>
<td>10.0</td>
<td>9.0</td>
<td>0.991</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>5.9</td>
<td>5.6</td>
<td>0.994</td>
</tr>
<tr>
<td>Total suspended solids</td>
<td>8.6</td>
<td>6.3</td>
<td>0.969</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>7.8</td>
<td>11.4</td>
<td>1.046</td>
</tr>
<tr>
<td>Ammonia-nitrogen</td>
<td>9.3</td>
<td>15.4</td>
<td>1.064</td>
</tr>
<tr>
<td>Nitrate-nitrogen</td>
<td>4.9</td>
<td>5.1</td>
<td>1.004</td>
</tr>
</tbody>
</table>

The ICW cell surface was frozen from late December 2009 through early January 2010 and in mid-November 2010 through early March 2011, which may have created anaerobic conditions and decreased biodegradation (Boutilier et al., 2010). The decay of vegetation during cold months may also release organic N back into the system, which is then available for mineralisation (Kadlec and Wallace, 2009). In addition, short HRT may affect N removals within CWs. The increased HLR as a consequence of excessive rainfall recorded during winter periods might have reduced the HRT in the ICW and contributed to the reduced N removal performance during that period. This phenomenon has been observed in previous studies, which have indicated that pollutants removal efficiencies in CWs decreased significantly with HLR (Chung et al., 2008; Huang et al., 2000; Tanner et al., 1995a, b; Trang et al., 2010).

3.7 First-order rate coefficients

The removal rates of bulk pollutants and nitrogen in the ICW system varied slightly over the period of monitoring. Such variations are often partly attributable to the effect of water temperature on the microbial processes involved (Kadlec and Wallace, 2009). In order to assess the effects of temperature on bulk...
pollutants removal within the ICW, the first-order rate coefficients \( (k_T) \) were estimated by using the \( k-C^* \) model. Generally, the seasonal variations in wetland effluent constituent concentrations may be sufficiently described by the \( k-C^* \) model (Kadlec, 2009). The effects of temperature on \( k_T \) were then evaluated by using the modified Arrhenius equation. Overall, the \( k_T \) for bulk organic pollutants removal in the ICW (Table 5) were quite low and ranged from 5.9 m/year for COD to 10.0 m/year for BOD. The \( k_T \) values were at the low end of the range (at the 5th percentile point) for 63 other FWS wetlands treating primary wastewater sources reported by Kadlec and Wallace (2009). The implications of the low \( k_T \) values were that the high removal rates for bulk pollutants recorded in the ICW might be achieved at lower HLR.

The \( k_T \) calculated for total N, \( \text{NH}_4-N \) and \( \text{NO}_3-N \) reduction in the ICW were 7.8 m/year, 9.3 m/year and 4.9 m/year, respectively (Table 5). The average effects of temperature (\( \theta \)) on the N removal rate constants were estimated to be 1.046 for total N, 1.064 for \( \text{NH}_4-N \) and 1.004 for \( \text{NO}_3-N \). This yielded normalised N removal rate constants at 20˚C \( (k_{20}) \) of 11.4 m/year, 15.4 m/year and 5.1 m/year, respectively. The N removal rate constants estimated for the ICW were similar to typical values reported for FWS wetlands (Kadlec, 2009; Kadlec and Wallace, 2009). The rate coefficients were at the low to middle end of the range (TN and \( \text{NH}_4-N \) were at the 40th and 30th percentile points of 116 and 118 other FWS wetlands, respectively, whereas \( \text{NO}_3-N \) was below the 10th percentile point of 72 other FWS wetlands) for other comparison FWS wetlands reported by Kadlec and Wallace (2009).

When normalised, the rate coefficients at 20˚C \( k_{20} \) decreased slightly for BOD, COD and TSS. This reduction in the rate coefficients indicated that temperature did not strongly influence the removal of bulk organic pollutants in the ICW system. However, the temperature coefficients (\( \theta \)) were slightly less than unity for BOD, COD and TSS, and suggested a slightly reduced removal rate at higher temperatures (Kadlec, 2009; Kadlec and Wallace, 2009). The \( \theta \) values recorded in the ICW were similar to the average values of 0.985 ± 0.021 for 19 other FWS wetlands reported by Kadlec (2009) and Kadlec and Wallace (2009).

Nevertheless, Wallace (2007) indicated that in large systems, the influence by temperature may be masked by other factors. For the ICW, the relatively larger footprint facilitates a greater range of biological, chemical and physical processes that occur in the wetland environment (Harrington and Ryder, 2002). In addition, fundamental to the design of ICWs is incorporation of the widest possible range of ecological conditions normally found in natural wetlands (Dunne et al., 2005a, b). Therefore, the complex set of wetland ecosystem processes occurring in the ICW may mask the effects of temperature on bulk pollutants removal.

On the other hand, when normalised, the \( k_{20} \) for N removal increased slightly for TN, \( \text{NH}_4-N \) and \( \text{NO}_3-N \), indicating that N removal in the ICW were slightly influenced by temperature, with a reduced removal rate at lower temperatures. The relatively marginal increase in \( k_{20} \) for \( \text{NO}_3-N \) compared to TN and \( \text{NH}_4-N \) also indicated that a fairly constant \( \text{NO}_3-N \) removal rate was achieved in the ICW at all times of the year. Moreover, the estimated \( \theta \) values for the ICW were found to be similar to mean \( \theta \) values for reduction of TN and \( \text{NH}_4-N \) reported by Kadlec and Reddy (2001), Kadlec (2009) and Kadlec and Wallace (2009) for other FWS wetlands, whereas the \( \theta \) for \( \text{NO}_3-N \) reduction were found to be generally lower than the reported values. This finding is in agreement with earlier reports that N removal in CWs is influenced significantly by temperature (Kadlec and Reddy, 2001).
CONCLUSIONS

This study presented a case study of the full-scale application of CWs for decentralized wastewater treatment in rural communities. Findings indicate that using the multi-celled configuration of ICW systems presents a remarkable opportunity to remove significant amounts of N and organic substances from domestic wastewater. Removal rates were affected by variations in influent loadings, but not by variable HLRs. Areal removal rates were linearly related to loading rates, and were greatest when influent loadings were high. This increased the opportunity for N (and organic substance) removal, which typically occurred during warmer (cold) months. The effluent concentrations of NH$_4^+$-N and NO$_3^-$-N were typically less than 1.0 mg/L (2.0 mg/L for total N), with release of NH$_4^+$-N during winter periods. This was a result of plant senescence, and the decreased water temperature and increase in DO, with subsequent increases in ammonification. To sustain high N and organic substance removal performance into the future, dynamic management of ICW systems is required, as it responds to both influent and within-system changes.

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