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Performance Evaluation of Integrated Constructed Wetlands Treating Domestic Wastewater

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Abstract The performances of a new and a mature integrated constructed wetland (ICW) system treating domestic wastewater were evaluated for the first time. The new ICW in Glaslough (near Monaghan, Ireland) comprises five wetland cells, and the mature system in Dunhill (near Waterford, Ireland) comprises four cells. The performance assessment for these systems is based on physical and chemical parameters collected for 1 year in Glaslough and 5 years in Dunhill. The removal efficiencies for the former system were relatively good if compared to the international literature: biochemical oxygen demand (BOD, 99.4%), chemical oxygen demand (COD, 97.0%), suspended solids (SS, 99.5%), ammonia nitrogen (99.0%), nitrate nitrogen (93.5%), and molybdate-reactive phosphorus (MRP, 99.2%). However, the mature ICW had removal efficiencies that decreased over time as the Dunhill village expanded rapidly. The mean removal efficiencies were as follows: BOD (95.2%), COD (89.1%), SS (97.2%), ammonia nitrogen (58.2%), nitrate nitrogen (−11.8%), and MRP (34.0%). The findings indicate that ICW are efficient in removing BOD, COD, SS, and ammonia nitrogen from domestic wastewater. Moreover, both ICW systems did not pollute the receiving surface waters and the groundwater.

Keywords Ammonia nitrogen · Domestic wastewater · Groundwater · Integrated constructed wetland · Molybdate-reactive phosphorus · Surface water

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1 Introduction

Sustainable wastewater treatment is associated with low energy consumption, low capital cost, and, in some situations, low mechanical technology requirements. Therefore, wetland treatment systems could be efficient alternatives to conventional treatment systems, especially for small communities, typically rural or suburban areas, due to low treatment and maintenance costs (Soukup et al. 1994; Solano et al. 2003; Babatunde et al. 2008). Since the 1990s, wetland systems have been used for treating numerous domestic and industrial waste streams including those from tannery and textile industry, abattoirs, pulp and paper production, agriculture (animal farms and fish farm effluents), and various runoff waters (agriculture, airports, highway, and stormwater; Kadlec et al. 2000; Haberl et al. 2003; Scholz 2006; Vymazal 2007; Carty et al. 2008).

The concept of constructed wetlands applied for the purification of various wastewaters has received growing interest and is gaining popularity as a cost-effective wastewater management option in both developed and developing countries. Most of these systems are easy to operate, require low maintenance, and have low investment costs (Machate et al. 1997).

The treatment efficiency of most constructed wetlands depends on the water table level and the dissolved organic concentration of the influent (Reddy and D'Angelo 1997). The water level within most wetland systems (except for tidal flow vertical-flow constructed wetlands (Scholz 2006)) is permanently kept above the wetland soils to create fully saturated soil conditions, resulting generally in high contaminant removal efficiencies. The treatment efficiencies of wetlands vary depending on the wetland design, type of wetland system, climate, vegetation, and microbial communities (Vacca et al. 2005; Ström and Christensen 2007; Picek et al. 2007; Weishampel et al. 2009).

The integrated constructed wetland (ICW) concept was developed not only to address water pollution from different sources including domestic, industrial, and agriculture but also to provide ecological services by restoring potentially lost environmental infrastructure including wetlands. The main features of integrated constructed wetlands are shallow water depth, emergent vegetation, and the use of in situ soils that imitate those found in natural wetland ecosystems. No artificial liners (e.g., plastic or concrete) are used in the construction of ICW. Scholz et al. (2007) and Babatunde et al. (2008) described the detailed concept and removal processes of these robust, sustainable, and synergistic systems by elucidating case studies in Ireland. Wastewater treatment in ICW systems takes place through various physical, chemical, and biological processes involving plants, microorganisms, water, soil, and sunlight (Kadlec and Knight 1996; Scholz 2006; Mitsch and Gosselink 2007).

Although constructed wetlands are mechanically simple treatment systems, the passive treatment processes that remove contaminants are intricate; for instance, the hydrology, microbiology, and water chemistry are complex and interconnected. Research conducted on these systems demonstrates high removal percentages for biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), and pathogens, whereas nutrient removal percentages are usually low and variable. Mitsch and Gosselink (2007) claimed that effective nutrient removal can be achieved after a few growing seasons because of the lack of well-developed belowground and aboveground plant–microbial interactions during the initial seasons. It is a common notion that the nutrient removal efficiency of constructed treatment wetlands decreases with age, especially for phosphorus removal, as the mineral sediment becomes fully saturated; i.e., no free adsorption sites remain (Kadlec 1999).

Most previous studies have been based on either pilot plant-scale or laboratory-scale experimental systems. Very few studies have been carried out on the assessment of performance of full-scale constructed wetlands treating domestic wastewater. There is currently no information on the performance of new and mature ICW systems treating domestic wastewater in the public domain. The purpose of this study was thus:

- To assess for the first time the treatment performance of an ICW system treating domestic wastewater on an industrial scale after 1 year of operation
- To compare for the first time the annual and seasonal treatment efficiency of a full-scale mature and new ICW system
- To investigate the impacts of potential contamination of nearby surface waters and groundwater, taking into consideration that an artificial liner is not present

2 Materials and Methods

2.1 Site Description

The case study systems comprise two ICW treating domestic wastewater in Ireland. The ICW in Glaslough (Fig. 1) is situated in the County of Monaghan (north of the Republic of Ireland), at a longitude of 06° 53′ 37.94″ W and a latitude of 54° 19′ 6.01″ N. The typical annual rainfall is approximately 970 mm over the last 50 years. However, the mean annual rainfall of 1,256 mm was exceptionally high in 2008.

The system was commissioned in October 2007. Its purpose is to treat sewage and to contribute to the improvement of the water quality of the Mountain

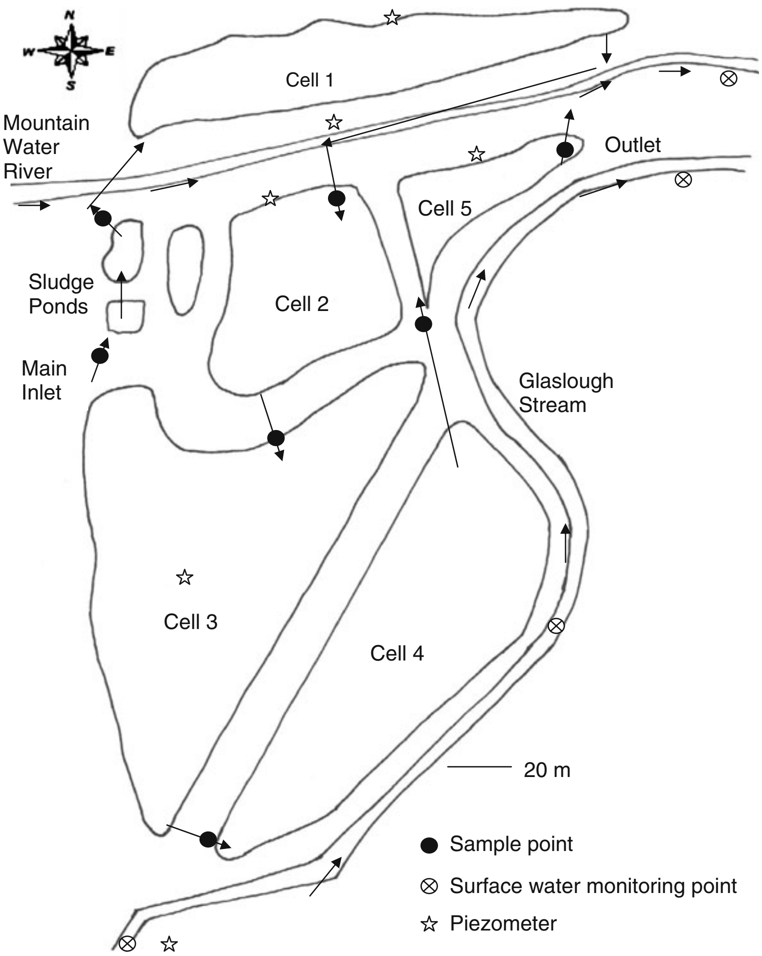

Fig. 1 Sketch showing the groundwater and surface water monitoring and inlet and outlet points for the integrated constructed wetland in Glaslough, near Monaghan (Ireland)

Water River. The inflow rate ranges between approximately 85 and 105 m^3/day. The corresponding outflow (approximately between 1 and 50 m^3/day) was very low due to evapotranspiration and infiltration of treated wastewater. The dilution of the wastewater due to rainfall on the wetlands is roughly between 35% and 65%, depending very much on the season and daily flow fluctuations.

