

Chapter 2

Wetland Case Studies

Abstract This chapter comprises six sections providing an overview of various representative and timely wetland case studies. The focus is on the treatment of domestic wastewater, farmyard runoff, swine wastewater, wood storage site runoff, and produced water containing hydrocarbons. Section 2.1 covers integrated constructed wetlands (ICW) for treating domestic wastewater. Section 2.2 provides guidelines for farmyard runoff treatment with farm constructed wetlands (FCW), a sub-type of ICW. Moreover, Section 2.3 covers a representative FCW case study on farmyard runoff treatment, and Section 2.4 describes ICW treating swine wastewater. Section 2.5 reviews wetlands controlling runoff from wood storage sites. Finally, Section 2.6 comprises a study reporting on wetlands for the treatment of hydrocarbons. All sections look at wetland systems from a holistic perspective, highlighting recent research findings relevant for practitioners. This chapter does not cover wetland systems such as sustainable (urban) drainage systems and sustainable flood retention basins used predominantly for flood control purposes. These techniques are discussed in detail in Chapter 4.

2.1 Integrated Constructed Wetlands for Treating Domestic Wastewater

2.1.1 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 (Scholz *et al.* 2002; 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 the tannery and textile industries, slaughterhouses, pulp and paper production, agriculture (animal farms and fish farm effluents), and various runoff waters (agriculture, airports, highways, and storm water) (Kadlec *et al.* 2000; Haberl *et al.* 2003; Scholz 2006a; Vymazal 2007; Carty *et al.* 2008).

The concept of constructed wetlands applied to 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 capital 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 vertical-flow constructed wetlands (Scholz 2006a)) is permanently kept above the wetland soils to create fully saturated soil conditions, generally resulting in high contaminant removal efficiencies. The treatment efficiencies of wetlands varies 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 ICW concept was developed not only to address water pollution from different sources including domestic, industrial, and agricultural, but also to provide ecological services by restoring potentially lost environmental infrastructure including wetlands. The main features of ICW 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.* (2007a, b) 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 2006a; 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 5-d 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 could be achieved after a few growing seasons because of the lack of well-developed below- 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 were 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; and
- to investigate the impacts of potential contamination of nearby surface waters and groundwater, taking into consideration that an artificial liner is not present.

2.1.2 Materials and Methods

2.1.2.1 Site Description

The case study systems comprise two ICW for treating domestic wastewater in Ireland. The ICW in Glaslough (Figures 2.1 and 2.2) is situated in the County of Monaghan (northern part 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 approx. 970 mm over the last 50 years. However, the mean annual rainfall of 1256 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 Water River. The inflow rate ranges between approx. 85 and 105 m³/d. The corresponding outflow (approx. between 1 and 50 m³/d) 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 equivalent to 1750 inhabitants 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 (Figure 2.1) 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.)

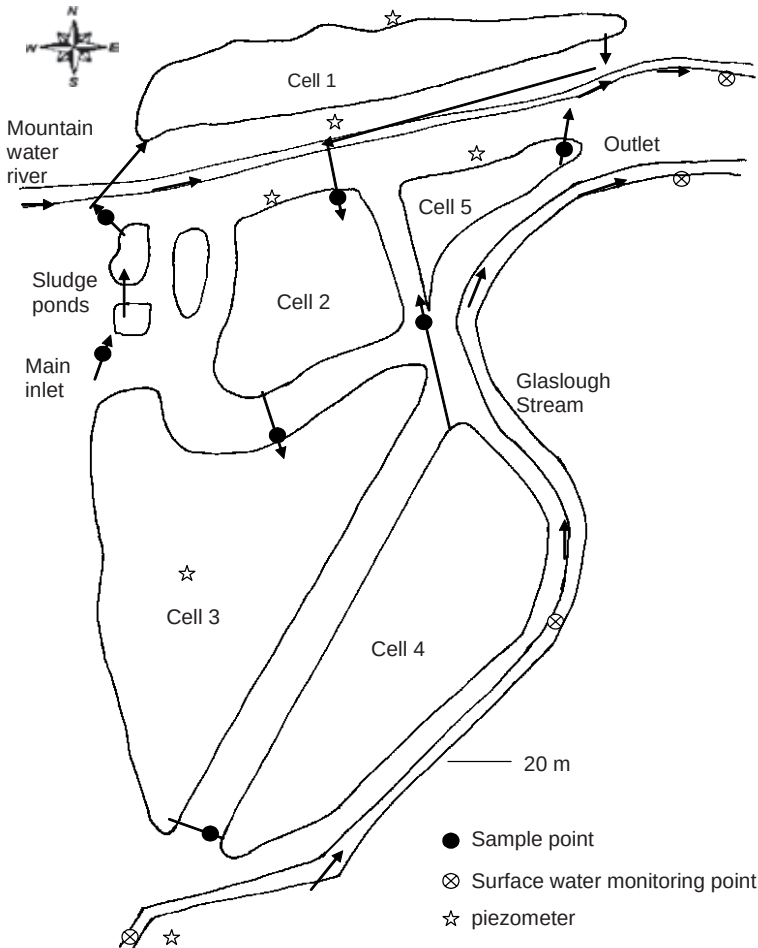


Figure 2.1 Sketch showing groundwater and surface water monitoring and inlet and outlet points for the integrated constructed wetland in Glaslough, near Monaghan, Ireland

Holmb., *Glyceria fluitans* (L.) R. Br., *Juncus effusus* L., *Sparganium erectum* L. emend Rchb, *Elisma natans* (L.) Raf. and *Scirpus pendulus*, Muhl. The ICW is flanked by the Mountain Water River and the Glaslough Stream.



Figure 2.2 Wetland cell in Glaslough, near Monaghan, Ireland

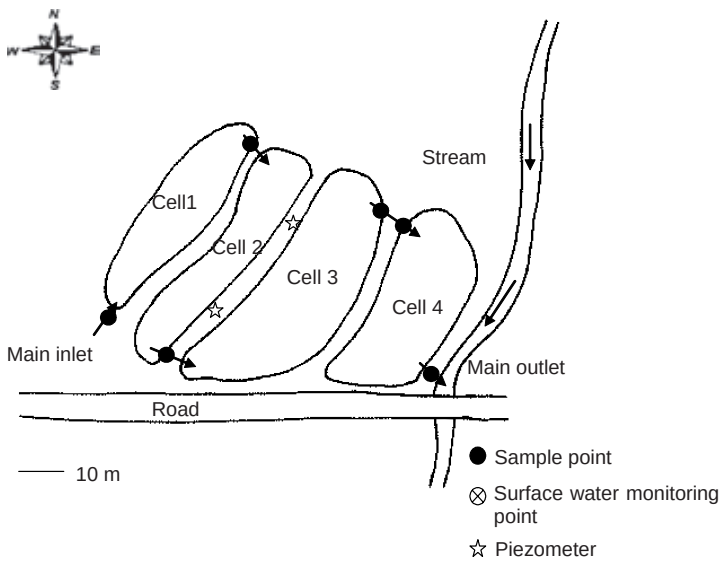


Figure 2.3 Sketch showing groundwater and surface water monitoring and inlet and outlet points for the integrated constructed wetland system in Dunhill, near Waterford, Ireland

The ICW system in Dunhill (County Waterford, southeastern part of Ireland) is situated at a longitude of 07°02'40"W and a latitude of 52°11'28"N (Figures 2.3 and 2.4). The typical annual rainfall is approx. 1000 mm. Until 2000, sewage at Dunhill village was directed to a wastewater treatment plant (septic tank system).



Figure 2.4 Integrated constructed wetland system in Dunhill, near Waterford, Ireland

In late 2000, the system was upgraded with the help of an ICW system that was fully operational by February 2001. The wastewater inflow was approx. $40 \text{ m}^3/\text{d}$. The corresponding outflow was roughly $24 \text{ m}^3/\text{d}$. Dilution of the wastewater due to rainfall is approx. 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 therefore has no energy consumption. Wastewater from households is collected via the sewage 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.

2.1.2.2 Sampling and Analytical Methods

Grab samples for the inlet and outlet of each wetland cell were taken approx. quarterly at the ICW in Dunhill, while a substantial suite of hi-tech automatic sampling and monitoring instrumentation has been used for approx. weekly sampling at the ICW in Glaslough: ISCO 4700 Refrigerated Automatic Wastewater Sampler (Teledyne Isco, Nebraska, USA), Siemens Electromagnetic Flow Meter F M MAGFLO and MAG5000 (Siemens Flow Instruments A/S, Nordborgvej, Nordborg, Denmark). Furthermore, both the flow rates (Figure 2.5) and key weather parameters (Figure 2.6) were automatically recorded.

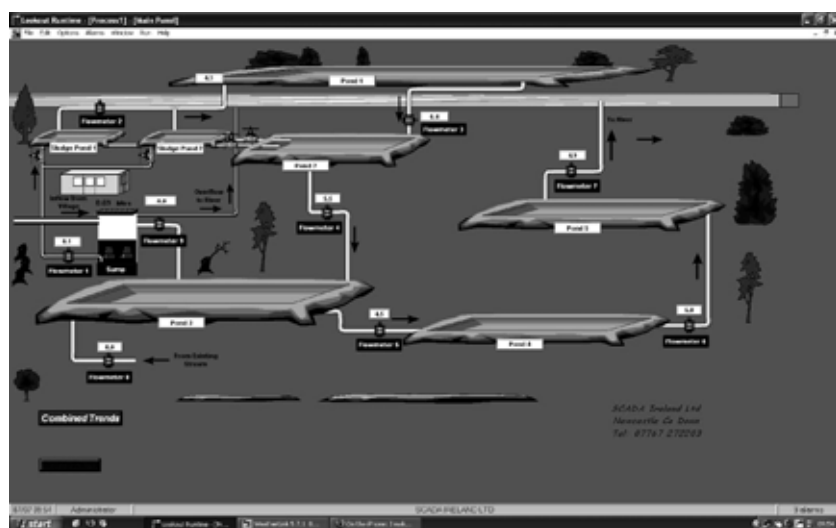


Figure 2.5 Computer screenshot of automatic flow recording software output in Glaslough, Ireland

Furthermore, the Mountain Water River and Glaslough Stream were also monitored (Figure 2.1). Water samples were analyzed for variables including the BOD, COD, SS, pH, ammonia nitrogen, nitrate–nitrogen, and molybdate reactive phosphorus (MRP; equivalent to SRP) at the Monaghan County Council water laboratory using American Public Health Association standard methods (APHA 1998) unless stated otherwise.

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

Six piezometric groundwater-monitoring wells (Figure 2.1) 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 through the wetland cells (Kadlec and Knight 1996; Scholz 2006a; Wallace and Knight 2006).

The water table at the ICW site is relatively high (*i.e.*, 1.8 to 2.0 m below the ICW beds), so it is very important to monitor the ICW system and ensure that it has no negative effect on the groundwater. Six piezometers were 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 approx.

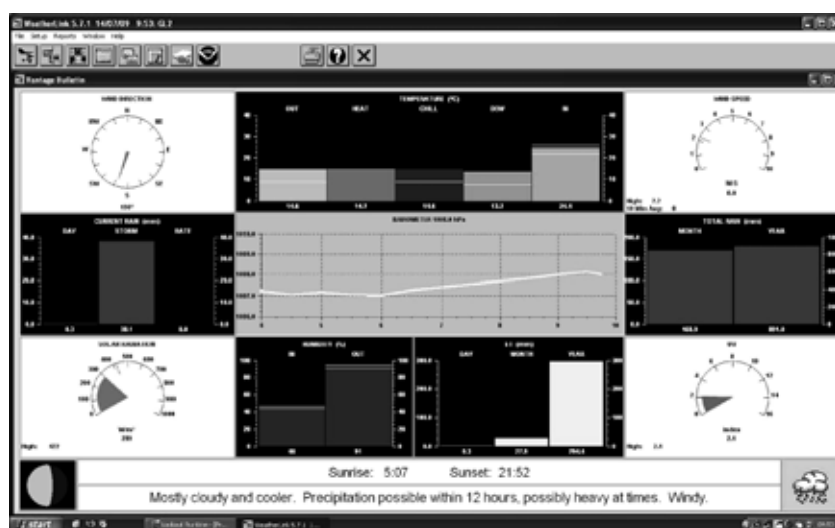


Figure 2.6 Computer screenshot of automatic weather station software output in Glaslough, Ireland

9×10^{-11} m/s. Piezometer 1 near wetland cell 1 (close to a small hill) and piezometer 6 (located across the Glaslough stream) are outside of the ICW system (Figure 2.1).

For the ICW system in Dunhill, two piezometric groundwater-monitoring wells were sampled at a depth of 5 m (Figure 2.3). The wells were placed within the ICW system and along the suspected flow path of contaminants to assess the risk of groundwater contamination. The subsequent water quality analysis for both ICW systems was carried out according to APHA (1998).

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 was not polluting the receiving waters. The Mountain Water River is sampled at two locations, one upstream and one approx. 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 with discharge standards. The two sampling points are located approx. 4 km upstream and 3.5 km downstream of Dunhill village.

2.1.2.3 Statistical Analyses

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

2.1.3 Results and Discussion

2.1.3.1 Water Quality of the ICW System in Glaslough

The approximate mean inflow values in Glaslough were as follows: BOD, 768 ± 451.0 mg/l; COD, 1279 ± 697.8 mg/l; SS, 2184 ± 3844.8 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 (Kayranli *et al.* 2010).

However, the ICW system has shown a very good treatment performance, despite being a new (*i.e.*, not mature) system. The ICW system removed approx. 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 the constructed wetland's age (Kadlec 1999). An increase in age is also associated with an increase in microbial popula-

tion. Furthermore, microbial biofilm formation on the bed material within the wetland leads to higher biological degradation rates (Picard *et al.* 2005).

There is very little information on full-scale representative constructed wetlands for treating domestic wastewater in the scientific literature. However, numerous studies refer to pilot-scale and microcosm-scale constructed wetlands for treating domestic wastewater. Ciria *et al.* (2005) assessed the role of *Typha latifolia* L. (reedmace, cattail, or bulrush) in constructed wetlands of 40 m² each, filled with gravel as the supporting medium. Their study showed that BOD and COD removal efficiencies were $97 \pm 1.2\%$ and $79 \pm 0.3\%$ in the first year, respectively, and in the second year, the BOD removal efficiency did not change ($97 \pm 3.0\%$), while the COD removal efficiency increased slightly ($81 \pm 1.0\%$). Furthermore, a study by Hamouri *et al.* (2007) achieved removal efficiencies of 78% for COD and 79% for BOD with respect to the combination of a two-step up-flow anaerobic reactor and sub-surface horizontal-flow constructed wetland (Hamouri *et al.* 2007). Between 71 and 75% removal efficiencies for COD and BOD were noted for constructed wetlands treating secondary treated sewage (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 (Newman *et al.* 1999; Yang *et al.* 2001; Scholz 2006a; Wallace and Knight 2006; Mitsch and Gosselink 2007). Kadlec *et al.* (2000) have explained that interactions between nitrogen on one hand and water, sediment, plant, and biomass on the other make it difficult to assess the real efficiency of nitrogen removal due to storage in the system.

The overall ammonia–nitrogen reduction is relatively high in comparison to other microcosm wetlands treating domestic wastewater: 76 to 92% for sub-surface constructed mangroves in Hong Kong (Wu *et al.* 2008); 52% for a combination of free surface-flow wetland cells that were fed with municipal lagoon effluents in Canada (Cameron *et al.* 2003); 10 to 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 sub-surface horizontal-flow constructed wetland in Morocco (Hamouri *et al.* 2007); and 14 to 24% for a constructed wetland treating secondary treated sewage in Australia (Thomas *et al.* 1995).

Seasonal variations in performance related to nutrient removal were also investigated. 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 in the influent was lower in spring compared to 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 with concentrations (52.6 ± 53.48 mg/l; removal efficiency 96%) in spring. In contrast, 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 nitrification rates within constructed wetlands (Kayranli *et al.* 2010). 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 rather 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 the daytime and supports the oxygen demand for stabilization of organics and nitrification. Plant rhizosphere aeration may stimulate aerobic decomposition processes, increasing nitrification and subsequent gaseous losses of nitrogen via denitrification and decreasing 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-dependent process. Werker *et al.* (2002) reported 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 sharply affected the system performance due to changing biological activities with temperature.

Ciria *et al.* (2005) reported 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 $22 \pm 0.8\%$ for ammonia–nitrogen and $64 \pm 0.9\%$ for nitrate–nitrogen in autumn of the first year and $40 \pm 3.5\%$ for ammonia–nitrogen in summer and $75 \pm 2.1\%$ for nitrate–nitrogen in winter for the second year. In comparison to the previous study, the ammonia–nitrogen and nitrate–nitrogen removal efficiencies for all seasons are significantly higher for the ICW system.

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 sediments (Bonomo *et al.* 1997). In contrast, Sakadevan and Bavor (1998) suggested that long-term phosphorus removal mechanisms in constructed wetland systems are likely due to uptake by the substratum, litter, and aluminum/iron compounds, while plant uptake is often a relatively small fraction. Furthermore, Kadlec (1999) reported 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 capacity and high plant uptake capacity. However, the system is relatively young as well.

2.1.3.2 Receiving Stream Water Quality

The Mountain Water River is adjacent to the sludge cells and to wetland cells 1, 2, and 5, whereas the Glaslough Stream is adjacent to wetland cells 4 and 5. 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. The river water quality did not change in the downstream compared to the upstream stretch. This indicates that the river has sufficient assimilative capacity.

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 depends predominantly on precipitation patterns. 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 mg/l, 1.0 ± 0.41 mg/l, and 0.1 ± 0.06 mg/l for the upstream and 0.4 ± 0.28 mg/l, 1.0 ± 0.38 mg/l, 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 mg/l, 1.2 ± 0.37 mg/l, 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 mg/l, 0.9 ± 0.44 and 0.8 ± 0.50 mg/l, 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 Phosphorus 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.

Concerning the Annestown Stream, the MRP concentration at the monitoring station Ballyphilip (4 km upstream of the ICW system) was well below the target phosphorus concentration of 0.03 mg/l, set by the Irish Phosphorus Regulations from 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 insignificant. The increase in nutrient concentration 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.

2.1.3.3 Groundwater Quality

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. Piezometer 2 is located in the west of the ICW system near cell 1, whereas 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 mg/l, 0.2 ± 0.10 mg/l, and 0.4 ± 0.31 mg/l, which indicate a slight increase in concentrations. 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 mg/l and 0.3 ± 0.28 mg/l, 0.8 ± 0.51 mg/l and 0.5 ± 0.55 mg/l, and 0.4 ± 0.53 mg/l 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 mg/l, 0.9 ± 0.83 mg/l, 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.

In Dunhill, piezometer 1 near wetland cell 2 and piezometer 2 near cell 3 are located within the ICW system to monitor groundwater quality (see also Figure 2.3). Concerning piezometer 1, 4.7 ± 12.25 mg/l 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 mg/l 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 because of overloading. 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 (Kayranli *et al.* 2010).

2.1.3.4 Comparison of Nutrient Removal Performances

Overall, both ICW systems indicate significant ($p < 0.05$) COD and BOD removal efficiencies. Concerning the other water quality data, ammonia–nitrogen, nitrate–nitrogen, and MRP removal efficiency 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. 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 three 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 also has 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 either system. This is probably due to the increase in organic loading rate as a consequence of an increase in the evaporation rate and a decrease in the precipitation rate. On the other hand, effluent ammonia–nitrogen, nitrate–nitrogen, and MRP concentrations did not change significantly 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 sediment, which may not have reached the saturation threshold (Kayranli *et al.* 2010).

2.1.3.5 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; 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; 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; 7.0 ± 4.69 mg/l for MRP.

These findings indicate that variables including BOD, COD, and SS were significantly reduced within the first cell of both systems even after 5 years of ICW operation in Dunhill. Nitrate–nitrogen and MRP concentrations were 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 that in Dunhill. This is most likely due to overloading of the ICW system in Dunhill.

As can be seen from Figures 2.7–2.9, ammonia–nitrogen and MRP were removed in significant ($p < 0.05$) amounts after the contaminated water passed approx. 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 pollutants in the contaminated water are biodegradable. Concerning the ICW in Dunhill, nutrient reduction rates were low in the first cell (Figures 2.10 and 2.11). The BOD, COD, and SS concentrations were mostly reduced within the first cell (Figure 2.12).

Kadlec *et al.* (2000) and Carty *et al.* (2008) report 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 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.