The Glaslough ICW system has a design capacity of 1,750 population equivalent and covers a total area of 6.74 ha. The water surface area of the constructed cells is 3.25 ha.

The ICW in Glaslough (Fig. 1) consists of a small pumping station, two sludge cells, and five shallow vegetated cells. Domestic sewage from the village is pumped to the pumping station on site and from there to one of the sludge cells. There are two sludge collection cells that can be operated alternately to allow for subsequent desludging of the other cell, if it is not in operation. From the sludge cell, the wastewater flows by gravity through the five vegetated cells, and the effluent finally discharges directly to the adjacent Mountain Water River. The wetlands were planted with *Carex riparia* Curtis, *Phragmites australis* (Cav.) Trin. ex Steud., *Typha latifolia* L., *Iris pseudacorus* L., *Glyceria maxima* (Hartm.) Holmb., *Glyceria fluitans* (L.) R.Br., *Juncus effusus* L., *Sparganium erectum* L. emend Rchb, *Elisma natans* (L.) Raf., and *Scirpus pendulus* Muhl. The main ICW system is flanked by the Mountain Water River and the Glaslough Stream.

The ICW system in Dunhill (County Waterford; southeast of Ireland) is situated at a longitude of 07° 02′ 40″ W and a latitude of 52° 11′ 28″ N (Fig. 2). The typical annual rainfall is approximately 1,000 mm. Until 2000, sewage at Dunhill village was directed to a wastewater treatment plant (septic tank system). In late 2000, the system was upgraded

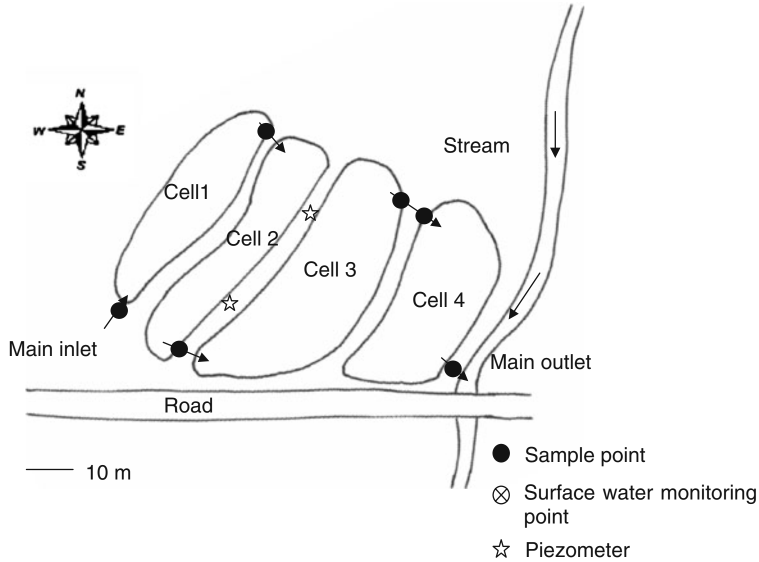

Fig. 2 Sketch showing the groundwater and surface water monitoring and inlet and outlet points for the integrated constructed wetland system in Dunhill, near Waterford (Ireland)

with the help of an ICW system that was fully operational by February 2001. The wastewater inflow was approximately 40 m^3/day. The corresponding outflow was roughly 24 m^3/day. Dilution of the wastewater due to rainfall was approximately between 5% and 20%, depending on season. However, detailed daily flow values are not available for this complex open system.

The main purpose was to treat sewage and to contribute to the improvement of the water quality of the Annestown stream. The system has a total area of 0.3 ha. The primary vegetation types used in the ICW are emergent plant species (helophytes). The system is gravity-fed and has therefore no energy consumption. Wastewater from households is collected via the sewerage system and then transported to the wetland system. A single influent entry point is located in the first cell. The ICW system was based on four cells operating in series. The final effluent enters the Annestown stream via the outlet of the final ICW cell.

All cells consist of one inflow and one outflow structure, and the flow between each cell has been by gravity through PVC pipes. Artificial liners were not used for both wetlands. However, the subsoil was worked and used as a natural liner.

2.2 Sampling and Analytical Methods

2.2.1 Water Quality

Grab samples for the inlet and outlet of each wetland cell were taken approximately quarterly at the ICW in Dunhill, while a substantial suite of hi-tech automatic sampling and monitoring instrumentation has been used for approximately weekly sampling at the ICW in Glaslough: ISCO 4700 Refrigerated Automatic Wastewater Sampler (Teledyne Isco, Inc., NE, USA), Siemens Electromagnetic Flow Meter F M MAGFLO and MAG5000 (Siemens Flow Instruments A/S, Nordborgrej 81, DK-6430 Nordborg, Denmark). Furthermore, the Mountain Water River and Glaslough Stream were also monitored (Fig. 1). Water samples were analyzed for variables including the 5 days at 20°C *N*-allylthiourea BOD, COD, SS, pH, ammonia nitrogen, nitrate nitrogen, molybdate-reactive phosphorus (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.

2.2.2 Groundwater Quality

Six piezometric groundwater-monitoring wells (Fig. 1) were sampled at the Glaslough site to monitor the groundwater quality. The wells were placed within the ICW system and along the suspected flow path of contaminants towards the receiving watercourse. The ICW system was constructed using in situ soils. Subsoil obtained from the ICW site was reworked to line the ICW banks and cell beds to reduce groundwater infiltration and subsequently pollution. When polluted water flows through the ICW system, suspended solids settle naturally on the soil surface, obstructing infiltration of pollutants trough the wetland cells (Kadlec and Knight 1996; Scholz 2006; Wallace and Knight 2006).

The water table at the ICW site is relatively high (i.e., 1.8–2.0 m below the ICW beds), so it is very important to monitor that the ICW system has no negative effect on the groundwater. Six piezometers have been placed at various depths (between 2.49 and 3.87 m). A site investigation by the Geological Survey of Ireland (IGSL Ltd., Unit F, M7 Business Park, Naas, County Kildare, Ireland) in September 2005 indicated a soil coefficient of permeability of approximately 9×10^{-11} m/s. The piezometer 1 near the wetland cell 1 (close to a small hill) and piezometer 6 (located across the Glaslough stream) are outside of the ICW system (see Fig. 1).

For the ICW system in Dunhill, two piezometric groundwater-monitoring wells were sampled at a depth of 5 m (Fig. 2). The wells were placed within the ICW system and along the suspected flow path of contaminants to assess the risk of groundwater pollution. The subsequent water quality analysis for both ICW systems was carried out according to APHA (1998).

2.2.3 Stream Water Quality

The stream water quality adjacent to the ICW systems was regularly monitored to assess the impact of ICW on receiving waters and verify that the ICW discharge is not polluting the receiving waters. The Mountain Water River is sampled at two locations, one upstream and one approximately 400 m downstream of the discharge point. Moreover, the Glaslough stream (not a directly receiving watercourse) is monitored at three points.