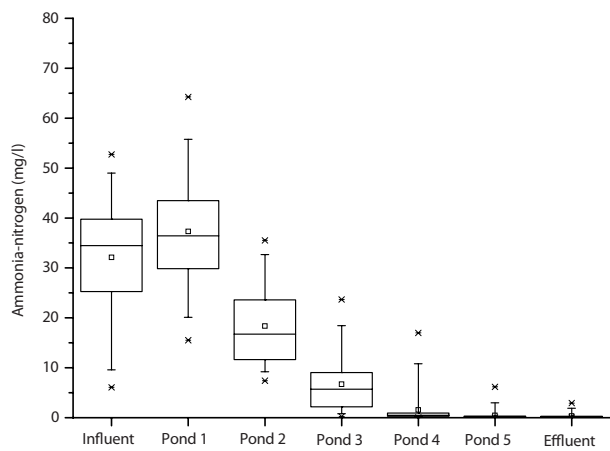


Figure 2.7 Ammonia–nitrogen concentrations for the integrated constructed wetland system in Glaslough, Ireland

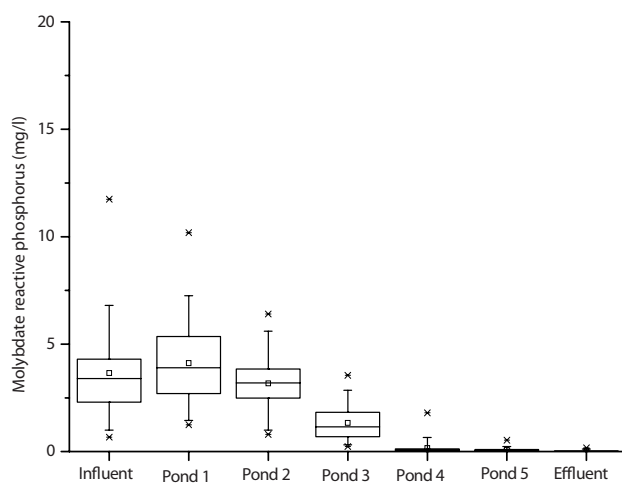


Figure 2.8 Molybdate reactive phosphorus concentrations for the integrated constructed wetland system in Glaslough, Ireland

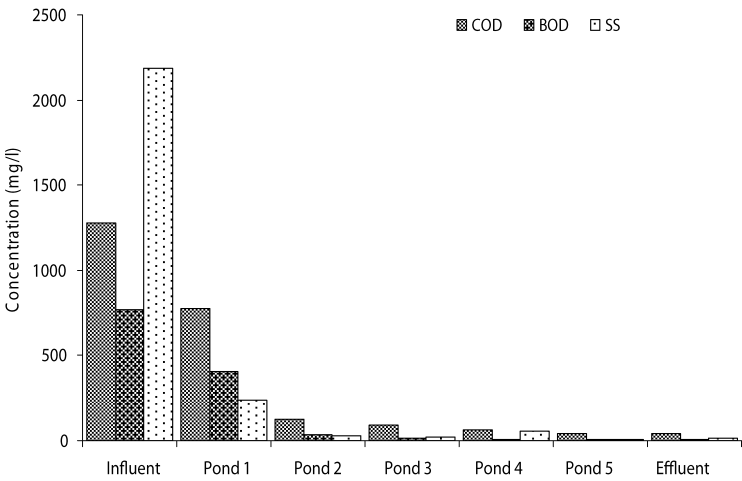


Figure 2.9 Biochemical oxygen demand (BOD), chemical oxygen demand (COD), and suspended solid (SS) concentrations for the integrated constructed wetland system in Glaslough, Ireland

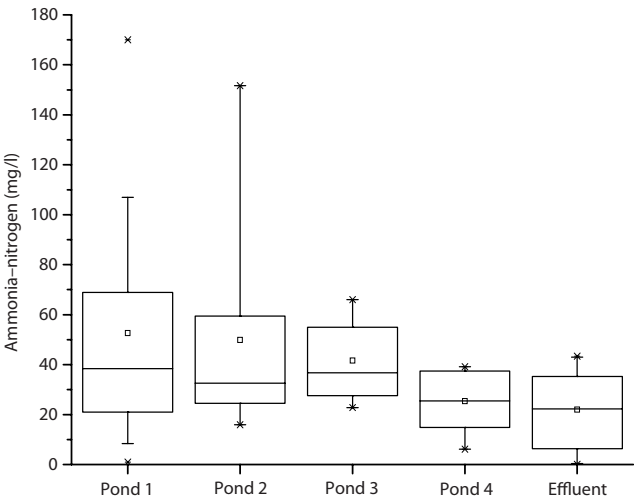


Figure 2.10 Ammonia–nitrogen concentrations for the integrated constructed wetland system in Dunhill, Ireland

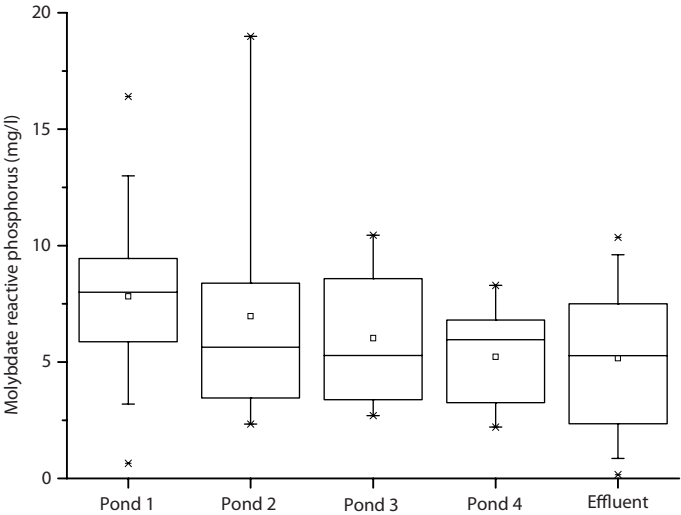


Figure 2.11 Molybdate reactive phosphorus concentrations for the integrated constructed wetland system in Dunhill, Ireland

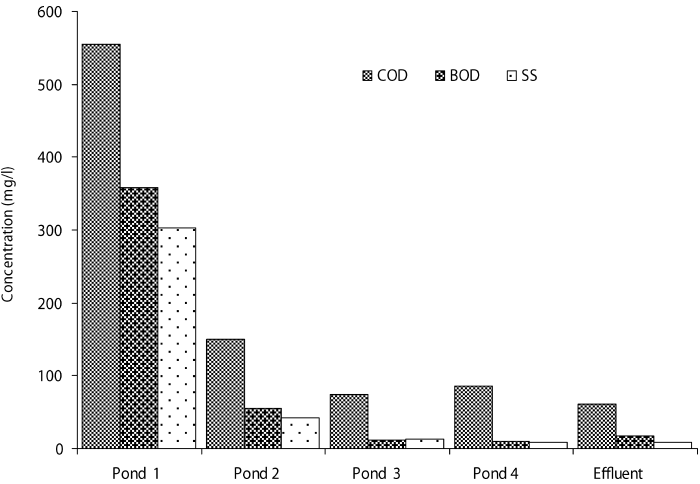


Figure 2.12 Biochemical oxygen demand (BOD), chemical oxygen demand (COD), and suspended solid (SS) concentrations for the integrated constructed wetland system in Dunhill, Ireland

2.1.4 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 MRP) started to decrease after 5 years of operation due to overloading of the mature ICW. However, the BOD, COD, SS, and ammonium–nitrogen concentrations were reduced within the mature ICW system even after approx. 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 treatment performance would therefore be beneficial.

2.2 Guidelines for Farmyard Runoff Treatment with Wetlands

2.2.1 Introduction

The loss of nutrients and contaminants from agricultural land, farmyards (Figure 2.13), dairy parlors, tracks, and roofs to rivers, lakes, and groundwater can have a detrimental impact on water quality. Both point and diffuse sources of pollution from agriculture contribute to the degradation of water quality and aquatic ecosystems (e.g., fish kills and loss of habitat) through eutrophication, contamination of groundwater, siltation, and direct toxicity to organisms, which



Figure 2.13 Typical farmyard in Scotland

consequently affect biodiversity, fisheries, recreation, and public health. They also affect farmers, exposing them to fines and prosecutions, and the wider community by the subsequent degradation and loss of water supply within affected watersheds (Harrington and Ryder 2002; Mantovi *et al.* 2003; Scholz *et al.* 2007a, b).

There is a wide range of laws and policies at national and international levels (*e.g.*, Zedler 2003 and Scholz 2006) that aim to control and mitigate the risks of water pollution caused by agricultural contaminants. Several BMP are available for farmyard runoff treatment as part of “treatment trains” (Scholz 2006a, b); *i.e.*, sets of measures that range from pollution source control to dirty water collection and treatment. The most common measures implemented at the farm scale include the following (Rice *et al.* 2002; Hilton 2003):

- animal diet improvement to reduce nutrient losses;
- roofing of silage pits and areas of farmyards where excrement is expected to accumulate;
- clean roof water diversion to drains to reduce the volume of dirty water to be stored and spread;
- upgrading of buildings and of manure, slurry, fuel, and pesticide storages to avoid leaks and spillages;
- basins and biobeds for pesticide wash water storage and treatment; and
- swales, buffer strips, and wetlands to store and clean farmyard runoff before discharge to watercourses (Poe *et al.* 2003).

In addition to the previously mentioned BMP, farm constructed wetlands (FCW) are recommended for the collection and treatment of farmyard runoff, before it is released into watercourses, to protect surface water and groundwater resources (Scholz *et al.* 2007a, b). The US and New Zealand approaches to FCW have been summarized by USEPA (1995a, b) and NIWA (1997), respectively.

2.2.2 Farm Constructed Wetlands: Definition and Background

2.2.2.1 Introduction

An FCW is defined as an ecologically engineered system comprising a series of shallow, free-surface-flow constructed wetland cells (see below) containing emergent vegetation, which is designed to receive and treat farmyard runoff (Carty *et al.* 2008; Scholz *et al.* 2007a, b). Most FCW are being developed to benefit both the environment and farmers by reducing the impact of potential pollution incidents, helping to manage farm effluents, and enhancing habitat, biodiversity, and landscape in a way that is practical, efficient, affordable, and cost-effective (Harrington *et al.* 2005). Typical FCW design and operation contrast with those of constructed reed beds, which can be seen as being ‘closer’ to the traditional wastewater treatment philosophy based predominantly on civil engineering principles (Mantovi *et al.* 2003; Kantawanichkul and Somprasert 2005; Sun *et al.* 2006).

The design approach for the construction of farm wetlands proposed in this universal guideline for temperate climate is largely inspired by the ICW concept (Harrington and Ryder 2002; Scholz *et al.* 2007a, b) pioneered in Ireland by the National Parks and Wildlife Service (Department Environment Heritage and Local Government, Ireland) and is based upon data on performance over the last 10 years of 13 ICW constructed within the Anne Valley in southern Ireland. The ICW concept is based on the following principles (Harrington *et al.* 2005):

- the containment and treatment of influents within shallow vegetated ponds using local soil material, wherever possible;
- the aesthetic placement of the containing wetland structure into the local landscape (“landscape fit”) to enhance a site’s ancillary values; and
- the enhancement of habitat and biodiversity.

The design endeavors to optimize natural biological, chemical, and physical processes of pollutant removal in a way that does not produce a negative impact on aquatic and terrestrial ecosystems (Scholz *et al.* 2007a, b).

2.2.2.2 Effluent Types and Processes

All FCW can be designed to treat various types of wastewater from farms including dairy, machinery, vegetable and mushroom washings, runoff from silos, yards and other areas of hard-standing (usually only lightly contaminated by manure or silage), and dairy cow access tracks (Poe *et al.* 2003; Carty *et al.* 2008; Scholz *et al.* 2007). However, FCW are usually not designed for the treatment of more nutrient-rich effluent types such as slurries, raw milk, and washings from pesticide sprayer and dipping equipment.

Within an FCW, the contaminated effluent is treated through various physical, chemical, and biological processes involving plants (predominantly macrophytes),

algae, microorganisms, water, soil, and sun (e.g., direct process via photodegradation). The main processes for which detailed examples can be found elsewhere (Carty *et al.* 2008; Mantovi *et al.* 2003; Scholz and Xu 2002; Scholz and Lee 2005; Scholz 2006a; Scholz *et al.* 2007a, b; Zedler 2003) are as follows:

- physical filtration of suspended solids by wetland vegetative biomass acting as a hydrological baffle to incoming flows (optimized by high vegetation density and low flow velocity);
- settling of suspended particulate matter by gravity (optimized by low flow velocity, low wind speed, low disturbance, and long residence time);
- uptake, transformation and breakdown of nutrients, hydrocarbons and pesticides by biomass, plants and microbes (increased by a relatively high temperature, long residence time, contact with microorganisms and plants, high microorganism and plant density, and a relatively high organic matter content);
- accumulation and decomposition of organic matter, which is important for nutrient cycling (optimized by low velocity and availability of adsorption sites on suitable aggregates);
- microbial-mediated processes such as nitrification (aerobic) and denitrification (anaerobic), which are important for the cycling of nitrogen (see also Poe *et al.* (2003));
- chemical precipitation and sorption of nutrients such as phosphorus (see also Braskerud (2002)) by soil (influenced by the availability of sorption sites, pH, and redox potential); and
- predation and natural dieoff of pathogens (e.g., *Escherichia coli* and *Cryptosporidium*) (optimized by high diversity and density of natural predators (e.g., protozoa), and increased exposure to ultraviolet light).

2.2.2.3 Functions, Values, and Principles

The profiles and infrastructural details required to support habitat development and biodiversity enhancement are, on the basis of experience, best addressed in the design and construction of the wetland. This is particularly relevant to the development of transitional habitats between the terrestrial embankment and wetland zones (Scholz and Lee 2005; Scholz 2006a).

Wide, shallow, and low elevated embankments promote floral and faunal diversity. Shallow and deep wetland areas with either south- or north-facing aspects are also important to enhance the habitat. Local vegetation is best incorporated wherever possible. Care should be taken when locating an FCW immediately adjacent to woodland as problems may arise from shading and seepage of water via root systems. Many FCW can be dynamic ecosystems and their habitats may change unless managed and maintained. For example, the management of water depth to facilitate optimal water treatment is particularly important in this regard (Dunne *et al.* 2005a, b; Scholz 2007; Scholz *et al.* 2007a, b; see also below).

Wetlands, including FCW, can have considerable aesthetic appeal. The combination of water, vegetation, and associated wildlife are the principal elements of visual enhancement, and there are many examples of this throughout the world. The aesthetic appeal of an FCW can be maximized through appropriate land-forming design that is implemented during construction (Scholz 2007).

The process of design ensures that the final wetland structure ‘fits’ well into the landscape, *e.g.*, by designing curvilinear enclosing embankments that conform to a site’s topography. Subsequent vegetation development will further enhance the visual natural appearance of the system. Appropriate land forming of the structure to fit the landscape also reduces FCW maintenance, thus enhancing a variety of amenity values and improving its sustained functionality (Dunne *et al.* 2005a, b; Scholz 2006b; Scholz *et al.* 2007a, b).

2.2.2.4 Benefits of Farm Constructed Wetlands

The main benefits of FCW as discussed by Zedler (2003), Scholz and Xu (2002), Scholz *et al.* (2007), and other researchers (see below) are summarized below:

- High level of treatment and robustness: efficient treatment (up to 99% reduction by concentration) of contaminants such as phosphorus (Braskerud 2002), nitrate–nitrogen, nitrite–nitrogen, ammonia–nitrogen, BOD, suspended solids, hydrocarbons, and pathogens.
- Runoff and flood management: FCW are designed sufficiently large to manage heavy rainfall events, providing attenuation for increased effluent volumes during storm events. By functioning as ‘buffer zones’ and attenuating peak flows, they contribute to the reduction of flooding incidents downstream (Scholz and Lee 2005; Scholz 2007).
- Relatively low cost and simplicity of operation: FCW have minimal equipment needs and little, if any, energy use since water can be transferred by gravity through the system. They are also simple to operate and more cost-effective than alternative methods of disposing of farm runoff (Poe *et al.* 2003).
- Odor minimization: odor can be a serious problem when handling and treating agricultural wastes. Odors are minimized in FCW through the use of a dense plant cover, appropriate plant species, a shallow water level, and surface-flow maintenance.
- Aesthetically pleasing: FCW enhance the landscape by adding colors and texture and by increasing the diversity of plants and habitats (Scholz 2007).
- Habitat and biodiversity enhancement: FCW provide habitats for a wide variety of birds, mammals, reptiles, amphibians, and invertebrates (*e.g.*, Froneman *et al.* 2001).
- Contingency measures: FCW can help to mitigate the impact of accidental spillages, acting as buffer zones and giving time to implement emergency measures.

2.2.2.5 Limitations of Farm Constructed Wetlands

All FCW have some limitations, although these can be managed by designing and maintaining FCW for their design goals, as outlined in this sub-section. The main limitations as discussed by Scholz *et al.* (2007) and Carty *et al.* (2008) are as follows:

- Some FCW have a relatively large land requirement compared to conventional wastewater treatment systems (Scholz 2006).
- The removal of pollutants may vary during the year and in the long term due to seasonal weather patterns (*e.g.*, as reported by Dunne *et al.* (2005b)) and also variations in the inflow of pollutants. In some circumstances, pollutants might even be released (*e.g.*, as reported by Thorén *et al.* (2004)), but this can be minimized by the use of a modular approach (*i.e.*, using a series of FCW cells) and by designing the system for extreme rainfall events.
- Emission of greenhouse gases (*e.g.*, methane and nitrous oxide as reported by Hefting *et al.* (2003)).
- Most FCW should be regarded primarily as water treatment systems and treated accordingly; *i.e.*, not used for bathing, fishing, or animal watering due to the possible presence of pathogens, toxins, and parasites.
- Although safety and health concerns for humans and livestock may arise from FCW because they contain standing water, such concerns can be minimized by the use of gently sloping sides, marginal vegetation, and by raising public awareness.
- Some FCW pollutants (*e.g.*, heavy metals and pesticides) may cause harm to wildlife, but this impact can be mitigated by an appropriate design taking into account ecological aspects and by keeping certain types of effluents (*e.g.*, pesticide washings) away from the FCW.
- Some infiltration of water to groundwater may occur. However, infiltration is reduced by the use of shallow water depths and an adequate substrate such as clay (Hill *et al.* 2000) and decreases over time through sealing of the bed of the wetland by accumulated organic matter and sediment (Harrington *et al.* 2005).

2.2.3 Farm Constructed Wetland Site Suitability

2.2.3.1 Effluent to Be Treated

In order to decide if a FCW is needed and appropriate for a given farm, a site-specific approach is required, usually involving the following steps (Zedler 2003; Carty *et al.* 2008):

1. assessment of the type and volume of the effluent to be treated (present and future) and the infrastructure present on the farm;
2. determination of the need for a FCW from an assessment of existing on-farm measures and potential cost-effective alternatives for dirty water management; and

3. determination of the characteristics of the site available for FCW construction and assessment of any potential impacts that may result from FCW implementation.

It is recommended that specialist advice from a suitably qualified consultant or engineer be sought when assessing the need for a FCW and the suitability of an FCW site. To decide whether a FCW is an appropriate option for a farm, the farmer or farm advisor will need to assess the present effluent management, types of effluent, present storage facilities, the economics of developing a FCW, and the likelihood that a FCW will improve existing conditions. When assessing the type and volume of effluent to be treated, the present and possible future loading (*e.g.*, increased stock numbers, impervious areas, and shed roofs) should be considered.

Precipitation events are the primary factor affecting design, as there is almost no attenuation of farmyard-intercepted rain. The hydraulic flux in any year arising from storm events can be between 25 and 100 times the mean flow from the farmyard. It is this flux that must be appropriately managed within the FCW design.

Most FCW are designed to deal with storm events and the variable composition of influents through including sufficient wetland area for flow attenuation and water treatment. The areas of yards, tracks, and roofs within the farmyard must be calculated to determine the volumes to be treated. Each farmyard will have varying daily water usages (*e.g.*, yard washings and parlor washings), which must be calculated as part of the total volumes of water and runoff produced on site (Carty *et al.* 2008).

Before implementing a FCW, it is necessary to ensure that BMP are employed to decrease the contamination of the farmyard runoff (*e.g.*, improved cattle diet, roofing of feeding areas, and separation of roof runoff). It is also important to look at potential alternatives available to the farmer to deal with the contaminated farmyard runoff (*e.g.*, storage and subsequent land spreading (Bowmer and Laut 1992), and overland flow) as discussed elsewhere (Harrington *et al.* 2005; Scholz *et al.* 2007a, b).

Carty *et al.* (2008) provide a decision support tree for the treatment of farmyard and roof runoff with FCW. This tree can be used as a template and might require modification to address differences in regional and national legislation.

2.2.3.2 Site Characteristics

Each FCW design is site specific requiring “expert judgment” as discussed by Scholz (2006a, 2007). Therefore, a comprehensive site-specific assessment combining site investigation and desk study is necessary to determine the site characteristics, assess any potential impacts, decide on any groundwater and surface water protection measures, and provide data that will be used in the design of the FCW.

The site assessment should be undertaken by a person who is professionally qualified, has experience in all of the required disciplines, and can call in experts if necessary to clarify any anomalies that may arise. When assessing the site suit-

ability, the relevant bodies and authorities should be consulted in the context of the prevailing legislation. The following existing characteristics of a site should be assessed: topography, geology, soils and subsoils, hydrogeology, hydrology, flora and fauna, archaeological and architectural features, and natural interest.