The Annestown stream near the ICW in Dunhill is also monitored to check for compliance to discharge standards. The two sampling points are located approximately 4 km upstream and 3.5 km downstream of Dunhill village.

2.3 Statistical Analyses

All statistical analyses were carried out by using the computer software package Origin 7.5. A parametric analysis of variance was used to determine any significant ($p<0.05$) differences in removal percentages and the seasonal effect on water quality for both ICW systems.

3 Results and Discussion

3.1 Water Quality of the ICW System in Glaslough

The mean influent and effluent concentrations and seasonal comparisons of the water quality variables are presented in Tables 1 and 2, respectively. The approximate mean inflow values were as follows: BOD, 768±451.0 mg/l; COD, 1,279±697.8 mg/l; SS, 2,184±3,844.8 mg/l; ammonia-nitrogen, 32±11.1 mg/l; nitrate-nitrogen, 5±3.8 mg/l; MRP, 4±2.0 mg/l; pH, 7±0.4. These values indicate a very high variability of the domestic wastewater entering the ICW system.

However, the ICW system has shown a very good treatment performance, despite being a new (i.e., not mature) system. The ICW system removed approximately 99% of BOD, 97% of COD, 100% of SS, 99% of ammonia-nitrogen, 94% of nitrate-nitrogen, and 99% of MRP during this period. The results show that the pollutant removal capacity is very high due to its large wetland size, providing high mean retention times. Findings contrast with the general idea that the organic matter removal rate increases depending on constructed wetland age (Kadlec 1999). An increase in age is also associated with an increase in microbial population. Furthermore, microbial biofilm formation on the bed material within the wetlands leads to higher biological degradation rates (Picard et al. 2005).

There is very little information on full-scale representative constructed wetlands treating domestic wastewater in the scientific literature. However, numerous studies refer to pilot-scale and microcosm-scale constructed wetlands treating domestic wastewater treatment. Ciria et al. (2005) assessed the role of *T. latifolia* L. (reedmace, cattail, or bullrush) in constructed wetlands of 40 m^2 each, filled with gravel as the supporting medium. Their study showed that BOD and COD removal efficiencies were 97±1.2% and 79±0.3% in first year, respectively, and in the second year, the BOD removal efficiency did not change (97±3.0%), while the COD removal efficiency increased slightly (81±1.0%). Furthermore, a study by Hamouri et al. (2007) achieved removal efficiencies of 78% for COD and 79% for BOD with respect to a combination of a two-step upflow anaerobic reactor and subsurface horizontal-flow constructed wetland (Hamouri et al. 2007). Between 71% and 75% removal efficiencies for COD and BOD were noted for

Table 1 Water quality variables for the ICW

Variables	ICW in Glaslough (February 2008–March 2009)								ICW in Dunhill (August 2001–January 2006)						
	Unit	Influent			Effluent			Removal (%)	Influent			Effluent			Removal (%)
		n	Mean	SD	*n*	Mean	SD		*n*	Mean	SD	*n*	Mean	SD	
Biochemical oxygen demand	mg/l	45	768.1	450.99	62	5.0	3.94	99.4	23	358.4	200.57	27	17.2	12.33	95.20
Chemical oxygen demand	mg/l	65	1279.3	697.79	68	39.0	31.02	97.0	25	554.4	288.19	35	60.6	33.37	89.07
Suspended solids	mg/l	62	2183.8	3844.82	66	11.9	21.69	99.5	24	303.5	335.46	26	8.5	6.77	97.21
Ammonia-nitrogen	mg/l	67	32.1	11.07	71	0.3	0.52	99.0	24	52.6	39.30	61	22.0	15.04	58.20
Nitrate-nitrogen	mg/l	67	4.8	3.77	70	0.3	0.29	93.5	9	0.6	1.47	9	0.7	1.29	-11.98
Molybdate-reactive-phosphorus	mg/l	66	3.7	2.00	70	0.0	0.04	99.2	25	7.8	3.38	62	5.2	3.00	33.97
pH	–	63	6.9	0.39	66	7.6	0.37	–	21	7.0	0.27	23	7.0	0.31	–

n sample number, *SD* standard deviation

constructed wetlands treating secondary treated sewerage (Thomas et al. 1995).

The main nitrogen removal process within constructed wetland systems include uptake from plants and other living organisms, sedimentation, nitrification, denitrification, ammonia volatilization, and cation exchange for ammonium (Majer Newman et al. 1999; Yang et al. 2001; Scholz 2006; Wallace and Knight 2006; Mitsch and Gosselink 2007). Kadlec et al. (2000) have explained that interactions between nitrogen on one side and water, sediment, plant, and biomass on the other side make it difficult to assess the real efficiency of nitrogen removal due to storage in the system.

The overall ammonia-nitrogen reduction is high (Table 1) in comparison to other microcosm wetlands treating domestic wastewater: 76–92% for subsurface constructed mangroves in Hong Kong (Wu et al. 2008); 52% for a combination of a free surface-flow wetland cells, which were fed with municipal lagoon effluents in Canada (Cameron et al. 2003); 10–20% for a continuous-flow, free water surface pilot wetland planted with *Lemna gibba* L. (duck weed) in Israel (Ran et al. 2004); 9% for a subsurface horizontal-flow constructed wetland in Morocco (Hamouri et al. 2007); 14–24% for a constructed wetland treating secondary treated sewerage in Australia (Thomas et al. 1995).

Seasonal variations in performance related to nutrient removal were also investigated (Table 2). Monthly water quality variables are provided in Table 3. The BOD concentrations within the influent and the effluent were relatively high in summer and autumn compared to other seasons. However, the BOD removal efficiency in summer was similar to those of the other seasons. The COD concentration of the influent was lower in spring compared with the other seasons. However, lower COD concentrations within the effluent (27.0±9.52 mg/l; removal efficiency of 98%) were recorded during winter in comparison to concentrations (52.6±53.48 mg/l; removal efficiency 96%) in spring. On the contrary, the lower effluent concentrations (1.3±1.49 mg/l) were recorded for SS in autumn. Hunt and Poach (2001) explain that constructed wetland systems cannot completely remove carbon and solid compounds because wetland plants produce plant litter, which continuously adds carbon and other compounds to the system.

The highest ammonia-nitrogen and nitrate-nitrogen concentrations within the influent were 35.3±9.87 mg/l in spring and 7.7±2.60 mg/l in summer, respectively. On the other hand, the lowest ammonia-nitrogen and nitrate-nitrogen concentrations in the effluent were 0.0±0.15 mg/l in autumn (removal efficiency of 100%) and 0.2±0.12 mg/l (removal efficiency of 83%) in winter, respectively. The effluent ammonia-nitrogen concentrations were slightly higher in winter compared to the other seasons. This can be explained by the fact that nitrification of ammonia-nitrogen is relatively low in winter due to the lack of oxygen and low temperatures, which negatively affect

Table 2 Seasonal comparison of the nutrient removal efficiency for the integrated constructed wetland in Glaslough (February 2008–March 2009)