Climatic conditions, including rainfall and evapotranspiration, will determine the volumes to be treated by the FCW system and the residence time within the FCW. Due to the sources of effluent (mainly precipitation-generated runoff) and the effects of climate, discharge from the FCW may be confined to wet periods.

Ideally, an FCW should be developed on gently sloping land (Harrington *et al.* 2005). Areas that are steeply sloping require larger wetland areas, deeper soils and subsoils, more excavation, and probably increased costs of construction. When assessing the proposed site, consideration should be given to the approximate area of land required for the wetland, embankments, and access (see below).

The location of a wetland down gradient from the farmyard will allow for the effluent to flow by gravity and thus remove the need for pumping. Existing Irish FCW have been located at varying distances from 5 m to greater than 500 m from the farmyard. The FCW cells and embankments should be designed so that water flows through the FCW by gravity (Scholz *et al.* 2007).

A topographical survey of the farmyard and the proposed FCW site, at a scale of at least 1:500, is recommended. The survey should include contours (0.5-m contours), location and use of buildings, boundaries, hydrological features, and any features of archaeological and architectural interest.

A desk study should be undertaken in advance of the site visit to establish the geological context of the site. A number of trial holes should be dug on the proposed site to examine the type, depth, and texture of the underlying soil and subsoil. Trial holes should be dug to a depth of between 2 and 3 m, with a minimum of four trial holes per approx. 4000 m². Where the soil texture is not easily determined by hand assessment methods on site, a laboratory test can be conducted to determine soil permeability, particle size distribution, or both.

Where low-permeability soils suitable for the construction of a wetland (such as clays) are not found on site, alternative measures should be taken (Carty *et al.* 2008):

- Import suitable soils: there may be a requirement to import soils for part or all of the FCW. Additional soils can sometimes be found on the farm or locally, but this may incur additional costs for transportation, particularly if located away from the farm. Importing soil from another site or farm may require a license subject to national guidelines and laws.
- Use of an artificial liner: the use of an artificial plastic liner, similar to that required for a landfill, can also be considered. However, its use will incur a much greater cost and may require replacement in the future. Furthermore, the installation of an artificial liner requires the expertise of an engineer and consultation with the local authorities and regulators.

Field drains are widely used in agricultural land. All field drains must be located (through local knowledge or through site excavation) to assess whether they are likely to conduct water to the FCW or provide a pathway for water to leave the

FCW before completing treatment. A trench or drain dug during construction around the perimeter of the proposed FCW site to a depth of at least 1 m should intercept any field drains and divert their flow to surface water.

The types of flora, fauna, and habitats on and adjacent to the site should be described to determine whether the development of an FCW will have any negative impacts (e.g., removal of hedgerows, nesting areas, and trees). The positive contribution of the FCW in creating new habitats or enhancing existing habitats and biodiversity should also be considered (Froneman *et al.* 2001).

The proximity to dwelling houses must be determined to assess issues such as security, health, safety, and odors. In some instances, access to the FCW by humans or livestock may need to be restricted by appropriate fencing or hedging, although this may only be required around the initial FCW pond.

The location and type of any archaeological or architectural feature on or adjacent to the site must be assessed, subject to national guidelines, using a combination of site investigation and desk study (Scholz 2007; Carty *et al.* 2008).

Carty *et al.* (2008) show a decision support tree for finding a suitable site for an FCW. This tree may require alteration depending on regional and national differences in managing agricultural land.

2.2.3.3 Discharge Options

Any nearby surface water features such as rivers, streams, lakes, and drains should be noted during the survey. In most cases, a local river, stream, ditch, or even woodland will provide the final discharge point for the water that has been treated by the FCW. As well as assessing the suitability of a watercourse as a potential discharge point, consideration must be given to high water levels, other discharges upstream and downstream, water abstractions downstream, and flooding. Where an FCW is to be developed in a floodplain, the potential impacts upstream and downstream must be assessed. In some instances, the floodplain may be protected, in which case an FCW may not be permitted. Regional and national authorities need to be consulted (e.g., UNEP 2003; Zedler 2003; Scholz 2006a).

The final discharge should be to surface water with sufficient assimilative capacity, such as a stream or river with significant flows throughout the year, rather than to a field drain that has low assimilative capacity and often dries out. The discharge of the treated waters should have a negligible effect on the receiving water (e.g., Zedler 2003).

Where a discharge to surface water is not possible or not suitable, a wet woodland or willow bed could be used. The woodland can be designed to have a zero discharge or minimal discharge; however, it must be noted that this may require a large area (Scholz 2006; Zheng *et al.* 2006).

It is recommended that an FCW should not be immediately adjacent to surface water to minimize the impact of any failure of the FCW. The buffer distance depends on the adjacent surface water and the ground conditions; a minimum of 5 to 10 m is recommended.

During the site assessment, information on the following should be collected: wells, springs, water table elevation, aquifers, nearby surface and groundwater supplies, and connectivity with surface water features, since they will cause increased volumetric loading to the system, reducing residence time and treatment. Furthermore, sufficient distance must be provided between an FCW and up-gradient and down-gradient water supplies (Scholz 2006a; Zedler 2003).

The height of the water table must be recorded. The base of the FCW cells should be above the water table, so sites with high water tables will not be suitable. The vulnerability of groundwater to pollution is assessed through combining information from groundwater vulnerability maps (where available) with a site-specific study and investigation of soil and geological conditions.

2.2.4 Design Guidelines for Farm Constructed Wetlands

2.2.4.1 Background and Water Treatment Requirements

Once site assessment and selection has been completed, the detailed design of an FCW can be conducted, taking into account the farm and farmyard structure and management practices. When designing a FCW, the following aspects are therefore of major importance (Zedler 2003; Harrington *et al.* 2005):

- objectives behind the construction of the FCW;
- characteristics of the farmyard runoff to be treated (volume and quality);
- water quality targets to be achieved; and
- land availability to achieve the target water quality.

Adequate pretreatment, retention time, management, and operation (*e.g.*, removal of sediment and regular inspection) and design for management (*e.g.*, access) are required to achieve effective water treatment through a FCW. A FCW that is fit for a certain purpose should have the following attributes (Harrington and Ryder 2002; Rice *et al.* 2002; Scholz *et al.* 2007a, b):

- be reliable and efficient in water treatment, particularly during storm events, extreme rainfall with increased hydraulic loadings, and also under relatively cold conditions;
- be capable of coping with accidental spillages;
- be flexible and versatile;
- be relatively simple to build;
- have low operation and maintenance requirements and costs;
- have low energy consumption;
- be a good landscape fit;
- enhance habitat and biodiversity (Froneman *et al.* 2001); and
- be safe for farmers and for the public.

2.2.4.2 Runoff Capture and Conveyance

One of the early steps in FCW construction should be to ensure that any contaminated runoff, such as from roofs, farmyards, and tracks, be captured properly. Runoff from adjacent land should usually not enter the FCW. However, regional recommendations may vary.

The conveyance of waters to, within, and from the FCW must consider the following (Scholz 2006a): collecting water from the farm, conveying that water to the wetland, moving water within the wetland, and moving water out of the wetland. Where possible, the effluent should flow by gravity to minimize maintenance and energy costs.

It is essential that any containment be secure and that only water with acceptable concentrations of contaminants be discharged to watercourses or groundwater. The FCW embankments retaining the water flowing through the system must be sufficiently high to allow for the accumulation of sediment and detritus. The soil lining the base must adequately impede infiltration to protect groundwater (Dunne *et al.* 2005a, b; Keohane *et al.* 2005; Scholz *et al.* 2007b).

2.2.4.3 Hydraulics, Water Balance, and Residence Time

The periodic nature of precipitation and the interception and uptake of water by emergent vegetation, evaporation, and ground infiltration has the capacity to arrest water flow between the individual segments of a FCW. This creates a freeboard between the outlet level and the level of the water contained within an individual wetland cell. It also provides each wetland cell with ‘additional’ receiving hydraulic capacity before flow to the next segment can resume, thus enhancing the hydraulic residence time (Dunne *et al.* 2005a, b; Scholz *et al.* 2007).

The treatment effectiveness of surface-flow wetland systems in comparison to sub-surface-flow systems (Mantovi *et al.* 2003) is typically based on having appropriate hydraulic residence times, which depend very much on the specific site conditions (Harrington *et al.* 2005). In shallow, emergent, or vegetated wetlands, such as FCW, this depends on having sufficient functional wetland area with an appropriate length-to-width ratio and a high emergent vegetation density. The hydraulic effectiveness of the FCW can be maximized by the following measures (Scholz *et al.* 2007):

- segmentation of the wetland into a number of wetland cells of appropriate configuration (see below);
- avoidance of preferential flow;
- dense vegetation stand; and
- managing the water depth to ensure optimal functioning (Scholz 2007).

The velocity of the water flow through the FCW is determined by the volumetric flow and the cross-sectional area of the water channel. Minimizing the velocity enhances the settling of suspended solids and promotes a longer contact time with

emergent vegetation whose surfaces support biofilms (Scholz *et al.* 2002; Kantawanichkul and Somprasert 2005; Scholz *et al.* 2007).

Wind and temperature gradients can generate water movement between the different aquatic strata within a wetland cell. Emergent vegetation minimizes mixing, thereby allowing the cleaner water to flow preferentially along the surface, especially during periods of large precipitation-generated flow. In the initial receiving wetland cell, floating vegetation may develop (typically *Glyceria fluitans* and *Agrostis stolonifera*) and water flow will be partially sub-surface, thus having the additional advantage of reducing odors (Dunne *et al.* 2005a; Scholz and Lee 2005; Scholz *et al.* 2007).

2.2.4.4 Wetland Sizing, Inlet, and Outlet

The design and sizing of FCW has often focused on phosphorus, which is recognized as one of the most difficult contaminants to remove from water and is a limiting nutrient in many freshwater ecosystems (Braskerud 2002). For example, a catchment-specific study of 13 wetland systems in Waterford (Ireland) showed that, to achieve a mean MRP concentration at the outlet of 1 mg/l or less, the wetland area required was at least 1.3 times the farmyard area, and that each system should contain approximately four cells.

The design is based on two assumptions: the larger a wetland, the more phosphorus removal can be expected; and all ICW studied near Waterford (Ireland) were at the designated threshold of failure for phosphorus (1 mg/l MRP for the outflow or near it in cases of no flow (Scholz *et al.* 2007). This finding relating to MRP is, however, not universally applicable.

The aspect ratio is defined as the mean length of the wetland system divided by the mean width. The study conducted in Ireland showed that to obtain an outlet MRP concentration of 1 mg/l or less, the FCW aspect ratio should be less than 2.2. In fact, the closer the aspect ratio is to 1 (*i.e.*, the more the FCW shape is square or round), the better the wetland treatment (Scholz *et al.* 2007).

Sizing must also take into account the footprint of the upper embankments, which should be between 2 and 3 m wide to ensure stability and to provide easy access for maintenance and monitoring. For safety reasons, inner embankments should be gently sloping.

Inlets and outlets should be kept as simple as possible and avoid the use of concrete and overengineered structures. Pipe diameters should be at least 150 mm to avoid clogging. Stone chippings should be placed beneath the inlet and outlet pipes to prevent scouring. Elbow pipes fitted to linear ones can be used to control the water level and the outflow of each cell. Carty *et al.* (2008) provide further details on sustainable wetland design.

2.2.4.5 Landscape Fit, Biodiversity, and Life Span

The potential visual aspect of the FCW system design is important for achieving empathy from both farm dwellers and the local community. Usually, FCW

with curvilinear shaping and virtually level embankments have a more ‘natural’ appearance.

Several measures can improve the landscape fit and biodiversity of an FCW. Through BMP on the farm, the level of contaminated water discharging to the ponds can be reduced (Scholz 2006a; Zheng *et al.* 2006). Wherever possible, the FCW should be located near (but not connected to) existing wetlands, ponds, and lakes to allow for natural colonization by plants and animals. The FCW cells should be irregular in shape, with gently sloping embankments and areas of deeper water, and contain islands where sufficient area is available. The use of locally occurring wetland plant species for establishing habitats and enhancing biodiversity appropriate to the locality is also likely to further increase the robustness and sustainability of the system (Froneman *et al.* 2001; Scholz 2007; Scholz *et al.* 2007b).

The area surrounding the FCW can be planted with trees and shrubs, but trees are not recommended on the FCW embankments. If possible, small pools around the main system should be created to collect runoff from adjacent fields and create additional aquatic habitat (Froneman *et al.* 2001; Carty *et al.* 2008).

Wetland embankment height, inflowing solids, and accumulating detritus determine the functional life span of each segment of the FCW. With detritus accumulation and a minimum embankment height of 1 m, a life span of between 50 and 100 years is expected. However, the life span can be virtually indefinite if detritus removal takes place regularly, as discussed by Scholz *et al.* (2007b) and in the section on maintenance below.

2.2.5 Construction and Planting

2.2.5.1 Construction

Ideally, the construction of an FCW should be undertaken during the dry season and the involvement of the farmer is encouraged. Construction of an FCW should be undertaken by a competent machine operator and signed off by a qualified engineer on completion, although regional and national requirements for construction vary. The main stages of construction are as follows (McCuskey *et al.* 1994; Scholz *et al.* 2007a, b):

1. Stripping of topsoil from FCW area and retention of it for later use (if applicable);
2. excavation of subsoil;
3. layering and compaction of soils for cell liner (laying of an artificial liner if required);
4. creation of gently sloping embankments (at least 1 in 3), with height >1.0 m and tops 2 to 3 m wide to guarantee stability and sufficient access around the wetland;

5. potential redistribution of topsoil (if nutrient poor) over the base of the ponds;
6. pipe laying between cells;
7. placement of stones or chippings beneath inlet and outlet pipes; and
8. planting of FCW cells.

The machinery required for construction comprises a tracked excavator and bulldozers and a vibrating roller where a higher degree of soil compaction is required. The topsoil is stripped from the FCW area and retained for later use. The depth of excavation will vary depending on the depth of the overlying topsoil (McCuskey *et al.* 1994; Carty *et al.* 2008).

Subsoil is excavated to form the base of each pond and to provide material for the embankments. The depth of excavation will depend on the topography and elevations required between each pond; typically, an excavation of 0.5 m below the ground level is required for flat sites. Where surrounding land slopes towards the FCW system, a drain or an embankment should be constructed around the system to divert potentially large volumes of runoff away from the system (McCuskey *et al.* 1994; Carty *et al.* 2008).

The amount of compaction and layering will depend on the type of material being used to construct the FCW and should be determined during the site assessment of the soils. Where the soil permeability is sufficiently low, no layering is required. Medium- to low-permeability soils will require layering and compaction to ensure a permeability of approx. 10^{-8} m/s. Regional and national guidelines on soil permeability thresholds should be consulted by the designers.

Pipe ducting and elbows are placed at the inlet and outlet points for each FCW cell (see above) to ensure that the movement of water in each cell is across the maximum distance from the point of inflow to the exit. Furthermore, piping should be positioned in accessible locations and placed as low as possible to the base of the exit point of the upper cell to ensure the possibility of drainage. In FCW discharging to surface waters prone to flooding, a non-return valve should be placed on the outlet pipe to prevent any flood water from entering the wetland system. Stones or chippings should be placed beneath the inlet and outlet pipes to prevent scouring of the wetland floor and to provide access for monitoring (McCuskey *et al.* 1994; Carty *et al.* 2008).

The banks and floor of the wetland cells should normally be compacted and smoothed off using tracked excavators that are suitable for use on difficult wet terrain. Where a greater degree of compaction is required, rollers are used.

Measures should be employed during the construction of an FCW to limit the impact on surface water (runoff and siltation) and groundwater through proper management and supervision following national guidelines.

Depending on the location of the site, there may be a need for fencing to restrict access to humans and livestock. When determining the type of fencing, the farmer will need to consider the extent of fencing, costs, and human and machinery access. In most cases, fencing similar to that already used on the farm will be suitable (Carty *et al.* 2008).

2.2.5.2 Planting

The primary vegetation used in FCW is composed of emergent aquatic plant species (helophytes). These species have evolved to enable them to root in soils with little or no available oxygen, growing vertically through the water column with most of their leaves in the air. They have specially adapted tissues and physiologies that facilitate oxygen storage and its transportation from the leaves through the stem to the roots. The soil and water characteristics influence the type of helophyte species and their treatment performance (Scholz 2006a; Scholz *et al.* 2007). Helophytes in FCW have important functions such as the following (Scholz and Lee 2005; Jiang *et al.* 2007):

- provision of a support structure for microbial colonies to develop;
- facilitation of aerobic microbial activity (principal treatment removal process);
- uptake of nutrients;
- source of organic matter;
- reduction of water flow to increase residence time and settlement; and
- reduction of final volumetric discharge through plant transpiration and interception.

While more than 100 native helophyte species can be used, about 20 species are actively planted in constructed wetland systems (Scholz and Lee 2005; Scholz 2006a). Initial plant establishment is the dominant influence on the vegetation structure on an FCW during its early years. Plant establishment is dependent upon the nature of the influent and water depth, plant species and physiological maturity, and planting density (Scholz *et al.* 2007b). Interspecies competition and other biotic factors, especially waterfowl, influence long-term vegetation development (Froneman *et al.* 2001; Scholz 2006a).

Plants should be sourced from existing (constructed or natural) wetlands where permitted or from accredited plant nurseries. Care must be taken to ensure that non-native species are not introduced. Seedlings are usually not recommended as they are more vulnerable to pollution and water level variations and take longer to establish (Carty *et al.* 2008).

If plants are sourced from existing wetlands, sufficient root material should be obtained to allow the plants to establish, while also ensuring that the root system of the original plant can regenerate. Harvesting should be carried out ideally by hand and over a sufficiently large area to minimize disturbance to the plants and its environment.

Planting of mature macrophytes should be carried out by hand into water or suitably saturated soils, ideally with 50 to 100 mm of water above the topsoil. Water levels should be maintained between 100 and 200 mm for at least the first 6 weeks after planting, and the wetland should not be allowed to dry out below the soil surface. Shallow waters (<100 mm) will encourage establishment of grasses and weeds, which can restrict the growth of the wetland vegetation. Water may need to be sourced from nearby surface water features during planting, as the

effluent from the farmyard may only provide sufficient cover for the first cell (McCuskey *et al.* 1994; Scholz and Lee 2005).

The area of each wetland should be planted with various wetland species to increase biodiversity. The first wetland cell will receive a higher pollution load, often with a high ammonia–nitrogen concentration, so a minimum of three plants per square meter is recommended to increase plant success. Subsequent cells will have reduced effluent loadings, and a minimum of two plants per square meter is therefore recommended. The farmer may wish to allow areas within the final cells to colonize naturally. Provided that planting is carried out at the beginning of the growing season and water levels are maintained between 100 and 200 mm for the initial few months, a desirable vegetation cover of at least 80% after 2 years should be attained. To establish a dense cover in a shorter period, the initial planting densities should be doubled (Carty *et al.* 2008).

Prior to the commissioning of the FCW, the system should be signed off by a competent engineer (depending on regional guidelines) and have all monitoring and maintenance features and fencing in place. During the first few months, water levels within the wetland ponds should be at a minimum, approx. 100 to 200 mm, to provide favorable conditions for plant establishment. It is possible that some effluent may need to be diverted away from the FCW during the start-up stage; *e.g.*, stored or land spread (McCuskey *et al.* 1994; Carty *et al.* 2008).

2.2.6 Maintenance and Operation

2.2.6.1 Pipe Maintenance and Flow Control

The success of a FCW will depend on the maintenance and operation of the system. While a FCW is designed to be as self-maintaining as possible, it is crucial that a maintenance program be adopted to ensure continued effective water treatment and ‘rejuvenation of the system’.

All inlet and outlet pipes within the FCW system should be visually inspected weekly for blockages, sediment accumulation, and debris. Blockages will affect the hydraulics of the FCW system, while sediment accumulation may indicate inadequate solids separation further up in the system. Any blockages and sediment or debris accumulations around the inlet or outlet pipes should be cleared (Harrington *et al.* 2005; Scholz 2006a).

During prolonged dry periods, water depths within the ponds will decrease, especially in down-gradient regions of the wetland. It is essential to ensure that soils are flooded (to at least 50 mm). Usually, FCW should not be allowed to dry out as cracks may form in the base, which may cause higher infiltration rates in the short term when effluent reenters the cell. Once the emergent vegetation is established, FCW in temperate regions should be able to cope with reduced water depths, and even drying out, for periods of several months.

Any adjustment of pipes must be carried out gradually as these movements may cause huge surges of effluent to subsequent wetland cells or receiving surface or groundwaters, reducing residence time and treatment within the system.

2.2.6.2 Vegetation and Sediment Maintenance

Water levels should be maintained at less than 300mm to ensure good plant growth. However, most macrophytes tolerate short periods of increased water depth (up to 500mm), such as that associated with high rainfall, as well as low water depths or even no water during dry weather. Major changes in vegetation cover and composition should be noted as a possible indicator of changes (*i.e.*, degradation or improvement) in wetland performance. Pest control might be required if unwanted plant species take over a wetland.

Sediment accumulating within a wetland comprises organic material from the influent and dead plant matter, and also mineral sediment from eroded tracks. Existing constructed wetlands have shown varying accumulation rates depending on the type of loading and the amount of vegetation cover within the wetland. The removal of accumulated sediment is usually confined to the first cell (Scholz *et al.* 2007).

For a heavily loaded system, the inclusion of an open water pond at the initial stage of the FCW as a sediment trap may extend the operational life of subsequent cells before the removal of material is required. Any pond located within the initial wetland cell would require relatively frequent material removal (most likely biannually) but could be configured to be as accessible as a standard slurry storage tank (Scholz 2006a; Scholz *et al.* 2007b).

The most appropriate way of managing the material removed is likely to be land spreading on the farm in accordance with good farm management practice. Information on the solid content and nutrient composition, particularly phosphorus, is required to ensure that the usage complies with farm nutrient management requirements (Bowmer and Laut 1992; Carty *et al.* 2008).

As the farm's nutrient management requirements may not allow the full amount of removed sediment to be spread on the land for one year, it is likely that occasional storage of removed sediment would be required for some sites. The storage requirements specified in good agricultural practice regulations with respect to farmyard manure should also suffice for FCW sediment to ensure environmental protection.

2.2.6.3 Safety, Security, and Maintenance

It is recommended that the farmer undertake regular visual inspection of the internal and external faces of the wetland embankments to check for any water leakage, slippage, or distortion. The internal embankment face can be checked by walking along the embankment crests and external embankment faces by walking

along the external boundary of each cell. Any defects such as leakages, slippages, or distorted areas should be addressed immediately.

Access around the wetland should be maintained by managing vegetation growth on the embankments. Under normal operating conditions, growth on some pond crests will need to be cut biannually using a mower or topper. Security and safety considerations for both humans and livestock should be incorporated into the design of the FCW subject to national guidelines. Access to the influent-receiving segment of the FCW may have to be limited through fencing or by existing hedgerows around the FCW.

2.2.6.4 Monitoring the Final Effluent and Receiving Watercourses

The monitoring of the FCW and the final effluent will allow the farmer to assess the performance of the wetland system and to detect any malfunction. The general appearance of the final effluent should be noted, paying particular attention to water color, the presence of “sewage fungi” (Scholz 2006a), smell, and any evidence of plant material in the discharge. If the final discharge water appears to be heavily discolored or polluted or contains plant material, then the outlet pipe should be isolated immediately by closing the gate valve. However, water that is visibly clear may also have a high nutrient load, which can only be determined by laboratory analysis.

The condition and appearance of the receiving waters at the point of discharge should be checked on a monthly basis and following extreme events such as high rainfall. The farmer should assess the condition and appearance of water, both upstream and downstream of the discharge location. Heavily discolored water or the appearance of sludge-type material may indicate an upstream pollution source. Foaming immediately downstream of the discharge point may indicate pollution in the final effluent. The outlet pipe from the FCW should be isolated immediately by closing the gate valve in the event of any suspected pollution incident. The farmer should obtain advice from a suitable agricultural advisor and the regulator (Scholz 2006a; Carty *et al.* 2008).

2.2.7 Conclusions

This sub-section proposes, for temperate climates, universal design, operation, and maintenance guidelines for a FCW, which are a specific application of ICW. Guidelines have been proposed for assessing the need for FCW and their site suitability. The overall design of FCW is empirical, site specific, and predominantly based on expert judgment. The guideline is based on minimizing costs for the benefit of the farm owner but not at the expense of the environment. Therefore, FCW may require considerably more land than conventional treatment technologies, which are usually associated with greater maintenance and capital costs.

2.3 Integrated Constructed Wetland for Treating Farmyard Runoff

2.3.1 Introduction

Farmyard runoff, which is rich in nutrients, is a source of diffuse pollution and potentially a serious risk to receiving watercourses by contributing to eutrophication (Cleneghan 2003). Constructed wetlands have been used worldwide to treat different categories of wastewater including domestic, industrial, acid mine drainage, agricultural runoff, and landfill leachate (Kadlec and Knight 1996; Scholz 2006a).

The ICW concept (Harrington *et al.* 2007; Scholz *et al.* 2007a, b) is based on an approach that endeavors to achieve water treatment, landscape fit, and biodiversity enhancement targets by an innovative wetland design methodology. One subgroup of ICW is the FCW as defined by Carty *et al.* (2008).

The conventional practice in Ireland is land spreading of farmyard dirty water (Tunney *et al.* 1997). The storage and spreading is governed by rules to prevent water pollution. However, this practice requires considerable labor and machinery resources as well as storage infrastructure. Improper storage and spreading has been linked to water pollution, particularly to increased levels of nitrogen and phosphorus in surface and ground waters (Healy *et al.* 2007). In contrast, the ICW concept is founded on the holistic use of land to protect and improve water quality. These systems are areas of land–water interface that form an integral part of the environmental and ecological structure of the landscape. They act as buffer lands that control the transfer and storage of farmyard dirty water rich in nutrients (Scholz *et al.* 2007b).

The main characteristics of an ICW, such as shallow water depth, emergent vegetation, and the use of *in situ* soils, mimic those found in natural wetland ecosystems (Harrington *et al.* 2005). Scholz *et al.* (2007a, b) reported on the detailed concept of these synergistic, robust, and sustainable systems by referring to case studies in Ireland.

Contaminated effluent is treated in an ICW through various physical, chemical, and biological processes involving plants, microorganisms, water, soil, and sunlight (Kadlec and Knight 1996; Scholz 2006a). The extent of treatment in constructed wetlands depends upon the wetland design, microbial community, and types of aquatic plants involved. Water quality improvements are predominantly caused by bacteria (Ibekwe *et al.* 2003); for instance, most of the biological degradation takes place within bacterial films present on sediments, soils, live submerged plants, and the associated litter. Microbes catalyze chemical changes in wetland soils and indirectly control the nutrient availability to plants and, in turn, the water quality (Mitsch and Gosselink 2007). The litter and associated sediment originating from decaying macrophytes provide considerable surface area for the attachment of biofilms and are therefore important for microbial processes such as the transformation of nutrients in wetlands (Scholz 2006a).

Some of the main processes for nitrogen transformation in constructed wetlands are nitrification, denitrification, and anammox (Shipin *et al.* 2005; Vymazal 2007), which are all microbe-mediated processes (Wallace and Knight 2006). It is expected that nitrogen removal will fluctuate and increase over time as vegetation becomes established and sufficient carbon is available for denitrification (Kadlec and Knight 1996).

Wetlands have a relatively high capacity to store nutrients (Braskerud 2002). Phosphorus assimilation in constructed wetlands depends on factors such as the influent phosphorus concentration, the rate of internal biomass cycling, and the wetland age (Kadlec 1999; Wallace and Knight 2006). However, the long-term storage of phosphorus in these systems is linked to the cycling of phosphorus through the growth, death, and decay of plant biomass. Previous studies on wetland ecosystem structure and function have shown that soil is the most important long-term ecosystem phosphorus storage compartment (Richardson and Marshall 1986). Processes such as soil adsorption and peat accumulation control the long-term sequestration (*i.e.*, capture) of phosphorus (Richardson and Marshall 1986; Reddy *et al.* 1999).

Most previous studies have been short term, and on either pilot plant- or laboratory-scale experimental systems. Very few studies have been conducted on full-scale constructed wetlands with long-term evaluations (Brix *et al.* 2007; Newman *et al.* 1999). There is a lack of information on the performance of mature constructed wetland and ICW systems including FCW structures. The purpose of this study was therefore:

- to evaluate the treatment efficiency of a full-scale mature ICW system that has been in operation for 7 years;
- to assess the annual and seasonal variations in nutrient removal;
- to identify the presence of microorganisms responsible for nitrogen transformations within these systems; and
- to investigate the impacts of potential contamination of nearby surface waters and groundwater.

2.3.2 Material and Methods

2.3.2.1 Site Description

The researched ICW treatment system is a FCW, situated in County Waterford (southeast of Ireland) at a longitude of 07°02'40" W and a latitude of 52°11'28" N (Figure 2.14). The case study area is located in a temperate zone with a mean annual temperature and precipitation of 11.4°C and 1094 mm, respectively (Met Éireann 2007). Mean seasonal temperatures for the region in 2007 were as follows: winter, 7.8°C; spring, 10.3°C; summer, 14.9°C; and fall, 12.2°C.

The ICW system was constructed in 2000 and commissioned in February 2001 to contribute to the improvement of the water quality of the Annewtown Stream.



Figure 2.14 Location of the integrated constructed wetland site within the Annestown catchment near Waterford, Ireland

The ICW system has a total area of 0.76 ha. The primary vegetation types used in the ICW are emergent plant species (helophytes). The first three cells were densely vegetated, while the last cell had only sparse vegetation.

The cells were not lined with an artificial liner. However, the subsoil was re-worked and used as a natural liner. The cells were only partly planted with vegetation naturally available on the site. Further plant growth occurred by natural recruitment. Each cell had one inflow and one outflow structure. The water flow between each cell was by gravity through a PVC pipe.

The effluent entering the ICW system comes from a dairy farm of 0.5 ha with 77 cows. The wastewater contained farmyard and roof runoff occasionally contaminated by manure and was conveyed to the ICW system by gravity through a pipe.

The key features of this constructed wetland were horizontal surface flow and intermittent hydraulic loading. The ICW system was based on four cells operated in series. A single influent entry point was located at the first cell. The first three cells had depths of approx. 1 m, while the fourth cell was approx. 1.5 m deep.

2.3.2.2 ICW Design

Information on topography, hydrogeology, surface water, groundwater, and subsoil is important for assessing the ICW site suitability prior to the design and construction phase. Details concerning site assessment criteria have been documented by Scholz *et al.* (2007b).

The ICW system was designed not only to improve water quality but also to integrate the structure naturally into the landscape and enhance biodiversity. The design philosophy is in agreement with the guidelines proposed by Carty *et al.* (2008).

A detailed site-specific assessment combining site investigation and desk study was conducted to determine the site characteristics and assess any potential impacts on groundwater and surface waters. The site was assessed by characteristics such as topography, geology, soils and subsoils, hydrogeology, hydrology, flora and fauna, archaeological and architectural features, and natural interest.

A topographical survey of the farmyard and the proposed ICW site was conducted that included contours (0.5 m contours), location and use of buildings, boundaries, and hydrological features. The nearby surface water features such as rivers, streams, and drains were also noted during the survey. The suitability of the nearby stream as a potential discharge point was also assessed.

During the site assessment, information on wells, springs, water table elevation, aquifers, nearby surface and groundwater supplies, and connectivity with surface water features was also gathered. The soil and geological conditions were also studied.

2.3.2.3 Sampling and Analytical Methods

Grab samples for each wetland cell inlet and outlet were taken on an approximately fortnightly basis. Liquid samples were analyzed for variables including the 5 d at 20°C N-allylthiourea biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SSs), ammonia–nitrogen, nitrate–nitrogen, molybdate reactive phosphorus (MRP; equivalent to SRP), total coliforms (TCs), and *E. coli* at the Waterford County Council water laboratory using predominantly American Public Health Association standard methods (APHA, 1998) unless stated otherwise.

Continuous flow measurements were undertaken between April 2003 and June 2004. From June 2004, spot measurements were made of flows into and out of each ICW cell. Furthermore, flows entering and leaving ICW cells were monitored and recorded using standard Greyline flow meters and associated data loggers. Blockages due to vegetation required frequent maintenance of the flow meters.

Four piezometric groundwater-monitoring wells were placed up-gradient (one well), within (two wells), and down-gradient (one well) at different depths at the ICW site in fall 2004. These wells were sampled on a quarterly basis throughout 2004 and 2007. The day before sample extraction, all wells were purged (*i.e.*, process of removing water that is unrepresentative of the surrounding strata) prior to sampling, and subsequent analysis was carried out according to APHA (1998).

The water quality of the receiving stream, which is a tributary of the Annes-town Stream, was monitored (Figure 2.14). Grab samples were collected from 21 locations along the stream located adjacent to the ICW. Three points were sampled upstream, up to 16 parallel to cell 4, and 2 points downstream to assess the effect of the ICW system on the associated receiving watercourse. In 2007, grab samples were taken predominantly during an intense period for monitoring the water quality of the receiving watercourse. The sampling scheme was designed to monitor the water quality during periods of relatively low and normal flows.

Weekly samples were collected in late spring and during the following summer. Later on, monthly samples were collected during fall and winter. Nutrient analysis was conducted at the Waterford County Council water laboratory using American standard methods (APHA 1998).

In May 2007, sediment samples were collected with a sediment sampler (diameter of 4 cm) from the wetland cells to identify bacterial groups responsible for nitrogen transformations in the ICW. Predominantly triplicate field sediment samples were collected 1 m to the left, 1 m in front, and 1 m to the right of each sampling point located near the inlet of each ICW cell and outlet of the last ICW cell to reduce bias and to gain more representative samples. Samples were stored at -20°C before analysis.

All samples were frozen immediately after collection and sent off to Linköping University (Sweden) for subsequent molecular microbiological analysis involving standard techniques (*e.g.*, extraction of deoxyribonucleic acid (DNA), polymerase chain reaction (PCR), and gel electrophoresis) as outlined by Sundberg *et al.* (2007).

Sediment samples were subjected to DNA extraction using a FastDNA SPIN kit for Soil (Bio 101, La Jolla, CA). Samples (0.25 g) were suspended in a sodium phosphate buffer supplied with the kit as stipulated by the manufacturer and homogenized for 180 s with a handheld blender (DIAX 900 Homogeniser Tool G6, Heidolph, Kelheim, Germany).

Bead beating was undertaken to disrupt the soil aggregates and to lyse bacterial cells mechanically. Bead beating was extended to 3×30 s to achieve good homogenization of the samples. The subsequent centrifugation was extended to 2×5 min and the centrifugation after washing with SEWS-M, a salt and ethanol wash solution (Qbiogene, USA), was extended to 5 min. The extracted DNA was stored at -20°C .

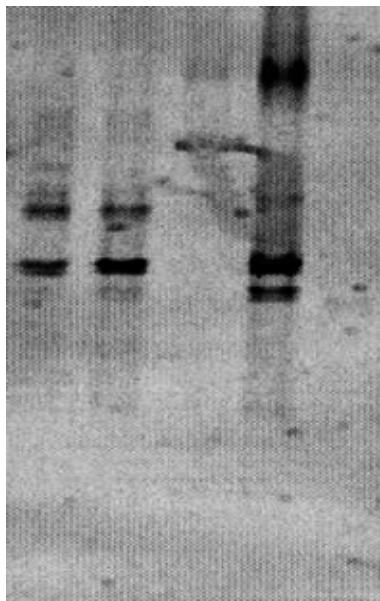


Figure 2.15 Denaturing gradient gel electrophoresis profiles of polymerase chain reaction products: ammonia-oxidizing bacteria for four example wetland cells

The ammonia-oxidizing bacterial community (Figure 2.15) was assessed using group-specific PCR primers, which were unique sequences from 16S ribosomal ribonucleic acid (rRNA) genes, while the denitrifying bacterial community (Figure 2.16) was assessed using primers that were unique sequences from the functional gene nitrous oxide reductase (*nosZ*). A primer is a nucleic acid strand that serves as a starting point for DNA replication and is required because most DNA polymerases (*i.e.*, enzymes that catalyze the replication of DNA) cannot synthesize *de novo* DNA. Ribosomal ribonucleic acid is one of the three major types of ribonucleic acid (RNA). Ribosomal ribonucleic acid and the genes that encode them are ideal biomarkers (Head *et al.* 1998).

The extracted DNA from all samples was diluted tenfold to avoid inhibition of the PCR by humic substances. The PCR amplification was undertaken using forward and reverse primers (CTO189f – GC A/B – GC, CCGCCGCGCGGGCGGGCGG-GGCGGGGGCACGGGGGGAGRAAAGCAGGGGATCG; CTO189r – GC C, CGCCCGCCGCGCGGGCGGGCGGGGGCGGGGGCACGGGGGGAGGAAAGTAGGGGATCG; CTO654r, CTAGCYTTGTAGTTTCAAACGC) for ammonia-oxidizing bacteria as reported by Kowalchuk *et al.* (1997). The gas chromatograph clamps for primers are underlined (if used). The PCR was performed on a PTC-100 thermal cycler (MJ Research, San Francisco, CA, USA) in a 50- μ L mixture.

The forward and reverse primers (*nosZF*, CG (C/T) TGT TC (A/C) TCG ACA GCC AG; *nosZ1622R*-GC, GGCGGCGCGCCGCCC GCCCCGCCCCCGTCG-CCC CGC (G/A) A (C/G) GGC AA (G/C) AAG GT (G/C) CG) targeting the

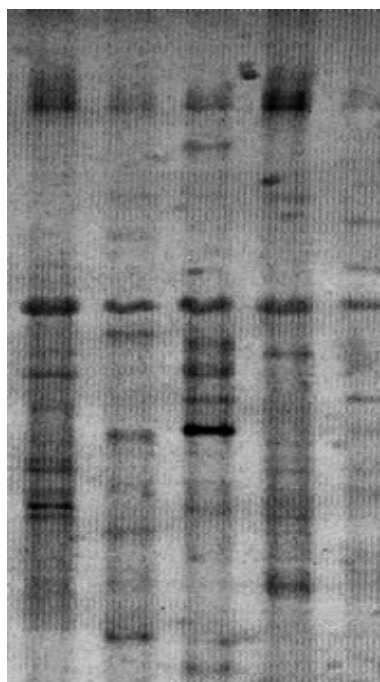


Figure 2.16 Denaturing gradient gel electrophoresis profiles of polymerase chain reaction products: denitrifying bacteria for four example wetland cells

nosZ gene (Throbäck *et al.* 2004) were used in a subsequent PCR. The GC clamps for all primers are underlined.

The PCR was again performed on a PTC-100 thermal cycler in a 50- μ L mixture including 1.33 U of Taq polymerase and 5 μ L of the supplied buffer (including 1.5 mM $MgCl_2$; Roche Diagnostic, Mannheim, Germany), each nucleotide at a concentration of 200 μ M, the primers at 0.125 μ M each, 600 ng μ /l bovine serum albumin, and 2 μ L of the DNA template.

The PCR products of DNA extraction and PCR reactions were examined by agarose gel electrophoresis. The agarose mixture (1% agarose in Tris acetate ethylenediaminetetraacetic acid (EDTA)) was melted by heating in a microwave for approx. 2 min and subsequently poured into an agarose gel casting tray. The solidified gel was covered with an electrophoresis Tris acetate EDTA buffer before running electrophoresis at 120 V for 40 min.

The PCR products and dye supplied with the DNA extraction kit (2 μ L of dye and 4 μ L of PCR products) were placed into loading wells. The first well of each row was loaded with 2 μ L of GeneRuler (1 kb DNA ladder; 1000 base pairs for ammonia-oxidizing bacteria and nitrous oxide reductase *nosZ*) and 4 μ L of distilled water. The electrophoresis was run for 40 min at 120 V (Owl Scientific, Woburn, MA, USA). The gel was then placed in ethidium bromide solution (immersed for 15 min), located in the fume cupboard, and subsequently washed with tap water. The ethidium-bromide-stained gel was then visualized by UV illumination.

2.3.2.4 Statistical Analyses and Limitations

All statistical analyses were performed using the analytical and graphical software tool Origin 7.5. The seasonal effects on water quality improvement were tested by ANOVA at $p < 0.05$.

There was no true experimental replication because all ICW are different from each other, and only one ICW was studied in detail. However, the experimental error was reduced due to an assessment of potential outliers and sub-sampling where required. The limitations associated with the statistical analysis of this case study suggest that the outcomes presented may only be valid for the specific ICW system assessed and very similar systems.

The ICW system studied is semi-natural and open. Considering that flow rates are therefore partly unknown and that flow estimations would result in very inaccurate constituent mass estimations, the display of constituent concentrations was chosen instead. This is common practice for less engineered and rather natural systems.

It has to be emphasized that ICW are not fully engineered treatment wetlands where the inflow and outflow rates are known and where losses to groundwater are zero. Moreover, ICW are purely driven by hydrology (*i.e.*, storm events) and not by rather constant inflow rates, which are common for constructed wetlands treating, for example, domestic wastewater.

2.3.3 Results and Discussion

2.3.3.1 Water Quality

The mean inflow values were as follows: BOD, 540.2 ± 697.13 mg/l; COD, 1501.7 ± 2007.07 mg/l; SS, 261.4 ± 1004.74 mg/l; ammonia–nitrogen, 39.62 ± 41.762 mg/l; nitrate–nitrogen, 3.81 ± 3.375 mg/l; MRP, 11.55 ± 10.049 mg/l; total coliforms, $1.3 \times 10^6 \pm 1.5 \times 10^6$ colony-forming units (CFU)/100 ml; *E. coli*, $9.6 \times 10^5 \pm 10.0 \times 10^5$ CFU per 100 ml; pH, 7.5 ± 0.90 . These values indicate the very high variability of the farmyard runoff quality entering the ICW system (Mustafa *et al.* 2009).