Variables	Spring			Summer			Autumn			Winter		
	n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Biochemical oxygen demand												
Inflow (mg/l)	14	706.1	418.81	10	1125.0	468.32	11	743.6	466.52	10	525.0	256.73
Effluent (mg/l)	14	9.1	4.91	25	4.5	2.98	12	2.6	1.62	11	3.4	2.26
Removal (%)	98.7			99.6			99.7			99.4		
Chemical oxygen demand												
Inflow (mg/l)	19	1249.7	767.94	24	1237.7	649.29	13	1272.6	625.02	9	1462.6	851.63
Effluent (mg/l)	19	52.6	53.48	25	38.5	14.36	14	30.0	9.41	10	27.1	9.52
Removal (%)	95.8			96.9			97.6			98.2		
Suspended solids												
Inflow (mg/l)	17	797.5	821.79	24	4567.1	5373.86	13	459.3	337.03	8	782.2	534.59
Effluent (mg/l)	16	27.7	38.24	25	8.4	8.61	14	1.3	1.49	9	9.8	10.44
Removal (%)	96.5			99.8			99.7			98.8		
Ammonia-nitrogen												
Inflow (mg/l)	19	35.3	9.87	24	32.6	11.14	13	30.1	9.13	11	28.1	14.33
Effluent (mg/l)	20	0.2	0.17	25	0.3	0.14	14	0.0	0.15	12	1.0	1.04
Removal (%)	99.4			99.2			99.6			96.6		
Nitrate-nitrogen												
Inflow (mg/l)	19	2.2	2.41	24	7.7	2.60	13	6.3	3.63	11	1.3	1.32
Effluent (mg/l)	19	0.4	0.47	25	0.4	0.20	14	0.3	0.16	12	0.2	0.12
Removal (%)	82.7			95.5			95.9			82.8		
Molybdate-reactive-phosphorus												
Inflow (mg/l)	19	3.6	1.72	23	3.5	1.76	13	3.4	1.28	11	4.3	3.36
Effluent (mg/l)	20	0.1	0.1	24	0.1	0.05	14	0.0	0.01	12	0.1	0.04
Removal (%)	99.8			98.6			99.7			98.8		

n sample number, *SD* standard deviation

nitrification rates within constructed wetlands. Moreover, the effluent nitrate-nitrogen concentration was slightly different irrespective of seasonal change, although the influent nitrate-nitrogen concentration was considerably higher in summer and autumn than in spring and winter. This indicates that the denitrification rate for the system was considerably high, especially in summer and autumn due to high temperatures, elevated microbial activity, and the presence and easy availability of organic carbon.

Oxygen is generally used for nitrification and organic matter reduction. However, oxygen is partly generated due to photosynthesis during daytime and supports the oxygen demand for stabilization of organics and nitrification. Plant rhizosphere aeration may stimulate aerobic decomposition processes, increase nitrification and subsequent gaseous losses of nitrogen via denitrification, and decrease relative levels of dissimilatory nitrate reduction to ammonium (Tanner et al. 1995). Microbial activity and plant nutrient uptake within wetlands are both directly and indirectly affected by temperature. Nitrification is a temperature-dependant process. Werker et al. (2002) informed that nitrification rates in constructed wetlands become increasingly inhibited at temperatures of about 10°C, and rates drop rapidly at 6°C. Akratos and Tsihrintzis (2007) noted high-temperature requirements for the removal of nitrogen compounds such as ammonia at temperatures above 15°C. Temperature affected the system performance considerably due to changing biological activities with temperature.

Table 3 Water quality variables (mean ± standard deviation) for the integrated constructed wetland in Glaslough (February 2008–March 2009)

Month	Biochemical oxygen demand (mg/l)		Chemical oxygen demand (mg/l)		Ammonia-nitrogen (mg/l)		Nitrate-nitrogen (mg/l)		Molybdate-reactive-phosphorus (mg/l)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
January	456.0±50.91	2.3±0.58	807.0±469.52	20.5±6.38	19.2±18.50	2.44±0.44	2.1±2.69	0.1±0.06	1.7±0.89	0.1±0.02
February	476.4±185.18	4.4±2.66	1,686.8±911.08	29.6±11.06	29.6±15.01	0.6±0.63	1.1±1.07	0.3±0.01	5.1±3.93	0.0±0.04
March	303.2±89.39	3.8±1.50	914.1±663.83	18.6±10.02	27.6±10.54	0.0±0.08	0.7±0.75	0.1±0.12	3.2±1.49	0.0±0.01
April	596.4±278.94	8.7±0.00	1,188.8±493.83	19.0±17.00	38.1±8.81	0.1±0.24	0.8±0.77	0.6±0.71	4.9±2.36	0.0±0.00
May	1,045.3±271.01	11.5±4.19	1,591.1±838.02	90.3±57.01	41.0±4.71	0.3±0.14	3.9±2.51	0.5±0.53	10.3±4.27	0.2±0.12
June	1,423.6±237.10	6.9±4.31	1,305.6±450.11	41.1±11.03	35.3±6.64	0.23±0.12	9.4±2.07	0.3±0.20	3.8±1.78	0.1±0.04
July	777.8±1,100.00	3.1±0.88	1,388.9±868.94	35.8±18.91	34.2±11.47	0.2±0.07	8.2±1.99	0.3±0.09	3.3±1.8	0.0±0.01
August	800.0±428.66	3.9±0.90	965.7±496.18	39.4±11.12	27.3±14.21	0.3±0.20	5.1±1.95	0.5±0.26	3.3±1.90	0.1±0.07
September	526.3±329.21	3.3±2.08	1,078.0±561.47	32.5±4.44	28.3±8.29	0.1±0.11	3.9±3.21	0.3±0.12	3.6±0.45	0.0±0.01
October	850.0±480.88	2.5±1.87	1,195.0±449.92	31.3±11.73	29.3±10.81	0.1±0.02	6.7±2.53	0.3±0.21	3.3±1.02	0.0±0.01
November	783.3±644.85	2.0±0.00	1,535.0±934.51	24.3±3.21	32.7±8.91	0.1±0.03	7.6±5.19	0.2±0.08	3.4±2.17	0.0±0.01
December	739.5±550.84	2.0±1.41	1,557.5±1,064.2	30.5±7.78	31.7±12.60	0.2±0.02	1.1±1.27	0.1±0.12	4.2±1.19	0.0±0.03

SD standard deviation

Ciria et al. (2005) informed that ammonia-nitrogen and nitrate-nitrogen reduction rates were significantly higher in the second compared to the first year of operation, and they obtained the highest removal rates of 22±0.8% for ammonia-nitrogen and 64±0.9% for nitrate-nitrogen in the autumn of the first year and 40±3.5% for ammonia-nitrogen in summer and 75±2.1% for nitrate-nitrogen in winter for the second year. In comparison to this previous study, the ammonia-nitrogen and nitrate-nitrogen removal efficiencies for all seasons are significantly higher for the ICW system (Table 2).

Phosphate removal mechanisms are based on physical (sedimentation), chemical (adsorption), and biological processes. Phosphorus can be reduced directly by plant uptake or chemical storage within the sediments (Bonomo et al. 1997). On the contrary, Sakadevan and Bavor (1998) suggest that long-term phosphorus removal mechanisms in constructed wetland systems is likely due to uptake by the sub-stratum, litter, and aluminum/iron compounds, while plant uptake is often a relatively small fraction. Furthermore, Kadlec (1999) informed that the phosphorus reduction capacity decreases with age because the mineral sediments become fully saturated within the wetland systems; i.e., no free adsorption sites remain. The annual MRP reduction rate for Glaslough was 99.2%, which means that the system has a high phosphorus adsorption and storage, and plant uptake capacity. However, the system is relatively young as well.

3.2 Receiving Stream Water Quality

3.2.1 Mountain Water River and Glaslough Stream

The Mountain Water River is adjacent to the sludge cells and wetland cells 1, 2, and 5, whereas the Glaslough Stream is adjacent to the wetland cells 4 and 5. Annual and seasonal surface water quality data are provided in Tables 4 and 5, respectively. At the discharge point, mean effluent concentrations were 0.4 mg/l for ammonia-nitrogen, 1.0 mg/l for nitrate-nitrogen, and 0.1 mg/l for MRP, whereas mean river downstream concentrations were 0.40 mg/l for ammonia-nitrogen, 1.02 mg/l for nitrate-nitrogen, and 0.1 mg/l for MRP (Table 4). The river water quality did not change in the downstream compared to the upstream stretch. This indicates that the river has sufficient assimilative capacity.