The mean effluent water quality was as follows: COD, 75.5 ± 92.68 mg/l; BOD, 12.9 ± 10.60 mg/l; SS, 16.3 ± 17.61 mg/l; ammonia–nitrogen, 0.37 ± 0.562 mg/l; nitrate–nitrogen, 0.99 ± 1.812 mg/l; MRP, 0.94 ± 0.628 mg/l; TC, $1.0 \times 10^4 \pm 1.3 \times 10^4$ CFU per 100 ml; *E. coli*, 30.1 ± 34.4 CFU/100 ml; pH, 7.6 ± 0.55 . The findings indicate that the ICW system had a high capacity to remove pollutants due to the large size of the wetland cells and the relative high mean retention time. Moreover, microbiological contaminants were removed well, most likely due to predation and unfavorable growth conditions at low temperature (Scholz 2006a).

For example, the overall reduction of 97.6% in BOD is similar to the 97% reported for a comparable system in the state of Maryland, USA (Schaafsma *et al.* 2000), and the 99% calculated for an ecological treatment system comprising a series of reactors, clarifiers, and wetlands in the state of Ohio, USA (Lansing and Martin 2006). Furthermore, the removal efficiency is higher in comparison to other wetlands treating similar types of influents: 65 and 76%, Connecticut, USA (Knight *et al.* 2000 and Newman *et al.* 1999, respectively); 40 to 50%, New Zealand (Tanner *et al.* 1995); 80%, Oregon, USA (Skarda *et al.* 1994).

Concerning other parameters, the COD and SS removal efficiencies were also high: 94.9% and 93.7%, respectively. This can also be explained by the large size of the wetland cells and the high retention times. The mean pH value increased slightly between influent and effluent from 7.5 to 7.6.

2.3.3.2 Comparison of Annual Treatment Performances

There were no significant annual trends in mean ammonia–nitrogen and MRP reduction between 2001 and 2007. Concerning nitrate–nitrogen, however, the mean reduction decreased by 6.2% in 2007, which was, however, statistically not significant. The MRP reductions were similar to the SS reductions, which might indicate that phosphorus is often bound to SS within the inflow water (Scholz 2006a; Mustafa *et al.* 2009).

As the ICW system continued to mature, the microbial communities (see below) and aquatic vegetation became more established, resulting in a stable system with high pollutant removal capacity. The overall nutrient removal efficiency of

the system was high: 99.6% for ammonia–nitrogen, 86.8% for nitrate–nitrogen, and 93.2% for MRP, even in the sixth year of its operation. This finding contrasts with the 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; Kent 2000). This may imply that ICW cannot be compared with traditional treatment wetlands in terms of their capacity to retain nutrients.

The characteristics of soil have an important influence on the magnitude of phosphorus losses. Different factors such as particle size distribution, organic content, and iron and aluminum concentrations influence the ability of wetland soil to hold phosphorus (Sharpley 1995; Leinweber *et al.* 1999; Daly 2000). In general, soils with high clay content have a higher capacity to bind phosphorus than those with sandy and organic soils (Sharpley 1995; Maguire *et al.* 1997; Leinweber *et al.* 1999; Daly 2000).

Also soils with higher iron and aluminum content were found to have a greater capacity to bind phosphorus (Daly 2000). Dunne *et al.* (2005a) demonstrated with the help of an intact soil/water column study that the phosphorus sorption parameters were significantly related to the amorphous forms of iron and aluminum oxides in soils. The soils containing about twice as much amorphous forms of iron and aluminum had a higher phosphorus sorption capacity (1601 ± 32 mg/kg as compared to 674 ± 62 mg/kg). Previous research conducted by Dunne *et al.* (2005a, b) showed that the soil in the current study site area contains high proportions of sand, silt, and clay ($12 \pm 0.3\%$ sand, $54 \pm 2.0\%$ silt, and $33 \pm 1.7\%$ clay), and most likely also relatively high amounts of iron and aluminum, resulting in a greater phosphorus binding capacity of the soil.

Furthermore, as the wetland ages, the continuous accumulation of withered plant biomass increases the phosphorus storage capacity of the ICW system. Moreover, the wetland plants were not harvested, resulting in the accumulation of organic matter (Seo *et al.* 2005). The wetland subsequently changes from an initially mineral-based system to an organic-based system with higher phosphorus removal capacity.

2.3.3.3 Seasonal Performance

Seasonal variations in performance concerning nutrient removal were observed. The ammonia–nitrogen concentrations within the inflow were relatively high in summer and fall compared to winter and spring. In comparison, the ammonia–nitrogen outlet concentrations were higher during fall compared to other seasons. It follows that the mean removal efficiencies were higher during spring (99.4%) and summer (99.7%) and slightly lower during fall (98.7%) and winter (99.3%). Although the mean removal efficiencies in fall and winter were relatively high, the mean concentrations of the inflow drastically increased during these two seasons and the mean concentrations in the outflow were 0.92 ± 0.769 and 0.60 ± 0.201 mg/l, respectively. The ammonia–nitrogen concentrations in the outflow were signifi-

cantly greater ($p < 0.05$) during fall than summer. This can be explained by the observation that there was reduced outflow and a longer retention time during summer compared to fall (see below).

In comparison to previous constructed treatment wetland studies (Gottschall *et al.* 2007; Newman *et al.* 1999), the ICW system has very high nutrient removal efficiencies, indicating its high treatment performance. The MRP concentrations were also efficiently removed in spring (95.7%), summer (90.7%), fall (94.9%), and winter (93.2%). The MRP outlet concentrations were significantly greater during fall than during summer because there was reduced outflow and a longer retention time during summer.

2.3.3.4 Flows

Mean monthly inflows and outflows from the ICW system during the monitoring period (2003–2007) were $180 \pm 217.5 \text{ m}^3/\text{month}$ and $45 \pm 132.3 \text{ m}^3/\text{month}$, respectively. Although inflow to the ICW system was recorded usually between August and December, there was no corresponding outflow during most of this period, indicating that there was only a short discharge period to the receiving watercourse. The flows through the wetlands decreased during summer, and the ICW system was not discharging to the receiving watercourse.

Flow monitoring indicates that approx. 25% of the influent into the ICW system is subsequently discharged at the outlet as effluent. Based on flow and weather monitoring data, the breakups of inflow and outflow compositions are summarized in Figure 2.17. During dry periods, increased storage capacity was created within the ICW cells due to losses via evapotranspiration and infiltration. These processes provided additional storage buffer capacity for runoff within the ICW system prior to discharge during storm events.

The intermittent nature of the outflow and the great loss of partially treated runoff to the ground have a great impact on calculating and interpreting treatment performances. It follows that it is very difficult to accurately determine constituent masses. Therefore, calculations such as removal performances are based on con-

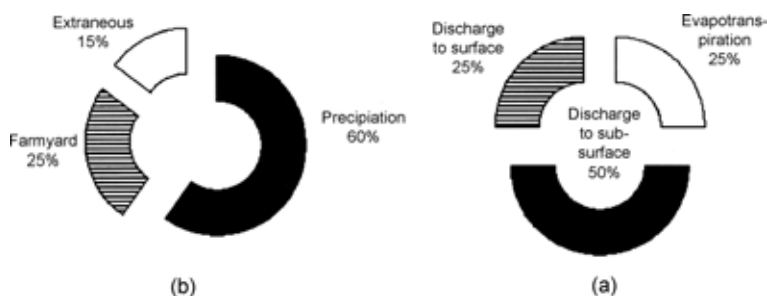


Figure 2.17 Composition of integrated constructed wetland: (a) inflow, and (b) outflow

stituent concentrations. Moreover, the lack of outflow data during drier in comparison to wetter periods is likely to result in less accurate interpretations for summer than winter data.

The microbial transformations within the wetland systems are a function of the available area for biofilm growth. In ICW systems, the dense vegetation stands and the associated litter provide large surface area for biofilm biomass and hence enhanced treatment capability. The biofilm entraps both organic and inorganic solids. The formation of biomass is greatest at the inlet of the wetland cell, where the organic loading is highest (Ragusa *et al.* 2004; Scholz 2006a), and decreases progressively as the wastewater passes through the system. The biofilm not only reduces the pollutant concentrations but also decreases the hydraulic conductivity by reducing the pore volume (Mustafa *et al.* 2009).

In addition to the biofilm formation, the organic matter and humic substances, which develop rapidly within the wetland soils, increase the availability of carbon supporting denitrification and the biological feedback mechanisms that secure water retention. These processes result in flow impedance and, subsequently, self-sealing of the ICW cell. This leads to a progressive contaminant reduction within the discharge to the groundwater.

2.3.3.5 Receiving Stream Water Quality

The construction and commissioning of the ICW system led to the transformation of a non-point source to an identifiable point source of potentially polluted water (*i.e.*, if treatment is insufficient). The land owner has, therefore, the responsibility to maintain the system to uphold BMP. However, there are currently no discharge standards for the final effluent in Ireland.

The final effluent of the ICW is discharged into a small stream through the outlet from cell 4. At the discharge point, mean outlet ammonia–nitrogen concentrations were 0.37 ± 0.562 mg/l, while nitrate–nitrogen and MRP concentrations were 0.99 ± 1.812 mg/l and 0.94 ± 0.628 mg/l, respectively.

The final ICW cell 4 is adjacent to the stream and has the second largest relative reduction in MRP, which is, however, not statistically significant. The most important and statistically significant ($p < 0.05$) observation is the great removal capacity of the first cell, where most SS-containing nutrients settle out.

Most nitrate–nitrogen concentrations at the point of confluence between the outlet of the final ICW cell 4 and the receiving stream, and after this point, are less than the corresponding concentrations for the preceding sampling points measured upstream. Further downstream, some measured nutrient concentrations decrease, indicating that the stream has sufficient assimilative capacity to buffer the ICW outflow concentrations, except for sampling point 14. This can be explained by two factors: the presence of a field drain discharging water rich in nutrients and the narrowing of the stream at that point. However, the trend reversed further downstream because field drains were absent.

Overall, the monitored stream had a median ammonia–nitrogen concentration of 0.05 mg/l, nitrate–nitrogen concentrations of 0.134 mg/l, and MRP concentrations of 0.029 mg. Phosphorus, which is a limiting nutrient, is in compliance with the Irish Phosphorus Regulations (1998). The key nutrient indicator for Irish rivers is MRP, which should be less than 0.03 mg/l for the annual median concentration.

Moreover, the ICW effluent is variable and depends predominantly on the patterns of precipitation. Outflows from the system are highly variable ($1.5 \pm 4.41 \text{ m}^3/\text{d}$) because the inflow quality and quantity of the ICW system is predominantly a function of precipitation intensities and patterns. Furthermore, there is no outflow from the system during periods of very low inflows, indicating that there is only a short discharge period to the receiving stream. Discharge from the ICW system during periods of high precipitation generally occurs during periods when the buffering capacity of the receiving water is enhanced by an increased dilution ratio (Mustafa *et al.* 2009).

2.3.3.6 Groundwater Quality

The ICW site was constructed using *in situ* soils. The subsoil was reworked and used in lining the bed and banks of all cells to reduce the risk of infiltrating pollutants. As the contaminated farmyard runoff passes through the ICW system, the suspended matter settles on the soil surface and subsequently hinders infiltration of contaminants through the wetland cells (Kadlec and Knight 1996; Scholz 2006a; Wallace and Knight 2006).

The mean ammonia–nitrogen concentration of the well located up-gradient was 0.79 mg/l, which indicates pollution but not originating from the ICW system. The nitrate–nitrogen and MRP concentrations were 0.02 mg/l and 0.12 mg/l, respectively. The mean ammonia–nitrogen concentration for the down-gradient well was 0.50 mg/l, which is less in comparison to the other wells located up-gradient and between ICW cells. Other nutrients determined down-gradient were below the detection limit of 0.02 mg/l. It follows that ammonia–nitrogen from the ICW is not impacting negatively on groundwater.

This finding is confirmed by an assessment of the nutrient data for the two wells between cells 3 and 4. The wells located between the ICW cells have a relatively high ammonia–nitrogen concentration because their sampling points are most influenced by the short distance to the cell sediments, which are rich in ammonia–nitrogen. The nitrate concentrations are also well below internationally recommended thresholds of approx. 25 mg/l nitrate.

Generally, the presence of high phosphate concentrations in groundwater is an indication of a shallow subsoil depth and the presence of preferential flow paths through the subsoil. However, in this case, where approx. 50% of the ICW water discharges to groundwater, the mean nitrate–nitrogen and MRP concentrations in all the down-gradient monitoring wells were <0.03 mg/l, indicating that the groundwater in the vicinity of the ICW system is not polluted by infiltration of contaminants from the ICW cells receiving nutrient-enriched runoff.

The soil investigation results at the time of construction indicated the presence of clay in the substratum of the ICW. Furthermore, the presence of hydrogel formed on the detritus confirms the low mobility of nutrients in subsoil and show the important role that subsoil plays in attenuating the pollutants.

Furthermore, there are many biogeochemical processes that play essential roles in impeding the infiltration of pollutants to the groundwater. For example, the ICW vegetation provides a large surface area to support microbial biofilms (Wallace and Knight 2006). Detritus provides a carbon source to microbes for denitrification and assists in the long-term sequestration of phosphorus (Wallace and Knight 2006). The production of methane during anaerobic metabolism inhibits the loss of water through its capillary-pore structures (Kellner *et al.* 2004; Tokida *et al.* 2005).

Virtually all ICW cells remove nutrients successfully. Nutrient enrichment of groundwater beneath the ICW system is of great concern, because high concentrations may have a direct effect on sensitive receptors (EPA 2002). The transportation and attenuation of pollutants in ICW cells depend predominantly on wetland soil, physical impedance and underlying geological formations; *e.g.*, attenuation of ammonia during migration in the sub-surface is known to occur (Erskine 2000). The key processes are sorption (cation exchange) and biological degradation control of ammonium in sub-surfaces (Buss *et al.* 2004). Concerning wetland peat, methane bubbles originating from microbial anaerobic processes lower the hydraulic conductivity (Kellner *et al.* 2004). The biofilm matrix formed on detritus acts like a hydrogel, which withstands changes in fluid shear stresses (Harrison *et al.* 2005).

2.3.3.7 Nitrogen Transformations Within the Sediment

In comparison to ammonia oxidizers, denitrifiers were more abundant in most of the collected sediment samples. Since the nitrate concentrations within the ICW systems were low, it is likely that oxygen and nitrate serve as electron acceptors in the lower layer of the wetland cells (Eriksson and Weisner 1996). This might have promoted the growth of denitrifying bacteria.

Denitrifying bacteria were detected in each cell of the ICW system, while ammonia-oxidizing bacteria were identified in the first, second, and third cells. Denitrifying bacteria were estimated to be present in higher numbers than ammonia-oxidizing bacteria.

The organic material present within the ICW cells also had an indirect impact on the bacterial community. For example, 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 oxygen availability within the sediment was reduced, and subsequently denitrification was supported (Bastviken *et al.* 2005).

There was a reduced availability of organic matter at the bottom of ICW cell 1, where the greatest biofilm formation took place and the maximum biodegradation of organic matter occurred (*e.g.*, >85% BOD reduction). This led to decreased numbers of heterotrophic bacteria and consequently created conditions that favored the growth of ammonia-oxidizing bacteria.

The high number of denitrifying bacteria was linked to the accumulation of plant detritus over the years. The overall heterotrophic activity was increased by the supply of sufficient organic matter. This led to the consumption and subsequent reduction of oxygen within the sediment, thus supporting denitrification (Nielsen *et al.* 1990).

2.3.3.8 Integrated Constructed Wetland Sizing for Nutrient Reduction

Most ICW systems have a multicellular configuration with a minimum number of four cells. The systems operate as a set of sequential containment and treatment structures that intercept and control contaminants. The first cell receives the most contaminated influent. It is lined with organic matter. The loss of water through infiltration to the ground is kept low by the dense vegetation and associated litter with biofilms, while the subsequent wetland cells receive less contaminated water (Carroll *et al.* 2005).

A regression analysis for the 13 ICW located in the Annetstown Stream watershed suggested that for sufficient MRP removal, a good correlation coefficient between the effective ICW area and the farmyard area can be determined (Scholz *et al.* 2007). The required effective ICW area for effective MRP removal should be at least 1.3 times the farmyard area. A factor of two is, however, recommended in practice to be on the safe side.

According to this design criterion, a farmyard area of 5000 m² required an effective ICW area for sufficient MRP removal of at least 6500 m². The actual ICW area was, however, 1.5 times the farmyard area, which is sufficiently large.

Most of the nutrient reductions occur in the initial cells. The effluent should stay as long as possible in the ICW system to allow for the development of mature ICW ecosystems effective in removing pollutants. This is achieved by allowing the pollutant plume to spread as slowly as possible throughout flatly designed ICW cells. Further details on sustainable wetland design have been published by Carty *et al.* (2008).

2.3.3.9 Overall Catchment Characteristics

Figures 2.18 to 2.21 show major overall catchment characteristics. The valley is dominated by pasture (Figure 2.18) used for cattle grazing. The runoff is usually higher in the upland areas, as indicated in Figure 2.19. Nitrogen loading rates are higher in the lowland areas (Figure 2.20). In comparison, no clear trend is visible for phosphorus (Figure 2.21).



Figure 2.18 Land use in the Annestown Stream watershed

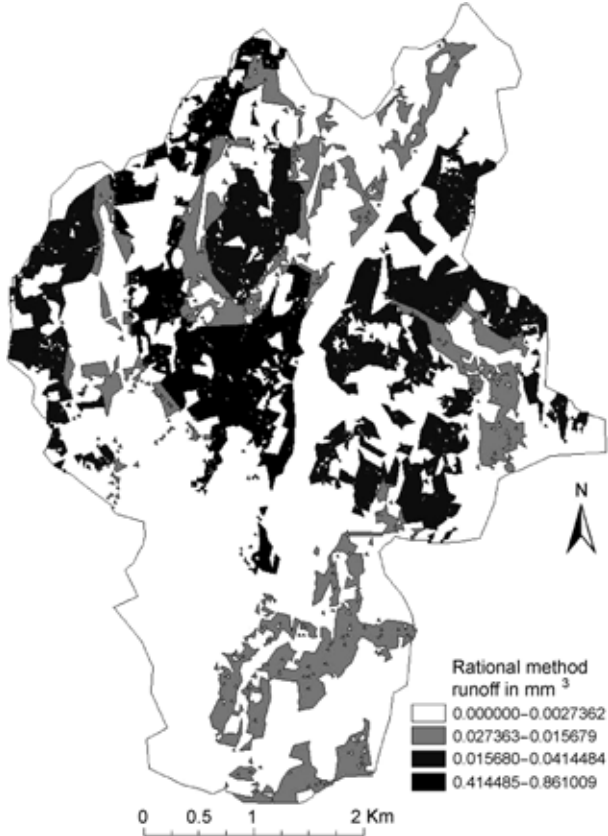


Figure 2.19 Annual runoff generated in the Annestown Stream watershed

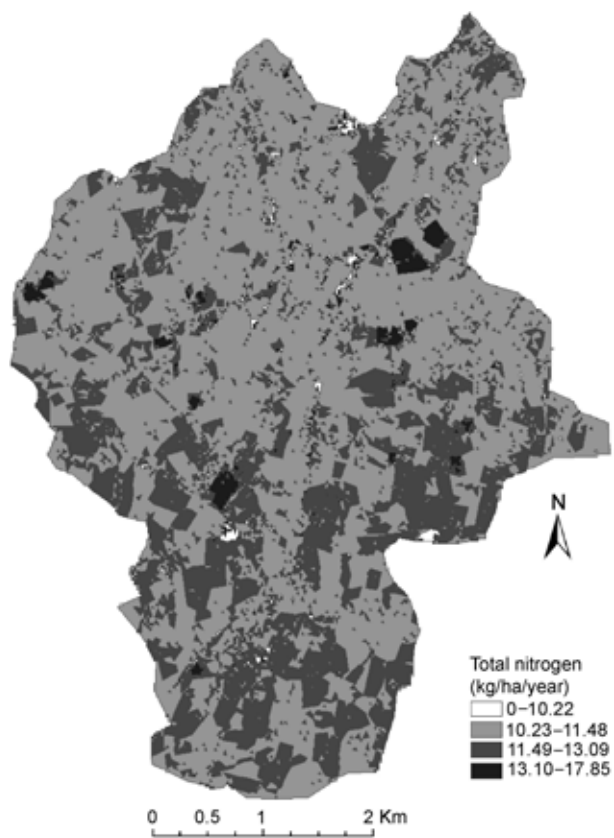


Figure 2.20 Total nitrogen loading rate calculated by the rational method for the Annestown Stream watershed

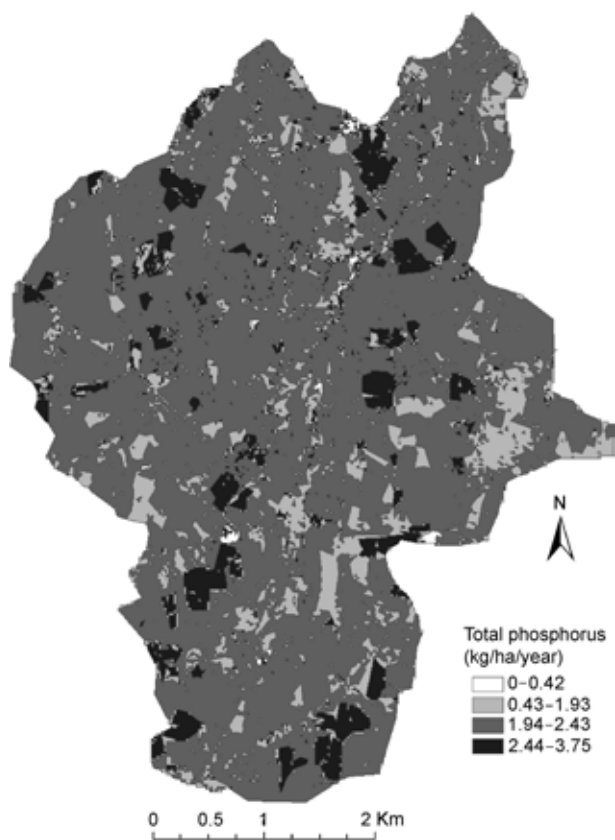


Figure 2.21 Total phosphorus loading rate calculated by the rational method for the Annestown Stream watershed

2.3.3.10 Soft Criteria

Landscape fit is one of the key objectives of the ICW concept (Harrington *et al.* 2005, 2007; Scholz *et al.* 2007b). The case study ICW system was therefore designed by utilizing the natural contours and features of the landscape. The utilization of natural contours resulted in reduced construction costs, leading to a cost-effective design of the ICW. Costs associated with cut and fill volumes were reduced by this technique. The shape of the valley and the crest and slope of hills were taken into account during the integration of the ICW site into the landscape.

Curvilinear shapes of cells that follow the natural contours were utilized to support the even distribution of flows within the system. The growth of vegetation in the cells of this ICW has assisted in the creation of a more natural look and a good fit with the surrounding landscape.

Furthermore, the ultimate aim of an ICW is to ensure that the water quality is improved and that the final result enhances the surrounding landscape and contributes to an improvement in the local biodiversity, as discussed by Carty *et al.* (2008). For example, the common newt (*Triturus vulgaris*) and invertebrates that are important indicators for 'good' water quality became abundant. Furthermore, *Dytiscus circumflexus*, a rare water beetle, has been identified along with mayflies and dragonflies within the case study ICW site boundaries. The ICW system has also attracted birds including the grey heron (*Ardea cinerea*) and mallard (*Anas platyrhynchos*). However, a quantitative assessment of the biodiversity was beyond the scope of this sub-section.

2.3.4 Conclusions

The studied ICW system reduced contaminant concentrations present in the farm-yard runoff. Nutrients including ammonia–nitrogen and MRP were effectively reduced even after 7 years of operation. Hence, the structure acts as a sink (not source) for nutrients. Furthermore, the BOD, COD, SS, total coliforms, and *E. coli* values were also reduced within the ICW system.

Denitrifying bacteria were counted in greater numbers than ammonia-oxidizing bacteria within the sediments of the ICW cells. Thus, the benthos of this system supported denitrification and, consequently, made it a sink for nutrients.

The groundwater and surface water monitoring results indicated that the ICW system had not polluted the groundwater or degraded the water quality of the receiving watercourse. Nutrients including nitrate–nitrogen within receiving water bodies were lower downstream than upstream of the ICW system. However, the case study system is highly complex, and it is therefore difficult to define reliable nutrient balances.

The long-term assessment demonstrates that the example ICW system can be considered an effective and sustainable wastewater treatment option for farm-yard runoff rich in nutrients. Dense stands of wetland plants contributed to very

high evapotranspiration rates, which reduced the likelihood of floods further downstream during the summer by providing additional storage volume. Furthermore, the natural design enhanced the local biodiversity and improved landscape aesthetics.

2.4 Integrated Constructed Wetlands for Treating Swine Wastewater

2.4.1 Introduction and Agricultural Practice

Constructed wetlands have become widespread as an alternative and cost-effective method of treating wastewaters including agriculture-related wastewater streams (Harrington *et al.* 2005; Kivaisi 2001; Scholz and Lee 2005; Sievers 1997; Stone *et al.* 2002). As opposed to traditional mechanical and chemical treatment techniques such as sludge treatment plants or trickling filters, constructed wetlands implement only processes that occur naturally. The effective treatment of swine wastewaters (also called pig slurry in Europe) has been studied in greater depth over the past two decades, with more and more processes being examined and scrutinized.

The majority of research originated in the USA, where government agency guidelines were published for the construction and operation of constructed wetlands for various applications, but predominantly for domestic wastewater and storm water treatment (USEPA 1988, 2000). However, with respect to the complex nature of diffuse agricultural pollution, these guidelines were found to be inadequate (Sievers 1997; Stone *et al.* 2002, 2004) and resulted in mediocre overall treatment results due to the absence of a pretreatment stage promoting sedimentation or nitrification. Particularly livestock waste and wastewater are difficult to treat sufficiently by constructed wetlands due to their high concentrations of pollutants including suspended solids, nutrients, and bacteria (He *et al.* 2006). Moreover, an increasing number of livestock on farms and high production rates result in very high volumes of wastewater that prove to be a challenge to constructed treatment wetlands.

In the European Union, the Nitrates Directive 91/676/EEC (EEC 1991) has put pressure on piggeries due to the restrictions placed on land spreading of corresponding wastewaters at certain times of the year. Considering that suitable farmland for spreading is limited and other landowners are often unable to spread piggy waste as fertilizer, expensive and high-maintenance equipment is often purchased that can put great financial burdens on piggeries and farmers. The holistic approach of using constructed wetlands for the treatment of wastewater is therefore rapidly becoming a far more appealing approach to many farmers. However, constructed wetland acceptance will be hindered without greater understanding of the processes involved and guidance to their construction, operation, and maintenance.

The initially widely used guidelines from the USEPA (1988, 2000) were sufficient for certain types of wastewaters. Long and narrow systems often yielded respectable results despite high flow velocity measurements. This basic design has progressed to more sophisticated wetland system designs adopted to cold, temperate, and tropical climates (Kantawanichkul 2003; Puustinen 2005). Different designs include vertical-flow, horizontal surface-flow, and sub-surface-flow systems (Scholz 2006a).

The basis for the use of constructed wetlands for wastewater treatment is well established and its ‘fine-tuning’ for the use of intense agricultural wastes is ongoing. However, some systems such as ICW have produced exceptionally positive treatment results for farmyard runoff and domestic wastewater (Scholz *et al.* 2007) and are therefore being considered also for swine wastewater treatment.

The aim of this review is to assess the current level of knowledge and understanding with respect to the treatment of wastewater high in ammonia–nitrogen with constructed wetlands. The review will focus on swine wastewater treatment representing a ‘worst-case scenario’ for wetland designers. The reasoning for further research needs incorporating multidisciplinary and holistic views is also outlined.

2.4.2 International Design Guidelines: Global Scenario

2.4.2.1 American Guidelines

Since the early 1980s, a wide range of constructed wetland publications originating from the USA have been published. These include guidance manuals, case studies, bibliographies, assessments of technologies involved (both free surface water wetlands and sub-surface-flow wetlands), and the availability of a description and performance database, the so-called North America Treatment Wetland Database, which is based on constructed wetlands currently in use (Knight *et al.* 2000).

The US Environmental Protection Agency (USEPA) released a manual in 1988 for the design of constructed wetlands for the treatment of municipal wastewaters (USEPA 1988). The design parameters presented in the manual are based on a large number of empirical formulae and led to what is known as the “rational method” (Sievers 1997), which predicts wetland area based upon desired organic matter (5-d biochemical oxygen demand) removal.

The specific processes taking place in constructed wetlands are similar to those that occur in conventional treatment systems including treatment methods designed to promote denitrification. The use of emergent vegetation is crucial in both sub-surface and free surface water systems. The emergent vegetation controls water flow and provides the main source of treatment through the plants’ rhizosphere where most treatment processes take place.

The USEPA (1988) design manual recommended the use of constructed wetlands for the treatment of acid mine drainage and storm water, and further sug-

gested that constructed wetlands could also enhance existing predominantly natural wetlands. The manual acknowledged the considerable costs of more ‘traditional’ treatment systems and emphasized that constructed wetlands are a cost-effective alternative that are based on natural processes occurring in natural wetlands. The construction of wetlands in areas where they did not previously exist highlights that there is less regulation with regard to constructed wetlands than in comparison to natural wetlands. These and similar aspects about the ease of implementation and construction of constructed wetlands have been the driving factors behind the development of constructed wetlands for wastewater treatment since the late 1970s. Moreover, their performance, especially when compared to industrial methods, is a crucial factor promoting their increase of acceptance (Scholz 2006a).

The early design manual (USEPA 1988) was followed up by a series of guidance documents and manuals in the USA and elsewhere (Carty *et al.* 2008; Conte *et al.* 2001; USDA 1991; USEPA 2000). Similar guidelines were released by the United States Department of Agriculture under the Natural Resource Conservation Service (USDA 1991). This handbook recommends constructed wetlands for the treatment of agricultural and domestic wastewater, mine drainage, and storm water. Similar to the USEPA (1988) manual, it also relies on the constructed wetland design based predominantly on biochemical oxygen demand concentrations, but expanded the design parameters by including ammonia–nitrogen as the new key design factor and recommended the implementation of a minimum 12-d residence time for the influent to be treated.

Both manuals are similar, but have led to mixed results. Some reports have shown that nutrient concentrations were significantly higher than expected (Siewers 1997) when using the USEPA (1988) guideline. In comparison, others have shown that results are directly in line with the expected effluent quality using the USDA (1991) guideline (Stone *et al.* 2002).

The manual “Constructed Wetland Treatment of Municipal Wastewaters” (USEPA 2000) is recommended for the design of constructed wetlands as a functional part of a wastewater management strategy, and not as a guidance manual for a “specific regulatory program.” It therefore gives in-depth guidance for both free surface-flow wetlands and vegetated submerged beds.

The USEPA (2000) manual describes constructed wetlands and the associated terminology, applications, and apparent misconceptions in a manner that is easy to understand. It allows for a wide array of people to comprehend the design, application, and implementation of constructed wetlands. However, its attempt to clarify certain aspects of constructed wetland design is confusing. On one hand, it explains that constructed wetlands are well defined by equations based upon a wide range of literature summarizing good research, but it states on the other hand that they cannot be designed purely by such equations. The document then gives example designs using these equations for the design of constructed wetlands for municipal wastewater treatment without outlining other design criteria in detail.

What has been consistent in the implementation of constructed wetlands is the focus on phosphorus treatment. It had been shown on several occasions that initial treatment was quite effective, but that after a relatively short period of time the

reduction levels diminished considerably (Scholz 2006a; Sievers 1997; Stone *et al.* 2002). As phosphorus is deemed very important with regard to water quality, several studies recommend constructed wetlands only as a method of pretreatment when based upon the USEPA (1988) and USDA (1991) manuals.

There is little or no information in early guidance documents with regard to the use of constructed wetlands for the treatment of piggery wastewater. The USEPA online database does include some information for constructed wetlands that are or were being used for the treatment of piggery wastewater, but the information is not comprehensive. Usually, only the number of cells involved in the wetland systems themselves is stated. The USEPA and NRCS documents have both been implemented for the treatment of piggery wastewater and have shown inconclusive findings (Sievers 1997; Stone *et al.* 2002).

Sievers (1997) implemented the USEPA (1988) guidelines for four constructed wetland cells, which were run in parallel; two were surface-flow wetlands and two were free surface water wetlands. Only one cell of each wetland system type received wastewater from a primary anaerobic lagoon, while the remaining cells received wastewater from a secondary lagoon. The results from these cells were highly variable and most likely due to seasonal changes. The inability of the constructed wetland to consistently achieve a biochemical oxygen demand concentration of 30 mg/l or less was presumed to be a result of the wetlands' maintaining anaerobic conditions. Over a 2-year study period, no nitrate was detected in the constructed wetland system, thus suggesting insignificant nitrification. Phosphorus removal was also very low. The primary removal method was adsorption on soil. Despite low overall removal rates, the cells receiving wastewater from the primary lagoon provided relatively good treatment, most likely due to a longer retention time.

Stone *et al.* (2002) used the Natural Resources Conservation Service presumptive design method. This involved four wetland cells (*i.e.*, two cells connected in series). These cells had a length-to-width ratio of 9:1, and both were planted with either bulrushes or cattails (*Typhaceae*). Over a 6-year period, the macrophytes were examined while receiving effluent from an anaerobic lagoon, which was mixed with freshwater prior to entering the wetland.

The wetlands were found to be relatively effective at the removal of nitrogen with mean total nitrogen and ammonia–nitrogen reductions of approx. 85% each over the duration of the study period. At lower rates, the total phosphorus removal rate was approx. 88%. Total phosphorus removal was reduced to 25% and 38% for the two systems, if operated under higher loading rates. Stone *et al.* (2002) suggested that pretreatment would be required for the effective removal of phosphorus when high loading rates were applied. The constructed wetland sizing that was applied to these systems was dependent upon the loading rates and the concentrations of the wastewater being treated. This was found to result in a constructed wetland that was only slightly bigger (approx. 5%) than a similar system designed according to Natural Resources Conservation Service guidelines (see above).

The American guidance manuals (USDA 1991; USEPA 1988, 2000) have been relied upon for use in a wide range of applications with regard to various waste-

waters. However, their usefulness regarding the treatment of swine wastewater was seen to be limited. Most constructed wetland designs lack the treatment capability to reach set standards for swine wastewater without pretreatment (Stone *et al.* 2002; Sievers *et al.* 1997). The manuals, while trying to be robust and highly descriptive, are overburdened with technical data and specific empirical formulae that are supposed to be applied to a treatment method that is extremely complex in both the reactions that are taking place in the constructed wetland as well as the external influences such as climatic conditions. These circumstances are stated in the more recent USEPA (2000) guidance manual for municipal wastewaters and described as a misconception.

Changes in the retention time impact the effectiveness of constructed treatment wetlands. The use of the Natural Resources Conservation Service method requires a minimum retention time of 12 d (Stone *et al.* 2002). Sievers (1997) noted improved performances in cells with longer retention times. This observation is taken into account in the more recent USEPA (2000) guidance manual. Earlier versions recommended between 6 and 7 d retention time.

The American guidelines for municipal water treatment are extensive and provide a large amount of information on all aspects of wetland design. They are recommended for industries and communities that can sustain the “expertise required to maintain them” (USEPA 2000). While being very technical in places, they do provide insight into all the major design aspects. However, there are no specific guidelines with regard to the implementation of constructed wetlands used for the treatment of agricultural wastewater. Therefore, municipal guidelines have been implemented by researchers and operators to examine their potential for use in swine wastewater treatment. However, these designs often fell short of the expected results (Cronk 1996).

2.4.2.2 Other Guidelines

Similar to what has been published in the USA, many other countries have their own guidance manuals for the use of constructed wetlands for the treatment of various wastewaters. Early guidelines were adapted for semi-arid (Australia), boreal (Finland), tropical (Thailand), Mediterranean (Italy), and temperate (Ireland, England, and Scotland) climates as discussed elsewhere (Carty *et al.* 2008; Conte *et al.* 2001; Puustinen and Jormola 2005).

The guidance manuals are used predominantly for the design and construction of wetlands for the treatment of municipal wastewater as well as storm water (Stone *et al.* 2002). Some of these guidelines are relatively new (Conte *et al.* 2001; Puustinen and Jormola 2005) in comparison to some that have been researched and implemented in other regions, namely Australia. A large proportion of the material that is used in the USA is based on data from Australia (USEPA 1988, 2000). Australia has been using constructed wetlands for several decades for the treatment of storm water in urban areas. Indeed, similar to the guidance material in the USA, other manuals and documents also primarily recommend constructed

wetlands for the treatment of (pretreated) municipal wastewaters and storm water (Conte *et al.* 2001).

Several countries with areas characterized by a tropical climate such as China, Thailand, Malaysia, Brazil, and Australia have implemented constructed wetlands for wastewater treatment (He *et al.* 2006; Kantawanichkul 2003; Li *et al.* 2008; Sezerino *et al.* 2003). The functionality of constructed wetlands and their efficiency and size is influenced by the significantly higher temperature of their surroundings. Tropical constructed wetlands, like their temperate counterparts in Europe and the USA, are used predominantly for storm water and wastewater management (Kantawanichkul and Somprasert 2005), but are also being implemented for the treatment of agricultural waste and wastewater. The outline designs that are in use vary very little from those in most other regions, and the main differences are in wetland system operation. Specific guidance manuals have not been developed, and notes refer to the same design types as those used elsewhere. However, a combination of various types of constructed wetlands has been used in some case studies where space was limited (Kantawanichkul 2003, 2005).

Due to the high robustness of constructed wetlands and their wide treatment applications, they can even be found in harsh boreal climates that have extreme weather conditions. For example, the use of constructed wetlands in Finland for agricultural wastewater treatment is relatively new in comparison to other countries (Puustinen and Jormola 2005). Finnish designers rely heavily on optimizing the hydraulic efficiency, preventing flows that are linear, and preferring diffuse flows instead.

Constructed wetlands in Finland, for example, are designed and built with a broader spectrum of use than just a single purpose such as municipal or agricultural wastewater treatment. They are designed for floodwater mitigation, wastewater treatment, habitat creation, biodiversity support, and esthetic functionality (Puustinen and Jormola 2005). The primary Finnish research site is the Hovi constructed wetland in southern Finland (Puustinen and Jormola 2005). In the material documenting this wetland, it states that the size of the constructed wetland with regard to the catchment area is critical. These data are referenced against data from constructed wetlands in the USA and compared with data from Finland.

The high number of studies performed on this single wetland, which was established in 1998, has resulted in two manuals (Puustinen and Jormola 2005) provided by the Finnish Environment Institute. One manual gives an overview of the implementation of constructed wetlands, their role, and general information, and the other provides specific design guidelines.

2.4.2.3 Recent Innovations

The application of constructed wetlands for the treatment of agricultural waste and wastewater is not a new concept, but is one that has been refined since the construction of wetlands became viable for this specific application (Dunne *et al.* 2005a; Hunt *et al.* 2002, 2004; Knight *et al.* 2000; Sievers 1997). There have been several

recent innovations regarding the design, operation, and management of constructed wetlands. For example, different types of vertical-flow wetlands are in use (He *et al.* 2006; Kantawanichkul *et al.* 2001; Molle *et al.* 2008; Sezerino *et al.* 2003). Some of these systems are common for swine wastewater treatment due to their increased capacity to remove organic matter and ammonia–nitrogen (He *et al.* 2006).

Different methods of introducing wastewater into constructed wetlands (*e.g.*, continuous, batch, and “tidal” (Lee *et al.* 2006) flow operation) and “antisized reed bed systems” (Zhao *et al.* 2004), where, in contrast to normal vertical-flow systems with finer material at the top of the system, the coarser material is located at the top instead, have been successfully operated (Lee *et al.* 2006; Scholz 2006a; Scholz and Xu 2001; Zhao *et al.* 2004). Zhao *et al.* (2004) examined the improved capability of such systems to deal with the clogging of the bedding material, as well as avoiding the possible detrimental effects of clogging on functionality and sustainability. The study found very little difference in the actual removal capacity of nutrients in the systems, but the accumulation of material in the set-ups was significantly slower in the antisized systems. In situations where vertical-flow wetlands are used, this would lead to the improved sustainability of the constructed wetland, especially in more remote areas where access to skilled personnel would be more difficult (USEPA 2000).

Another, more encompassing, approach has been undertaken in Ireland with regard to the overall design, construction, and management of wetlands. The ICW concept has been developed for the design of wetlands for agriculture including swine wastewater interception, and the design takes into account rather holistic variables before construction (Harrington *et al.* 2005). These additional variables are primarily of a social, economic, and environmental nature. The ICW approach also integrates wetlands into the surrounding landscape to give a more natural appearance of the overall structure (Scholz *et al.* 2007b).

Carty *et al.* (2008) have published the scientific justification for the Farm Constructed Wetland Design Manual for Scotland and Northern Ireland. This document addresses an international audience interested in applying wetland systems in the wider agricultural context. Most FCW combine farm wastewater treatment with landscape and biodiversity enhancements and are a specific application and class of ICW that have wider applications in the treatment of other wastewater types such as domestic sewage (Scholz *et al.* 2007b). Carty *et al.* (2008) discuss universal design, construction, planting, maintenance, and operation issues relevant specifically for FCW, including wetland systems treating piggery wastewater in a temperate climate, but highlight also catchment-specific requirements to protect the environment.