Seasonal surface water quality variables are provided in Table 5. In summer, the SS and nitrate-nitrogen concentrations in the Mountain Water River were slightly higher downstream than upstream, which was, however, not statistically significant ($p<0.05$). Similarly, the BOD and COD concentrations of the influent and effluent were higher in summer than during the other seasons. This can be explained by the decrease in flow rate and also by high evaporation rates in summer, while pollutant loads remain similar. The river and stream water quality is variable and predominantly depends on patterns of precipitation. Furthermore, especially in spring and autumn, high precipitation rates occur during periods when the buffering capacity of the receiving water is enhanced by an increased dilution ratio.

The Mountain Water River had mean ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations of 0.4±0.28, 1.0±0.41, and 0.1±0.06 mg/l for the upstream and 0.4±0.28, 1.0±0.38, and 0.1±0.06 mg/l for the downstream stretches, respectively. The Glaslough Stream, which originally passed through the site, was diverted and widened around the perimeter of cell 4. There is no discharge point from the ICW system into the Glaslough Stream, but there are three sampling points to assess potential contamination from the ICW. Concerning the mean water quality values, ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations were 0.3±0.32, 1.2±0.37, and 0.1±0.07 mg/l, respectively. The water quality was better downstream than upstream. The ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations for the sample points adjacent to cell 5 and after cell 5 were 0.3±0.12 and 0.2±0.11, 0.9±0.44 and 0.8±0.50, and 0.1±0.07 and 0.2±0.13 mg/l, respectively. This indicates that the ICW system does not pollute the nearby Glaslough Stream.

Molybdate-reactive phosphorus effluent concentrations are in compliance with the Irish Phosphorous Regulations 1998 (Environmental Protection Agency 1998), which set an annual median threshold of 0.03 mg/l for rivers. However, neither the Mountain Water River nor the Glaslough Stream upstream of the ICW system complied with this regulation.

3.2.2 Annestown Stream

The MRP concentration at the monitoring station Ballyphilip (4 km upstream of the ICW system) was

Table 4 Water quality variables for the surface water for Glaslough ICW (February 2008–March 2009)

Monitoring site	Biochemical oxygen demand (mg/l)			Chemical oxygen demand (mg/l)			Suspended solids (mg/l)			Ammonia-nitrogen (mg/l)			Nitrate-nitrogen (mg/l)			Molybdate-reactive phosphorus (mg/l)		
	n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
MRU	47	3.3	1.65	51	37.0	23.96	48	9.4	17.19	53	0.4	0.28	53	1.0	0.41	52	0.1	0.06
MRD	47	3.4	1.82	51	35.4	22.05	46	10.8	16.73	51	0.4	0.28	52	1.0	0.38	51	0.1	0.06
GSS	30	3.8	3.27	31	29.5	15.79	31	6.6	8.14	31	0.3	0.32	31	1.2	0.37	30	0.1	0.07
GSNC4	29	3.3	1.66	32	28.9	16.30	29	7.1	9.90	30	0.3	0.12	30	0.9	0.44	29	0.1	0.07
GSNC5	30	6.3	5.06	31	48.1	42.31	31	20.1	24.86	31	0.2	0.12	31	0.8	0.50	30	0.2	0.13

MRU Mountain Water River upstream, *MRD* Mountain Water River downstream, *GSS* Glaslough Stream source, *GSNC4* Glaslough Stream near cell 4, *GSNC5* Glaslough Stream near cell 5, *n* sampling number, *SD* standard deviation

Table 5 Water quality variables for the surface water near the integrated constructed wetland system in Glaslough (February 2008–March 2009)

Monitoring side	Site	Biochemical oxygen demand (mg/l)			Chemical oxygen demand (mg/l)			Suspended solids (mg/l)			Ammonia-nitrogen (mg/l)			Nitrate-nitrogen (mg/l)			Molybdate-reactive phosphorus (mg/l)		
		n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Spring	MRU	13	3.1	1.02	15	20.9	8.55	13	2.9	3.69	15	0.2	0.11	15	0.8	0.19	15	0.1	0.04
	MRD	13	3.4	0.99	15	22.7	10.06	11	4.6	4.39	15	0.2	0.12	15	0.8	0.24	15	0.1	0.03
	GSS	4	3.8	2.34	9	20.8	10.07	4	2.5	1.91	4	0.3	0.17	4	1.3	0.54	4	0.1	0.04
	GSNC4	3	3.2	0.59	10	21.5	10.18	2	33.0	21.21	3	0.3	0.20	3	1.0	0.32	3	0.1	0.02
	GSNC5	4	7.1	7.54	9	62.4	67.67	4	24.6	46.30	4	0.3	0.12	4	1.2	0.58	4	0.1	0.05
Summer	MRU	25	4.0	1.82	25	48.3	28.16	25	14.1	22.72	25	0.6	0.31	25	1.1	0.47	24	0.1	0.06
	MRD	25	3.9	2.14	25	44.7	26.38	25	14.9	21.71	24	0.5	0.30	25	1.2	0.45	24	0.1	0.07
	GSS	25	3.9	3.43	20	35.0	15.77	25	7.7	8.69	25	0.4	0.35	25	1.1	0.28	24	0.2	0.05
	GSNC4	25	3.5	1.70	20	34.0	17.52	25	4.2	2.11	25	0.3	0.12	25	0.8	0.41	24	0.1	0.06
	GSNC5	25	6.4	4.74	20	44.9	25.98	25	20.6	21.91	25	0.2	0.11	25	0.6	0.37	24	0.2	0.13
Autumn	MRU	6	1.8	0.75	8	35.9	12.67	8	5.3	3.69	8	0.4	0.15	8	0.9	0.16	8	0.1	0.02
	MRD	6	2.2	0.75	8	34.3	13.17	8	6.8	3.37	8	0.4	0.15	8	0.9	0.12	8	0.1	0.01
	GSS	–	–	–	1	22.0	–	1	0.0	–	1	0.2	–	1	1.0	–	1	0.0	–
	GSNC4	–	–	–	1	21.0	–	1	32.0	–	1	0.3	–	1	1.3	–	1	0.1	–
	GSNC5	–	–	–	1	14.0	–	1	8.0	–	1	0.2	–	1	1.0	–	1	0.1	–
Winter	MRU	3	2.0	1.00	3	26.2	5.84	2	9.4	0.85	5	0.3	0.29	5	1.3	0.60	5	0.1	0.04
	MRD	3	2.2	2.13	3	24.8	5.06	2	10.0	5.66	4	0.4	0.43	4	1.0	0.21	4	0.1	0.05
	GSS	1	0.5	–	1	5.0	–	1	1.6	–	1	0.1	–	1	2.3	–	1	0.0	–
	GSNC4	1	0.7	–	1	10.0	–	1	2.4	–	1	0.1	–	1	1.9	–	1	0.0	–
	GSNC5	1	0.7	–	1	16.0	–	1	1.2	–	1	0.1	–	1	2.2	–	1	0.0	–

MRU Mountain Water River upstream, *MRD* Mountain Water River downstream, *GSS* Glaslough Stream source, *GSNC4* Glaslough Stream near cell 4, *GSNC5* Glaslough Stream near cell 5, *n* sampling number, *SD* standard deviation well below the target phosphorus concentration of 0.03 mg/l, set by the Irish Phosphorus Regulations 1998 (Environmental Protection Agency 1998). However, there has been a very slight increase (0.002 mg/l) in the MRP concentration 3.5 km downstream of the ICW system.