Furthermore, the design suggestions by Healy *et al.* (2007) complement the guidelines proposed by Scholz *et al.* (2007b) and Carty *et al.* (2008). Healy *et al.* (2007) discuss the performance and design criteria of constructed wetlands for the treatment of domestic and agricultural wastewater, and sand filters for the treatment of domestic wastewater. It also proposes sand filtration as an alternative treatment mechanism for agricultural wastewater and suggests corresponding design guidelines.

Chen *et al.* (2008) examined modified free water surface-flow constructed wetlands to polish treated swine wastewater. The assessed treatment process (*i.e.*, conventional three-stage treatment scheme followed by a modified free water surface wetland (with or without plants) with a 2-d hydraulic retention time) was a promising option to meet Chinese swine wastewater discharge restrictions (COD: 600 mg/l, BOD: 80 mg/l, SS: 150 mg/l).

Most recently, Zhang *et al.* (2009) proposed the application of self-organizing map models as prediction tools for the performance of wetland-based agroecosystems for the treatment of agricultural wastewater to protect receiving watercourses. By utilizing the self-organizing map model, the expensive biochemical oxygen demand outflow concentrations, which are time consuming to measure, were predicted well by other inexpensive variables, which were quicker and easier to measure. This novel approach allows for the real-time control of the outflow water quality of wetland systems and potentially also of other treatment system applications.

2.4.3 Operations

2.4.3.1 Loading and Flow Rates

The operation of standard constructed wetlands is relatively uniform, regardless of the flow type (*e.g.*, surface flow, sub-surface flow, or vertical flow). The influent enters the wetland system and flows through one or a series of wetland cells, which are usually heavily vegetated. Where the differences lie are the various aspects with regard to specific design.

The design of constructed wetlands is therefore of great importance to ensure that they perform well and are suited to the location and to the treatment role that they are supposed to fulfill. Comprehensive research into the most appropriate loading rates, input concentrations, and flow rates has been undertaken (Kantawanichkul *et al.* 2001; Knight *et al.* 2000; Lee *et al.* 2004; Szogi and Hunt 2001).

The North American Treatment Wetland Database contains design data for over 400 constructed wetlands, both study- and full-scale systems. A study sponsored by the Gulf of Mexico Program established a similar database, the Livestock Wastewater Treatment Database (Knight *et al.* 2000). The North American Treatment Wetland Database stores information on all types of treatment wetlands, whereas the Livestock Wastewater Treatment Database stores information specifically for livestock-related wastewaters. These two databases were combined due to their similarity in design to form the North American Database version 2.0 (Knight *et al.* 2000).

The loading and flow rates in constructed wetlands vary greatly. The Livestock Wastewater Treatment Database recorded data from 68 sites in North America. The mean hydraulic loading rate was 4.7 cm/d. Mean system flows of 10 m³/d have been calculated. Most of the systems used for livestock wastewater treatment are of small size with a mean area of only 0.6 ha. Constructed wetlands treating

swine wastewater are slightly larger with mean areas of 1 ha. Knight *et al.* (2000) concluded that concentration removal rates were a function of the inlet concentration and the hydraulic loading rates.

The USEPA (1988) guidelines recommend loading rates of 112 kg BOD₅/ha/d and flow rates of 200 m³/ha/d. The Livestock Wastewater Treatment Database stated mean flow rate of 10 m³/ha/d is significantly lower than this, but the majority of the sites examined in the livestock database were research-based systems of relatively small size.

The loading rates that are used in constructed wetlands vary greatly with respect to their intended use. In dairy wastewater treatment, designers have applied very light loading rates (3.6 g/m²/d; Dunne *et al.* 2005a), heavy rates (Lee *et al.* 2004; USEPA 1988), tidal-flow-governed rates (Zhao *et al.* 2004), and changing rates (Lee *et al.* 2004). Several studies in the tropics with high hydraulic loading rates in constructed wetlands treating swine wastewater have shown that removal rates decrease with significant increases in loading rates (Kantawanichkul and Somprasert 2005; Kantawanichkul *et al.* 2003).

2.4.3.2 Water Depth

The water depths in constructed wetlands are highly variable. The USEPA (1988) guidelines recommend that the variable depth should be part of the design equations that are used to determine the sizing of the corresponding constructed wetland cells. Wetlands range from shallow systems with a depth of approx. 0.25 m (Harrington and Ryder 2002; Harrington *et al.* 2005) to deep ones of 1.2 m (USEPA 1988). A technical report by the USEPA (1999) recorded water depths of between 0.1 and 1.5 m. The depth of water is often the key parameter to nitrogen control in wastewater treatment. Shallow water depths are associated with the highest ammonia–nitrogen diffusion and nitrogen losses (Szogi and Hunt 2001).

The water depth also has an important impact on the growth of macrophytes that are used in constructed wetland systems. Depending on the species that are planted, greater water depth can inhibit macrophyte growth and colonization of the wetland cells. Some genera of macrophytes are capable of coping with deep water levels (Clarke and Baldwin 2002; USEPA 1988), but others are not and therefore require shallower water for most of their growing season. High water levels would result in wetlands being more susceptible to the effects of high nutrient concentrations and may lead to the death of more sensitive species. Shallower systems also help to increase nitrification by increasing the aerobic conditions present in the cells (Carty *et al.* 2008; Scholz 2006a).

2.4.3.3 Pretreatment of Wastewater

Pretreatment is often important to achieve a better and more effective treatment of wastewaters (Cronk 1996; Hunt and Poach, 2001). Early guidelines (USEPA 1988)

recommended the use of conventional pretreatment units such as sedimentation basins to reduce suspended solids and BOD concentrations and suggested the addition of chemicals to remove phosphorus.

Constructed wetlands are often viewed as long-term solutions to wastewater treatment. They are low-tech and low-cost and have a design life of usually more than 25 years. However, their longevity can be impeded by the accumulation of detritus and sediment build-up. The removal of solids in a pretreatment stage has therefore been recommended for many years (Hunt and Poach 2001; Cronk 1996; USEPA 1988). Some studies have suggested that some constructed wetland types themselves can be used as a pretreatment step, being part of a sustainable drainage system, also called BMP in the USA (Cook *et al.* 1996).

Furthermore, the dilution of wastewaters to improve treatment is a common practice in constructed wetland operation (Harrington *et al.* 2005). This method is often just a simple addition of water to wastewater. However, heavily polluted wastewater can also be diluted by less contaminated wastewaters such as roof and yard runoff. The dilution of wastewaters is important in promoting good nutrient removal within constructed wetlands. Moreover, if the organic loading rates are excessive, this can result in decreased removal performances (Kantawanichkul *et al.* 2003) and an increase in the risk of ammonia toxicity to some constructed wetland plants (Hunt *et al.* 2002, 2004).

Partial nitrification of wastewater prior to its treatment in a constructed wetland has also been examined as a means of affecting nitrogen removal and ammonia volatilization (Sommer *et al.* 1993; Poach *et al.* 2003). The most common method of additional nitrification of wastewaters is the recirculation of the wastewater itself or the use of partially nitrified lagoon wastewater, as shown by numerous previous researchers (He *et al.* 2006; Humenik *et al.* 1999; Hunt and Poach 2001; Kantawanichkul *et al.* 2001; Poach *et al.* 2003).

Pretreatment to achieve nitrification can reduce ammonia volatilization due to reduced concentrations of ammonia in the wastewater (Poach *et al.* 2003). As a result, these reduced concentrations help to minimize the potential risk of ammonia toxicity to wetland plants. The anaerobic conditions that exist in wetland soils (Scholz 2006a) limit the rate of nitrification, suggesting that denitrification in wetlands is nitrate-limited (Hunt and Poach 2001; Hunt *et al.* 2002, 2004).

Recirculation of wastewaters has been shown to have a positive effect on total nitrogen removal across the world. Kantawanichkul (2001) showed that the recirculation of effluent in a combined vertical-flow and horizontal-flow wetland system increased the total nitrogen removal rate from 71 to 85%. Humenik (1999) reported that nitrified lagoon water added to constructed wetland microcosms led to nitrogen removal rates that were four to five times that of non-nitrified liquid.

Pilot-scale plants in China (He *et al.* 2006) were used to test for various recirculation rates. Findings showed increased ammonia–nitrogen removal rates in comparison to non-recirculated effluent. However, the researchers reported no great increase in phosphorus removal. This study showed that the use of recirculation helped to form an oxide environment in the wetland.

Furthermore, the partially nitrified state of recirculated effluent has consistently shown its benefit to nitrogen removal in constructed wetland systems. Recirculation technologies decrease ammonia volatilization, which is not desirable due to it being an atmospheric pollutant (Sommer *et al.* 1993; Poach *et al.* 2004), and help to abate ammonia toxicity with regard to the most commonly used macrophytes.

2.4.4 Macrophytes and Rural Biodiversity

2.4.4.1 Macrophyte Types and Characteristics

The range of macrophytes that are planted in constructed wetlands is wide and varied. They are intrinsic to the use of constructed wetlands and play a vital role in nutrient removal (Brix 1994; Hunt and Poach 2001). The USEPA (1988) guidance manual refers to cattails, reeds, rushes, bulrush, and sedges, all of which have different ranges of pH tolerance. For example, cattails usually tolerate pH values of between 4 and 10, while other aquatic plants such as rushes and sedges have much narrower tolerance margins. The USDA (1991) guidebook on constructed wetlands lists several species that have been identified as being suitable for use in constructed wetland systems in North America. The guidebook also states that not all wetland plants are suitable for treatment systems, since they should be able to tolerate continuous flooding and exposure to high nutrient concentrations in the influent (USDA 1991). Tanner (1996) summarized a list of properties that wetland plants should have:

- ecological acceptability (no significant weed or disease risks or danger to the ecological or genetic integrity of surrounding natural ecosystems);
- tolerance of local climatic conditions, pests, and diseases;
- tolerance of pollutants and hypertrophic waterlogged conditions;
- ready propagation and rapid establishment, spread, and growth; and
- high pollutant removal capacity.

The principal functions that macrophytes provide are numerous: stabilization of the beds, provision of physical filtration, prevention of clogging of vertical systems, insulation against frost in winter, and provision of a large surface area for microbial communities, which are vital to successful wastewater treatment (Brix 1994; Scholz 2006a). In addition to supporting the treatment processes in constructed wetlands, macrophytes also serve highly underrated functions in traditional civil engineering design by promoting natural aesthetics and landscape integration.

Furthermore, planting of the most suitable and often native species is important in the ICW concept to improve the biodiversity of the vicinity around the structure (Harrington *et al.* 2005; Scholz *et al.* 2007b). The predominantly aquatic plants provide habitats for wildlife such as mammals, birds, and insects. The biodiversity of macroinvertebrates has been shown to be extremely high in certain ICW in

Ireland, *e.g.*, some wetland systems have up to 60% of the country's native species of aquatic macroinvertebrates present. The adaptation of wetland plants to live in anaerobic soils is important as their root structures provide aerobic areas that help to sustain nitrifying bacteria (Brix 1994). As well as providing oxygen for bacteria, they also provide oxygen to the anaerobic substrate and thus help to stimulate aerobic decomposition.

2.4.4.2 Toxicity Tolerance Thresholds

The toxicity tolerance thresholds and the corresponding uptake rates of pollutants by wetland plants have been researched (Brix 1994; Harrington 2005; Hill *et al.* 1997; Hubbard *et al.* 1999). However, these studies have usually examined the more common genera used in constructed wetlands (Brix 1994; Clarke and Baldwin 2002; Hubbard *et al.* 1999). For example, Clarke and Baldwin (2002) tested common species such as softrush (*Juncus effuses* L.), broadleaf arrowhead (*Sagittaria latifolia* Willd.), softstem bulrush (*Schoenoplectus tabernaemontani* C.C. Gmel.), lesser bulrush (*Typha angustifolia* L.), and common bulrush (*Typha latifolia* L.) at varying ammonia concentrations and water depths.

Other studies have assessed more genera growing in temperate or tropical climates (Tanner 1996). However, findings concerning the effect of ammonia on plants are not fully conclusive. The preference to pretreat wastewater prior to it entering the wetland system or the recycling of effluent suggests that the aquatic plants studied were most likely susceptible to ammonia toxicity, although some studies suggest that the plants are more tolerant than is commonly reported in the literature, stating that there was no apparent effect on some plants due to relatively high ammonia concentrations (Hill *et al.* 1997).

Comparisons between different plant species have been undertaken to examine their uptake rates (Hubbard *et al.* 1999; Poach *et al.* 2003; Tanner 1996). Brix (1994) reported on the uptake rates of common emergent, free-floating, and submerged plant species in wetlands. For example, bulrush (*Typha latifolia* L.) had an impressive nitrogen uptake rate for relatively small planted areas, but low phosphorus uptakes for considerably larger areas. The nutrients are, however, bound in the biomass but could be removed by harvesting.

With plants being an important integral part of constructed wetlands, attention has been paid to the opportunity to use them for additional purposes. Therefore, cash crops such as soybean and rice have been assessed in terms of their use in wastewater treatment (Humenick *et al.* 1999; Szogi *et al.* 2000, 2003, 2004).

These research studies indicate that such plants are able to grow in treatment wetlands receiving swine wastewater. The potential yield from such cash crops could make constructed wetlands that use these plants more attractive, particularly in developing countries. Constructed wetlands could be used for the treatment of wastewater and also to yield a steady food supply or income. This would be particularly beneficial for small-scale farmers, because they would be able to produce their own feed while treating their own wastewater at the same time.

Alternative methods of using aquatic plants are not limited to cash crops. The most common macrophytes planted on floating mats in anaerobic lagoons treating swine wastewater have been assessed by Hubbard *et al.* (2004). The nutrient uptake rates were relatively high. Less commonly used plant species native to certain regions have been examined as well. For example, vetivergrass (*Vetiveria zizanioides* Nash) was used in Thailand (Kantawanichkul *et al.* 2003). This grass was suitable for tropical hydraulic and organic loading rates.

2.4.5 Nutrients

2.4.5.1 Nutrient Transformation Processes

The primary objective of constructed wetlands is the removal of excessive concentrations of nutrients in wastewater (Lee *et al.* 2004; Poach *et al.* 2002; Prantner *et al.* 2001). The removal is supported by means of sedimentation, adsorption, organic matter accumulation, microbial assimilation, nitrification, denitrification, and ammonia volatilization (Brix 1994; Poach *et al.* 2003). However, denitrification is far more preferable to ammonia volatilization as a means of nitrogen reduction, considering that ammonia volatilization results in ammonia gas release, which is an atmospheric pollutant (Sommer *et al.* 1993; Poach *et al.* 2004). Denitrification, however, is limited in constructed wetlands by the availability of nitrate and nitrite (Hunt and Poach 2001; Hunt *et al.* 2002, 2004).

Denitrifying enzyme activity is correlated to areas where there are high amounts of nitrate and nitrite. Moreover, denitrifying enzyme activity has been shown to increase over time with the maturity of constructed wetlands, an increased rate of nitrogen application, and water depth (Hunt *et al.* 2002), which leads to increased ammonia volatilization. This is seen as a particular problem, especially with regard to swine wastewater and its high nutrient content.

The phosphorus cycle is dissimilar to both the carbon and nitrogen cycles in that it does not involve a series of oxidation-reduction reactions. In comparison, it is predominantly a sedimentary cycle (Van der Valk 2006), as shown in Figure 2.22.

The most common method of enhancing denitrification is to recirculate the wastewater or to add partially nitrified water (He *et al.* 2006; Kantawanichkul *et al.* 2001; Poach *et al.* 2003). This is done by recycling the effluent back into the system or by the addition of partially nitrified storage or lagoon water. It follows that denitrification is promoted by supplying the system with greater amounts of nitrate and nitrite throughout the treatment process, thereby reducing the risk of volatilization. However, a complete removal of the volatilization process is unrealistic.

Seasonal and temperature effects on denitrification have also been examined (Reddy *et al.* 2001; Trias *et al.* 2004). There is a moderate positive correlation between temperature and the removal rates in wastewaters. Trias *et al.* (2004)

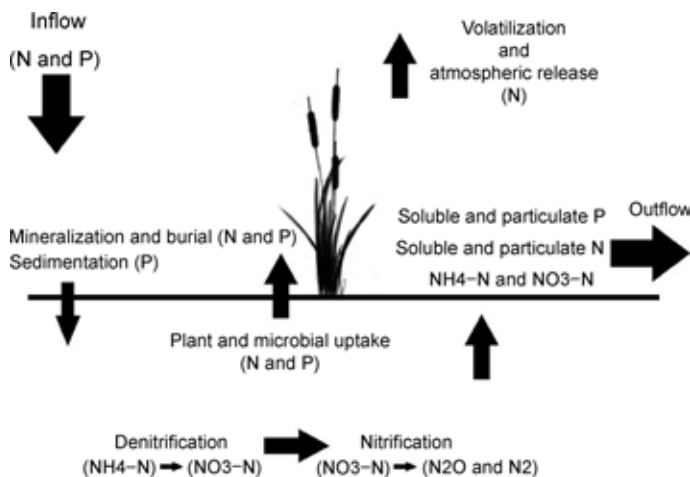


Figure 2.22 Simplified phosphorus and nitrogen cycles within integrated constructed wetlands treating swine wastewater

reported variable findings for swine wastewater treatment with respect to total suspended solids ranging from 77% at moderate temperatures to 42% during the warmest period.

Studies with marsh–pond–marsh designs have assessed differences in the amount of ammonia volatilization taking place in constructed wetlands (Poach *et al.* 2004; Reddy *et al.* 2001). While Reddy *et al.* (2001) saw rates of nitrogen removal similar to those of continuous marsh systems (often >70%) and medium phosphorus removal rates (30 to 45%), it was not observed how much of the nitrogen removal was associated with volatilization.

Poach *et al.* (2002) highlighted the fact that it was not known if volatilization of free ammonia governed nitrogen removal in constructed wetlands. Findings indicated that between 7 and 16% of the nitrogen removal was achieved by ammonia volatilization. This gave an indication as to how much nitrogen removal was being caused by volatilization, but also showed that it was not the principal nitrogen removal method.

Poach *et al.* (2004) used the marsh–pond–marsh design a further time to highlight the differences between the pond and marsh sections with regard to volatilization. The pond sections had significantly higher proportions (23 to 36%) of volatilization than the marsh areas (<12%). Volatilization was the dominant nitrogen removal mechanism in the pond sections (54 to 79%). It follows that marsh areas should be constructed within wetland systems used for the treatment of wastewaters from confined animal operations such as swine waste and wastewater. However, relatively low removal rates of nitrogen (30%) and phosphorus (8%) have been reported for long and narrow marsh–pond–marsh systems in the state of North Carolina, USA (Stone *et al.* 2004).

2.4.5.2 Phosphorus

Phosphorus is among the most difficult nutrients to remove from wastewater. This has created problems for many types of treatment systems such as constructed wetlands, where it is retained but often only temporarily (Pant *et al.* 2001; Reddy *et al.* 2001; Sievers 1997; Stone *et al.* 2002, 2004). Like nitrogen, phosphorus is partly removed in constructed wetlands by plant uptake, accretions of wetland soils, microbial immobilization, retention by root bed media, and precipitation in the water column (Reddy *et al.* 2001; Scholz 2006a).

Phosphorus removal rates in the literature are very diverse and subject to the boundary conditions of the individual case study. Some research studies with constructed wetlands have shown average phosphorus removal rates (He *et al.* 2006; Knight *et al.* 2000; Lee *et al.* 2004; Reddy *et al.* 2001; Sezerino *et al.* 2003). However, some other studies have reported very low rates (Shappell *et al.* 2007; Stone *et al.* 2004; Szogi *et al.* 2004), while a relatively large proportion of investigations have shown very high removal rates, usually >70% (Harrington *et al.* 2005; Hunt and Poach 2001; Prantner *et al.* 2001).

In wetland soils, phosphorus is present as soluble or insoluble, organic or inorganic complexes (Scholz and Lee 2005). The phosphorus cycle is not gaseous, as opposed to that of nitrogen; rather it is sedimentary and is retained in the wetland (Figure 2.22). Inorganic or mineralized organic phosphorus may be retained by oxyhydroxides of iron and aluminum in acidic soils and by calcium minerals in alkaline soils (Reddy and D'Angelo 1997; Scholz and Lee 2005). Soluble phosphorus can be removed by periphyton and subsequently by deposition of dead biomass on soil and detritus surfaces (Reddy and D'Angelo 1997).

In aerobic conditions, insoluble phosphates are precipitated with ferric iron, calcium, and aluminum (Drizo *et al.* 2002; Scholz and Lee 2005). This has led to experiments with various aggregates that are used as the primary substrate in constructed wetlands (Drizo *et al.* 2002; Scholz and Xu 2001). Most of these compounds readily bond to phosphorus and are seen as an effective method of increasing or sustaining phosphorus removal and retention within constructed wetlands.