A slight increase in nutrient concentrations between the two monitoring points upstream and downstream of the ICW system was noted. However, only ammonia-nitrogen increased significantly ($p<0.05$), while increases in nitrate-nitrogen and MRP concentrations were statistically not significant. The increase in nutrient concentrations downstream of the ICW system may be attributed to runoff containing nutrients originating from intensive cattle farming. Furthermore, the ICW system is overloaded; over the course of time, new housing developments in Dunhill have led to an increase in sewage (Table 1).

3.3 Groundwater Quality

3.3.1 Glaslough

The mean ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations within piezometer 1 were 0.5±0.28, 0.3±0.15, and 0.2±0.41 for MRP, respectively (Table 6). Piezometer 2 is located in the west of the ICW system near cell 1, whereas the piezometers 3 and 4 can be found in the east near cells 2 and 5. The mean ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations for piezometer 2 were 0.7±0.42, 0.2±0.10. and 0.4±0.31 mg/l, which indicates a slight increase in concentrations (Table 7). No groundwater contamination was observed. The water quality characteristics of piezometers 3 and 4 concerning ammonia-nitrogen, nitrate-nitrogen, and MRP were as follows: 4.6±4.42 and 0.3±0.28, 0.8±0.51 and 0.5±0.55, and 0.4±0.53 and 0.2±0.50 mg/l, respectively. The mean water quality concentrations for piezometer 3 are higher than for piezometer 4. This can be explained by the observation that piezometer 3 is located near wetland cell 2, which contains more pollutants than other cells (except for cell 1). Furthermore, low infiltration takes place within cell 2 due to the presence of a sandy layer at the bottom. The mean water quality values regarding piezometer 6, which is located across the Glaslough Stream, were 2.8±1.51, 0.9±0.83, and 0.2±0.20 mg/l for ammonia-nitrogen, nitrate-nitrogen, and MRP, respectively. The ammonia-nitrogen concentrations were often high at piezometer 6, which is located near an Equestrian Center, potentially polluting the groundwater.

3.3.2 Dunhill

Piezometer 1 near wetland cell 2 and piezometer 2 near cell 3 have been located within the ICW system to monitor groundwater quality (see also Fig. 2). Concerning piezometer 1, 4.7±12.25 and 0.4±0.93 mg/l have been measured for ammonia-nitrogen and MRP, respectively. The water quality for piezometer 2 was as follows: ammonia-nitrogen and MRP had mean concentrations of 2.6±1.16 and 0.0±0.02 mg/l, respectively. This can be explained by the fact that the ammonia-nitrogen reduction due to the ICW system is relatively low due to overloading.

Table 6 Nutrient concentrations for water samples taken from groundwater-monitoring wells at the integrated constructed wetland (ICW) site in Glaslough (11 March–28 April 2009)

Monitoring well number	Position	Piezometer depth (m)	Water level (m)	Ammonia-nitrogen (mg/l)			Nitrate-nitrogen (mg/l)			Molybdate-reactive phosphorus (mg/l)		
				n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Piezometer 1	Near cell 1 on hill	2.49	1.38	19	0.5	0.28	19	0.3	0.15	19	0.2	0.41
Piezometer 2	Near cell 1 on the path	3.87	2.57	27	0.7	0.42	25	0.2	0.10	27	0.4	0.31
Piezometer 3	Near cell 2 on the path	2.89	0.69	21	4.6	4.42	21	0.8	0.51	21	0.4	0.53
Piezometer 4	Near cell 5	3.29	0.69	27	0.3	0.28	27	0.5	0.55	27	0.2	0.50
Piezometer 5	On an island in cell 3	3.63	2.56	26	0.6	0.17	23	0.3	0.16	26	0.2	0.43
Piezometer 6	Near cells 3 and 4, and stream	3.00	1.75	27	2.8	1.51	25	0.9	0.83	27	0.2	0.20

n sample number, *SD* standard deviation

Table 7 Nutrient concentrations within the groundwater-monitoring wells before operation of the integrated constructed wetland in Glaslough

Monitoring well number	Position	Piezometer depth (m)	Water level (m)	Sample number	Biochemical oxygen demand (mg/l)	Chemical oxygen demand (mg/l)	Suspended solids (mg/l)	Ammonia-nitrogen (mg/l)
Piezometer 1	Near cell 1 on hill	2.49	1.38	2	2.6 9	18	–	0.03
Piezometer 2	Near cell 1 on the path	3.87	2.57	2	1.5	50	46.5	0.41
Piezometer 3	Near cell 2 on the path	2.89	0.69	2	4.95	185.5	578	0.29
Piezometer 4	Near cell 5	3.29	0.69	2	1.5	18	37	0.2
Piezometer 5	On an island in cell 3	3.63	2.56	2	1.5	15.5	7.5	0.15
Piezometer 6	Near cells 3 and 4, and stream	3.00	1.75	2	5.35	69.5	63	2.32

Moreover, some infiltration may occur from cell 2. Concerning MRP, values were low within the piezometer sample water. It is most likely that MRP is taken up by the soil via adsorption.

3.4 Comparison of Nutrient Removal Performances

The treatment performances of the ICW systems located at both Glaslough and Dunhill are summarized in Tables 1 and 2. Overall, both systems indicate significant ($p<0.05$) COD and BOD removal efficiencies. Concerning the other water quality data, ammonia nitrogen, nitrate nitrogen, and MRP removal efficiencies for the Glaslough system were high: 99.0%, 93.5%, and 99.2%, respectively. In comparison, the ICW in Dunhill had removal efficiencies of 58%, −80.8% (source rather than sink), and 34.0%, respectively. Nitrate nitrogen and MRP concentrations within the effluent gradually increased (see also Tables 8 and 9). The decreasing nutrient removal rates for Dunhill are probably due to the increased system overload. Ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations within the effluent are three times higher for the fourth year of its operation than for the previous 3 years. Nitrate-nitrogen concentrations within the effluent were higher than for the influent, which means that some ammonia-nitrogen is transferred into nitrate-nitrogen via nitrification. However, ammonia-nitrogen and nitrate-nitrogen were both released from the ICW in Dunhill.

The organic material present within the ICW cells has also an indirect impact on the bacterial community. For instance, the litter on top of the sediment is likely to have limited the diffusion of oxygen to the lower sediment layers, creating anoxic conditions and, hence, making conditions favorable for denitrification. This possible process has been described previously by Bastviken et al. (2005) for a comparable system. Most denitrifiers are heterotrophs, and the supply of organic carbon by macrophytes may have raised the overall heterotrophic activity, leading to the consumption of oxygen (Souza et al. 2008). Thus, it is likely that the oxygen availability within the sediment was reduced, and denitrification was subsequently supported (Bastviken et al. 2005).

The COD and BOD effluent concentrations within both ICW systems were generally higher in summer and autumn than in winter and spring. However, the removal efficiencies for these parameters did not change significantly for both systems (Table 2). This is probably due to the increase in organic loading rate as a consequence of an increase of the evaporation rate and a decrease of the precipitation rate. On the other hand, effluent ammonia-nitrogen, nitrate-nitrogen, and MRP concentrations did not change considerably for the ICW system in Glaslough, whereas these variables increased for the system in Dunhill.

The difference in removal rate could also be due to higher hydraulic retention times provided for the ICW system in Glaslough, performing better in terms of MRP removal compared to the system in Dunhill. The Glaslough system removed 99.2% more MRP than the ICW system in Dunhill. The difference in MRP reduction is likely due to Glaslough's subsoil and

Table 8 Seasonal comparison of the nutrient removal efficiency for the integrated constructed wetland in Dunhill (February 2001–March 2005)

Variable and statistics	Spring			Summer			Autumn			Winter		
	n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Biochemical oxygen demand												
Inflow (mg/l)	7	657.0	296.03	6	325.0	96.30	7	452.3	292.49	4	293.5	118.31
Effluent (mg/l)	8	24.7	11.00	8	17.2	8.37	8	23.1	17.64	4	18.3	9.22
Removal (%)	96.3			94.7			94.9			93.8		
Chemical oxygen demand												
Inflow (mg/l)	7	372.0	190.03	7	527.4	234.42	7	786.1	370.01	5	464.6	119.08
Effluent (mg/l)	8	46.0	19.01	10	74.0	32.82	10	74.6	41.06	8	37.4	13.78
Removal (%)	87.6			86.0			90.5			92.0		
Suspended solids												
Inflow (mg/l)	7	147.4	155.45	6	622.2	472.91	7	317.1	234.53	5	68.0	38.91
Effluent (mg/l)	7	11.1	6.54	7	8.9	9.14	7	8.1	6.18	6	7.2	6.77
Removal (%)	92.4			98.6			97.4			89.5		
Ammonia-nitrogen												
Inflow (mg/l)	8	60.0	37.87	6	66.0	61.91	7	38.1	16.09	4	38.0	21.17
Effluent (mg/l)	12	14.1	16.27	19	24.4	15.31	18	27.3	14.96	13	23.7	13.53
Removal (%)	76.5			63.1			41.4			37.7		
Nitrate-nitrogen												
Inflow (mg/l)	1	0.1	–	3	0.1	0	3	0.1	0	2	4.5	2.30
Effluent (mg/l)	1	0.4	–	3	0.1	0	3	0.1	0	2	0.82	0.68
Removal (%)	−300.0			0.0			0.0			64.4		
Molybdate-reactive-phosphorus												
Inflow (mg/l)	8	7.6	4.01	6	10.1	3.85	7	6.2	2.27	5	7.0	1.97
Effluent (mg/l)	12	4.2	3.07	19	6.3	3.33	18	5.2	3.04	14	4.0	1.85
Removal (%)	44.3			37.3			15.0			43.1		

n sample number, *SD* standard deviation

sediment, which may not have reached the saturation threshold.

3.5 Comparison of Nutrient Reduction in Wetland Cells

The ICW systems operate as sequential multicellular structures and have a minimum number of four wetland cells. The influent (i.e., effluent from the sedimentation cell) to the first wetland cell in Glaslough has the following characteristics: BOD, 405.1±204.03 mg/l; COD, 773.0±506.67 mg/l; SS, 238.3±158.03 mg/l; ammonia-nitrogen, 37.3±10.73 mg/l; nitrate-nitrogen, 3.6±2.54 mg/l; and MRP, 4.1±1.89 mg/l. The corresponding effluent of cell 1 is as follows: 35.6±29.59 mg/l for BOD, 127.3±71.00 mg/l for COD, 30.6±44.06 mg/l for SS, 18.4±7.53 mg/l for ammonia-nitrogen, 1.3±1.13 mg/l for nitrate-nitrogen, and 3.2±1.24 mg/l for MRP.

In comparison, the influent (i.e., effluent from a septic tank) to the ICW in Dunhill has the following water quality characteristics: 358.4±200.57 mg/l for BOD, 554.4±288.19 mg/l for COD, 303.5±335.46 mg/l for SS, 52.6±39.30 mg/l for ammonia-nitrogen, 0.6±1.74 mg/l for nitrate-nitrogen, and 7.8±3.38 mg/l for MRP. An improvement due to treatment in wetland cell 1 was noticed: 55.3±28.65 mg/l for BOD, 149.6±69.80 mg/l for COD, 42.8±36.71 mg/l for SS, 49.9±37.92 mg/l for ammonia-nitrogen, 1.1±1.37 mg/l for nitrate-nitrogen, and 7.0±4.69 mg/l for MRP.

These findings indicate that variables including BOD, COD, and SS were significantly reduced within

Table 9 Seasonal mean nutrient removal efficiencies for the integrated constructed wetland in Dunhill

Variables	2001			2002			2003			2004			2005		
	n	Mean	SD	*n*	Mean	SD		Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Biochemical oxygen demand															
Inflow (mg/l)	3	730.0	628.33	9	346.0	200.10	9	286.0	186.90	2	335.0	22.63	–	–	–
Effluent (mg/l)	4	18.7	7.37	9	21.7	11.87	8	9.1	6.67	4	24.5	21.21	1	11.5	3.54
Removal (%)	97.0			93.7			96.8			92.7			–		
Chemical oxygen demand															
Inflow (mg/l)	3	996.8	255.41	9	507.0	287.28	10	438.7	193.18	2	700.0	282.24	1	520.0	–
Effluent (mg/l)	4	46.3	18.87	9	34.0	16.82	10	58.1	19.78	4	8.0	25.82	6	105.57	39.47
Removal (%)	95.4			93.3			86.8			88.6			79.7		
Suspended solids															
Inflow (mg/l)	3	381.7	243.73	9	348.8	412.62	9	287.7	350.52	2	139.0	50.91	1	134.0	–
Effluent (mg/l)	4	6.8	3.77	9	10.6	7.13	9	9.2	7.85	2	4.0	5.66	1	7.0	–
Removal (%)	98.2			97.0			96.8			97.1			94.8		
Ammonia-nitrogen															
Inflow (mg/l)	3	32.6	20.69	9	40.1	29.47	8	72.8	54.69	3	64.0	2.00	1	30.0	–
Effluent (mg/l)	4	3.0	2.52	9	3.5	3.16	8	9.2	10.12	12	24.9	10.55	26	33.6	9.08
Removal (%)	90.6			91.2			87.3			61.2			−12.2		
Nitrate-nitrogen															
Inflow (mg/l)	4	0.1	0.1	1	0.1	–	1	0.1	–	2	0.1	0.1	1	4.5	–
Effluent (mg/l)	4	0.1	0.1	1	1.3	–	1	0.1	–	2	0.25	0.21	1	3.94	–
Removal (%)	0			−1,200.0			0.0			−150.0			12.6		
Molybdate-reactive-phosphorus															
Inflow (mg/l)	3	8.2	0.17	9	6.8	3.06	9	8.6	4.59	3	9.1	1.63	1	5.9	–
Effluent (mg/l)	4	1.1	0.33	9	2.6	2.34	9	2.5	1.82	12	6.2	1.14	26	7.3	2.28
Removal (%)	86.2			62.2			70.3			32.4			−24.2		

n sample number, *SD* standard deviation

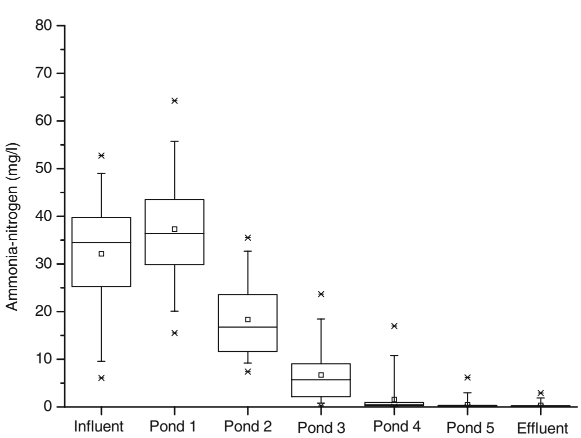

Fig. 3 Ammonia-nitrogen concentrations for the integrated constructed wetland system in Glaslough

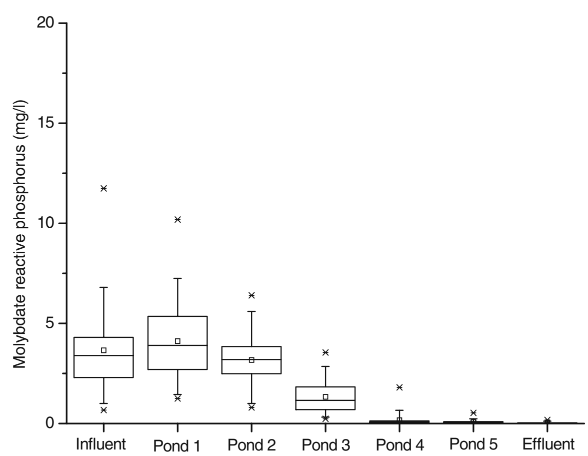

Fig. 4 Molybdate-reactive-phosphorus concentrations for the integrated constructed wetland system in Glaslough

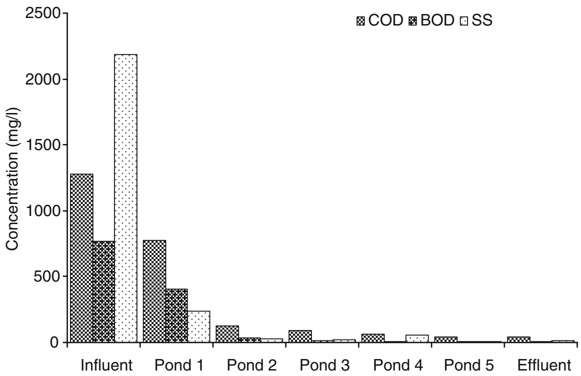

Fig. 5 Biochemical oxygen demand (*BOD*), chemical oxygen demand (*COD*), and suspended solids (*SS*) concentrations for the integrated constructed wetland system in Glaslough

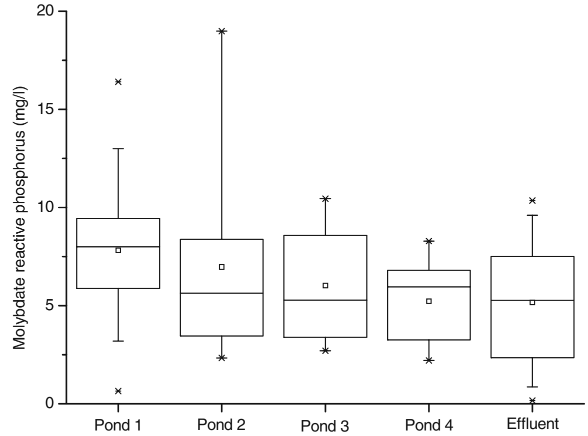

Fig. 7 Molybdate reactive phosphorus concentrations for the integrated constructed wetland system in Dunhill

the first cell of both systems even after 5 years of ICW operation in Dunhill. Nitrate-nitrogen and MRP concentrations reduced significantly within the first cell, whereas the ammonia-nitrogen reduction rate was higher in wetland cell 2 than in cell 1. However, the ICW in Glaslough had a higher pollutant reduction capacity than the ICW in Dunhill. This is most likely due to overloading of the ICW system in Dunhill.

As can be seen from Figs. 3, 4, and 5, ammonia-nitrogen and MRP were significantly ($p<0.05$) removed after the contaminated water passed approximately 30% of the ICW area in Glaslough, whereas the COD, BOD, and SS were reduced after passing 20% of the ICW area. This can be explained by the low COD/BOD ratio of 1.45, which means that most of the pollutants in the contaminated water are biodegradable. Concerning the ICW in Dunhill, nutrient reduction rates were low in the first cell (Figs. 6 and 7). The BOD, COD, and SS concentrations were mostly reduced within the first cell (Fig. 8).

Kadlec et al. (2000) and Carty et al. (2008) state that nutrient reductions occur predominantly within the initial wetland cells (as confirmed in this study; except for MRP) and that pollutants are reduced effectively if the hydraulic retention time is relatively high. This is promoted by allowing the pollutant plume to spread as slowly as possible throughout flatly designed ICW cells.

The nitrate-nitrogen concentrations within the ICW systems were low. It is likely that nitrate and oxygen provided electron acceptors in the lower layer of the

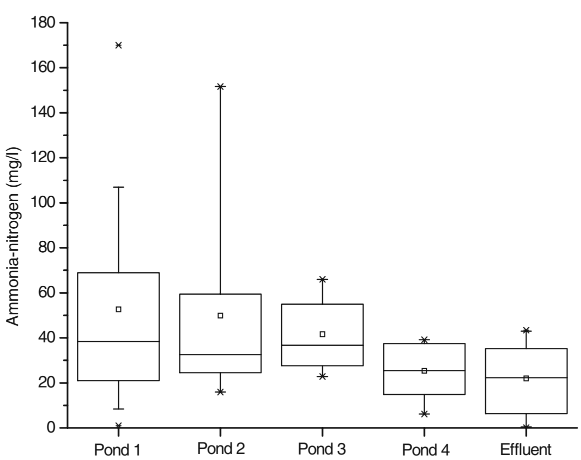

Fig. 6 Ammonia-nitrogen concentrations for the integrated constructed wetland system in Dunhill

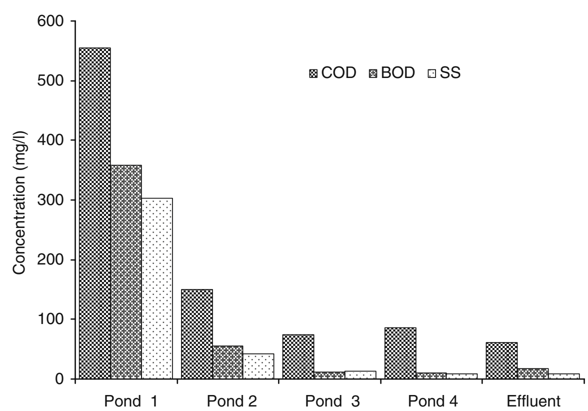

Fig. 8 Biochemical oxygen demand (*BOD*), chemical oxygen demand (*COD*), and suspended solids (*SS*) concentrations for the integrated constructed wetland system in Dunhill

wetland cells (Eriksson and Weisner 1996). Furthermore, Nielsen et al. (1990) reported that the high number of denitrifying bacteria was dependent on the accumulation of plant detritus within ICW systems. The overall heterotrophic activity is increased by the supply of sufficient organic matter. This leads to the consumption and subsequent reduction of oxygen within the sediment, thus supporting denitrification.

4 Conclusions and Further Research Needs

The ICW concept has been successfully applied for the first time on an industrial scale for the treatment of domestic wastewater in real case studies that were fully scientifically monitored and assessed. The new and mature ICW systems successfully removed traditional pollutants such as BOD from domestic wastewater. Concerning the new ICW system (1 year of operation) in Glaslough, the nutrient reduction efficiencies are significantly high, whereas the nutrient reduction efficiencies (including nitrate nitrogen and molybdate reactive phosphorus) started to decrease after 5 years of operation due to overloading of the mature ICW. However, the biochemical oxygen demand, chemical oxygen demand, suspended solids, and ammonia-nitrogen concentrations were reduced within the mature ICW system even after approximately 5 years of operation. However, while nitrification of ammonia-nitrogen was significant, the denitrification rate started to decrease as the ICW matured.

Both groundwater and surface water monitoring results indicated that the ICW system in Glaslough had neither polluted the groundwater nor decreased the water quality of the receiving watercourse. All nutrient concentrations for the receiving watercourse were lower downstream than upstream of the ICW system outlet. On the other hand, nutrient removal within the open ICW system is complex due to water, sediment, plant, and microbial interactions, so it was impossible to come up with consistent nutrient balances.

The novel use of ICW to treat domestic wastewater is a valuable and appropriate technology. It is especially suitable for small communities in both developed and developing countries. The absence of an artificial liner made of materials such as plastic or concrete makes the ICW technology affordable. However, any ICW system should be mature and sufficiently large to avoid potential groundwater contamination.

There is scope for further research on the assessment of water balances and processes responsible for the self-sealing effect observed in mature wetlands. A detailed assessment of the microbial population dynamics and the role of species influencing the treatment performance would therefore be beneficial.

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