The partly aerobic and partly anaerobic conditions within wetlands are important for phosphorus removal. In anaerobic soils and detritus, forms of organic phosphorus are relatively resistant to enzyme hydrolysis, which would otherwise release the phosphorus back to the bioavailable pool of nutrients (Reddy and D'Angelo 1997). In aerobic conditions, however, phosphorus is bound to organic matter and is incorporated into bacteria, algae, and macrophytes (Brix 1994; Reddy and D'Angelo 1997; Scholz and Lee 2002).

As phosphorus is retained in the constructed wetland itself or released in low concentrations as part of the wetland discharge, there has been concern among wetland critics with regard to the long-term ability of constructed wetlands to retain phosphorus effectively. Nevertheless, there seems to be an agreement that wetland size matters, *i.e.*, the larger the wetland, the better the long-term removal of phosphorus through adsorption and precipitation (Harrington *et al.* 2005; O' Sullivan *et al.* 1999; Pant *et al.* 2001). It follows that an appropriately sized

wetland system has a long design life and its capacity to treat phosphorus and other nutrients can be greatly increased. Moreover, the design life can be enhanced by harvesting macrophytes at the end of the growing season. This maintenance method leads to the removal of large amounts of phosphorus captured within a relatively small amount of biomass.

2.4.6 Pathogens, Odor, and Human Health

The ability of wetlands to remove pathogens and bacteria is well documented. Wetlands are similar to biofilters, which are governed by the following processes (Brix 1994; Scholz 2006a):

- chemical: oxidation; adsorption; exposure to toxins released by microorganisms;
- biological: antibiosis; ingestion by nematodes, protozoans, and cladocera; lytic bacteria; bacteriophage attacks; natural decay; and
- physical: filtration; sedimentation; aggregation; ultraviolet radiation.

The removal of pathogens from swine wastewater is important for human health and safety reasons. Pathogens such as *Salmonella* may enter the human body via surface waters contaminated by runoff containing traces of swine wastewater, which has been land-spread. Considering the treatment of swine wastewater with constructed wetlands, very high removal rates of enteric microbes and pathogens such as *Salmonella*, faecal coliforms, and *E. coli* (Hill and Sobsey 2001) as well as *Giardia* and *Cryptosporidium* (Karim *et al.* 2004) have been reported.

Hill and Sobsey (2001) documented pathogen removal ratios of between 70 and 90% in surface-flow wetlands and between 80 and 99.99% in sub-surface-flow wetlands with varying organic loading rates. A study of the removal of enteric pathogens showed that there were greater occurrences of *Giardia* cysts and *Cryptosporidium* oocysts in the sediments than in the water column itself. This may be indicative of the greater removal rates with regard to pathogens in sub-surface-flow systems.

Odor removal in wetlands has become an important factor with regard to the construction of wetlands for agricultural wastewater treatment systems (Harrington *et al.* 2005). The capacity for well-vegetated systems to reduce malodors is important for their acceptance, particularly in populated areas. Wetland acceptance would result in their wider application in wastewater treatment, especially for municipal systems and rural farming communities.

The odor-reduction capability of livestock wastewater treatment wetlands has been primarily anecdotal (Wheeler *et al.* 2007). However, studies have been performed to quantify odor from constructed wetlands treating swine wastewater. Findings indicate that constructed wetlands are very capable of reducing malodor from the wastewater being treated, which is seen as an additional benefit (Wheeler *et al.* 2007).

2.4.7 Conclusions and Further Research Needs

Wetlands have been part of human society for hundreds of years, but it is only in the past several decades that they have been used with the specific intent and purpose of treating wastewater. Different wetland systems, regardless of their given title (*e.g.*, constructed wetlands, treatment wetlands, or ICW), have proven to be highly efficient at treating agricultural wastewaters. A wide range of studies on their operation and inner workings allowed for much better understanding of the biochemical processes that take place. This led to their application in the treatment of high strength ammonia–nitrogen polluted waters such as swine wastewater, which is perceived to be difficult to purify because of its high strength and ammonia content.

Due to their high efficiency and relatively low land use, cost, maintenance, and operation effort, wetland systems are ideal as an alternative to traditional wastewater treatment technologies for the treatment of swine wastewaters. The restrictions that are put upon piggery farmers, for example, by the European Union Nitrates Directive prevent farmers from using methods of wastewater disposal such as land spreading that are not sustainable. The use of constructed wetlands for the treatment of swine wastewater, either as its main treatment method or as a secondary or even tertiary treatment, has shown consistently high removal rates for nutrient and other pollutants as well as a reduction of distinctive malodor that is associated with swine waste.

The greater use of ICW should be considered as a standalone technique or in conjunction with traditional aerobic digesters or land-spreading methods to improve the overall treatment efficiency, reduce costs, and allow for a more holistic approach to wastewater treatment.

There is a further research need for more well-documented case studies demonstrating good practice in the treatment of ammonia-rich wastewater with ICW systems. The greatest challenges entail overcoming the toxicity thresholds for different plant species and avoiding groundwater contamination if no impermeable artificial liner is applied for economic reasons. Furthermore, there is the potential that novel analytical tools such as self-organizing map models may help to improve traditional constructed wetland guidelines and manuals by optimizing design calculations based on artificial intelligence.

2.5 Wetlands to Control Runoff from Wood Storage Sites

2.5.1 Introduction and Objectives

Handling of wood in different shapes and forms at small or industrial scales takes place in all regions with climatic conditions allowing trees to grow and people to reside. Wood is an important natural resource and is used all over the world as, for

example, an energy source, building material, raw material for paper, furniture, and tools, and for art and decoration purposes. The environmental impact of logging, transport, storage, debarking, sawing, milling, chopping, and pulping of wood depends on the scale and location of the operation, tree species, handling methods, and preventative measures that have been chosen to reduce the corresponding impacts on the environment and receiving watercourses (Zenaitis *et al.* 2002).

The environmental impacts from sites handling wood include emissions of particles (Kauppinen *et al.* 2006) and volatile organic compounds from wood (Rice and Erich 2006; Welling *et al.* 2001), emissions from the use of energy fuels (Zelikoff *et al.* 2002), release of storm water that has been in contact with wood or irrigation water used to protect wood (Bailey *et al.* 1999; Woodhouse and Duff 2004), and emissions and spillages from machines and vehicles used at the site (Orban *et al.* 2002; WDOE 1995). Furthermore, noise (Kraus 1985), odor (Nicholson 1987), and light (Chalkias *et al.* 2006) emissions from wood handling sites are also seen as undesirable.

This section focuses on the impacts of wood storage and the corresponding mechanical wood handling processes on the receiving watercourses and the environment and assesses the techniques that could be used to reduce adverse impacts. The specific environmental effects of logging and transport (Lindholm and Berg 2005) are not discussed in this section. A detailed discussion on the environmental impacts of water emissions from pulp and paper mills (McMaster *et al.* 2006) is also excluded because the impact from these sites originates to a large extent from the extensive use of chemicals. However, a summary of experimental treatment techniques for organic material originating from wood in effluent from pulp and paper mills is included in this section because these methods can also be used for runoff treatment from other sites handling wood.

This sub-chapter summarizes the findings published in high-impact journal papers to enable further research into developing cost-effective and low-tech treatment methods to reduce the environmental impact of runoff from wood handling sites. The first sub-section presents the background and purpose of this section. The second sub-section describes the pollution potential that storage and different kinds of mechanical handling of wood have on the groundwater or the receiving watercourses in the vicinity of the handling sites. The third sub-section summarizes the methods that are currently applied to reduce the pollution. The section concludes with a brief discussion regarding the cost-effectiveness of the different methods for reducing the environmental impact of runoff, a summary of conclusions, and key recommendations for relevant future research.

The sub-chapter focuses on wetland treatment and soil infiltration because this kind of treatment can, given suitable ground conditions, be applied at sites of different sizes at a relatively low cost. Some more high-tech methods are also discussed briefly. The objectives are as follows:

- to summarize the reasons for pollution generation at wood handling sites;
- to characterize the runoff and its effect on the receiving watercourses;

- to summarize treatment methods including soil infiltration and wetland treatment; and
- to discuss treatments (used in recent experiments) for organic matter in pulp and paper mill wastewater.

2.5.2 *Pollution Potential of Runoff from Wood Handling Sites*

2.5.2.1 Reasons for Pollution Generation at Wood Handling Sites

Contaminated storm water runoff from log yards is generated when precipitation comes into contact with wood, woody debris, and equipment at outdoor wood sorting, processing, and storage facilities (Woodhouse and Duff 2004). Runoff can also be generated by applications of water for dust and fire control (Orban *et al.* 2002). In northern Europe and Canada, it is also common that stored logs are irrigated with water to protect them from cracking and biological attack (Webber and Gibbs 1996). The water keeps the moisture content in the sapwood above 50%, which prevents them from drying and being damaged by fungi and insects (Liukko and Elowsson 1999). There are techniques, such as climate-controlled sprinkling (Liukko 1997), that can be applied to minimize the amount of sprinkling water (Figure 2.23). The sprinkling intensities can be decreased with this method by between 31 and 97% over a 24-h period (Liukko and Elowson 1995). However, runoff is not completely eliminated because of necessary safety margins and maintenance water pressures in the sprinkling systems. A medium-sized log yard in central Sweden with climate-adapted sprinkling in a non-recirculating system uses approx. 100,000 m³ water between May and September. During this period, the amount of log yard runoff may be as high as 70,000 m³ (Jonsson 2004).



Figure 2.23 Log sprinkling resulting in large amounts of sprinkling water that runs off from a log yard

Soluble compounds and particles from bark and wood are taken up by the water and become part of the site runoff (Bailey *et al.* 1999; Woodhouse and Duff 2004). The pollution potential of the runoff depends on factors such as tree species, the amount of water that is in contact with the wood, whether irrigation water is collected and recirculated, if the runoff is treated to reduce the concentration of pollutants, and the size and condition of the receiving watercourse. If the runoff is released to a small watercourse, it will have a considerable negative effect on the ecosystem simply due to the resulting comparatively large proportion of polluted water (Hedmark and Scholz 2008).

2.5.2.2 Characteristics of Runoff

Because of dissimilar conditions at different log yards, scientific studies have come to different conclusions concerning the pollution potential of runoff from wood handling sites. There are considerable differences in the characteristics of the runoff. These differences exist for various reasons; *e.g.*, the species of tree stored on the log yard, the proportion of the runoff that has come in contact with stored wood, and the dilution or pretreatment of the runoff before sampling. Different species of trees contain varying concentrations and types of soluble compounds, and the ease with which the resultant extractives are leached from the wood also greatly influences the runoff concentration and the corresponding toxicity (Zenaitis *et al.* 2002). It is also reasonable to assume that the duration of time for which the wood has been stored before the samples are taken influences the concentration of pollutants in the runoff because of degradation processes in the wood (Feist *et al.* 1971).

Despite the differences between studies, some general conclusions can be drawn from the characteristics of various runoffs. It appears that the main problem connected with pollution from log yard runoff is the high amount of organic matter that results in increased oxygen demand when it degrades in the receiving watercourse (Jonsson *et al.* 2006). Some of the organic compounds may also be toxic to aquatic plants and animals. In addition, the increased concentrations of phosphorus in log yard runoff might lead to considerable loads in the large runoff volumes produced from a sprinkled log yard every year (Hedmark and Scholz 2008).

The amount of organic matter in runoff is generally relatively high if compared to concentrations in the receiving watercourses (*e.g.*, www.environment-agency.gov.uk). The published studies show that water-soluble compounds present in bark and wood are released and appear in high concentrations in the corresponding runoff.

Some researchers have gone into detail to find out which organic substances are present in runoff (Borgå *et al.* 1996a; DeHoop *et al.* 1998; Field *et al.* 1988; Tao *et al.* 2005; Taylor *et al.* 1996; Wang *et al.* 1999). The compounds that are generally found are phenols, resin acids, tannins and lignins, and volatile fatty acids. These compounds may be toxic to aquatic plants and animals. Of the organics extracted from softwood, tannins, lignins, phenols, tropolones, and resin acids are

of greatest concern because they are seen to be contributors to runoff toxicity (Samis *et al.* 1999). Two studies (Bailey *et al.* 1999; Doig *et al.* 2006) also found elevated concentrations of metal (zinc and aluminum, and zinc, respectively). In the study by Bailey *et al.* (1999), the high concentrations of zinc were thought to originate from buildings in the studied sawmill area.

The amount of particles in the runoff can also be relatively high. A study in British Columbia, Canada, focused on the characterization of particles in fresh and primary treated log sort yard runoff and found that the suspended and colloidal particles in the runoff were largely organic in nature (Doig *et al.* 2006).

Concentrations of phosphorus can be sufficiently high as to cause eutrophication in the receiving watercourses (Scholz *et al.* 2007). Another common characteristic of runoff is that the concentration of nitrogen is generally relatively low if compared to corresponding concentrations in natural watercourses (Scholz 2006a). The pH has been shown to be very low in some studies (*Thuja plicata* (red cedar), *Chamaecyparis nootkatensis* (yellow cedar), and *Populus tremuloides* Michx. (trembling aspen)), and neutral in other studies (*Picea abies* (L.) Karst (Norway spruce), and *Pinus sylvestris* L. (Scots pine)). According to studies where different tree species have been directly compared with each other, there are differences in the runoff generated. For example, emissions from storage of *Picea abies* are generally greater than those from *Pinus sylvestris* (Borgå *et al.* 1996b). In leaching experiments reported by Pease (1974), the relative COD leaching rates of four species of trees were, in descending order, *T. plicata*, *C. nootkatensis*, *Tsuga heterophylla* (hemlock), and *Picea sitchensis* (Sitka spruce).

An interesting finding by Borgå *et al.* (1996b) is that the release of pollutants from the stored wood might be affected by the chemistry of the water that comes in contact with the wood, in this case, sprinkling water. The nutrient content of the water used for sprinkling had a significant effect on the pollutants emitted from stacks of *Picea abies* and *Pinus sylvestris*. This was explained by the dependency of the growth of bacterial biomass in the log piles. Rapid growth of bacterial biomass caused by high concentrations and loads of nutrients in the sprinkling water reduced the initially high environmental loadings from the wood.

Many geoclimatic, operational, and physical factors contribute to the volume and characteristics of runoff, and a management tool to predict relative environmental risks from different sites would be of considerable value. In a study by Orban *et al.* (2002), the authors attempted to develop such a tool. A survey was devised and distributed to log yard operators in British Columbia, Canada. The survey provided information on site characteristics, volumes and types of wood processed, operational practices, incidence of runoff, runoff treatment practices, and the ultimate receiving environment. Multidiscriminant analysis was used to determine which factors were correlated to environmental risk posed by runoff. In order of importance, volume of wood stored onsite (largest contribution), frequency of runoff events, and color intensity of runoff (smallest contribution) were factors that significantly contributed to risk and were correlated positively. The methods used in this study could be applied as tools to evaluate the need for remediation at different log yards.

2.5.2.3 Effects of Runoff on Receiving Watercourses

Pollutants in runoff influence the receiving watercourse in different ways, largely depending on the geographical location, size, and chemical and biological characteristics of the watercourse. Organic material can cause oxygen depletion when it is biologically and chemically degraded, organic compounds and metals can have a toxic effect on plants and animals, nutrients in the runoff can lead to eutrophication, and particles in the runoff can have a negative impact on plant and animal life due to clouding of the water and sedimentation on substrates (Scholz 2006a; Hedmark and Scholz 2008).

A study of invertebrates in the Sapele stretch of the Benin River (Niger delta, Nigeria) found that sensitive species of genera such as *Ephemeroptera* (mayfly) or *Plecoptera* (stonefly) were absent from the location where wood waste discharge was emitted into the river (Arimoro and Osakwe 2006). The wood waste discharge altered the water chemistry and stimulated the abundance of less-sensitive macro-invertebrate species.

Dumping and rafting logs in water also leads to a potentially serious environmental impact. A study on water quality and benthic organisms at 16 marine sites in southeast Alaska (USA) showed a layer of bark covering the bottoms at all log dumping sites, and this layer was associated with a high biological oxygen demand (Pease 1974). The benthic epifauna was reduced in abundance only at the oldest active log dump studied, but the bottom fauna was reduced in all the bark-covered areas. Organic compounds leached rapidly from all species of log studied but were precipitated by the seawater. Significant effects on water quality were found at only two of the sites.

Some studies also report toxic effects, which seem to be closely related to the tree species investigated. In leaching experiments for *T. plicata*, *C. nootkatensis*, *T. heterophylla*, and *P. sitchensis* (Pease 1974), *P. sitchensis* and *T. plicata* were found to be the most toxic to *Oncorhynchus gorbuscha* (pink salmon) fry in freshwater, and *C. nootkatensis* was found to be the most toxic in seawater. All wood species were found to be more toxic in freshwater than in seawater.

Populus tremuloides logs in Canada have been reported to produce a dark, watery, and acutely toxic leachate (Taylor *et al.* 1996). Median acutely toxic concentrations of leachate were consistently between 1 and 2% of its full strength for *Oncorhynchus mykiss* (rainbow trout) and *Daphnia magna*. Inhibition of bacterial metabolism (Microtox) began at concentrations below 0.3%. The leachate was less toxic to plant life but inhibited growth at concentrations between 12 and 16%. Oxygen depletion, low pH, and phenolic compounds contribute to the toxicity of *P. tremuloides* leachate, but much of the toxic effect must be attributed to other unidentified constituents (Taylor *et al.* 1996).

Another Canadian study (Bailey *et al.* 1999) found that runoff from nine saw-mills in British Columbia was toxic to juvenile *O. mykiss*, mainly because of the content of zinc, tannins, and lignins. A third study (Taylor and Carmichael 2003) comprising tests on *O. mykiss*, *D. magna*, and luminescent bacteria (Microtox) showed that leachate from *P. tremuloides* varied from weakly toxic (median lethal concentration >10%) to very toxic (median lethal concentration <1%) levels.

A study on *T. plicata* and *C. nootkatensis* wood waste including trimmings, off-specification wood chips, shredded bark and roots, and sawdust (Tao *et al.* 2005) showed that the runoff was very toxic to aquatic life with a 96-h median lethal concentration of 0.74% leachate. According to Tao *et al.* (2007), the acute toxicity of wood leachate is usually attributed to tannins, lignins, tropolone, terpene, lignans, and low pH.

A study on log yard runoff from *Picea abies* and *Pinus sylvestris* found distillable phenols and diterpene resin acids that are potentially toxic to aquatic life (Borgå *et al.* 1996a). However, in a Swedish study (Arvidsson 2006), undiluted runoff from a sprinkled log yard at a sawmill storing the same two species was found not to have any negative effect on the reproduction of *Daphnia sp.*

It is difficult to directly compare the environmental impacts of pulp and paper mill effluents with the corresponding impacts from sawmills, log yards, and wood waste handling sites. The sizes of the sites and the corresponding scales of operation are generally greater at pulp and paper mills compared to other wood handling sites, and the use of chemicals in the processes is more intense. The impacts of the effluents on the local watercourses, however, have a lot in common because of the comparably high content of organic compounds from the wood handled at the sites. With this in mind, a few representative studies on the effects of the organic matter in effluents from pulp and paper mills on local watercourses have been included in this section.

Karrasch *et al.* (2006) investigated the impact of pulp and paper mill effluents on the water quality, microplankton system, and microbial self-purification capacity (*i.e.*, degradation of polymeric organic compounds via extracellular enzymes) of the Biobio River, Chile. They found that the impact of the effluents on the water quality was indicated by raised conductivity, increased concentrations of nitrate, nitrite, and SRP, the appearance of tannins and lignins, and a steady accumulation of inorganic and organic suspended matter along the river. Very low and declining concentrations of chlorophyll *a* and heterotrophic flagellate densities were reported. The pulp and paper mill effluents resulted in high bacterial abundances and biomass concentrations in the river water, showing that bacteria adapted to the effluent made effective use of the available nutrients and that the grazing pressure from heterotrophic flagellates was reduced (Hedmark and Scholz 2008).

Furthermore, Karrasch *et al.* (2006) showed that a pulp mill contributed to the self-purification of an affected river stretch. However, the elevated degradation capacity was not sufficient to compensate for the high load of organic material in the paper plant effluent, which, together with its toxic effects, significantly interfered with the ecological status of the Biobio River.

2.5.3 Treatment Methods

2.5.3.1 Overview of Applied Treatment Technologies and Methods

Several different technologies such as biological systems (*e.g.*, aeration lagoon, activated sludge process, and constructed wetland) and physical and chemical

systems (e.g., aeration, carbon adsorption, coagulation, ion exchange, neutralization, precipitation, reverse osmosis, chemical oxidation using ozone, calcium hypochlorite, hydrogen peroxide, and potassium permanganate, and chelation) have been proposed by Orban *et al.* (2002) and Samis *et al.* (1999) for the treatment of runoff from wood handling sites. Given the high load of organic compounds in the runoff, any method used to effectively treat this wastewater has to allow for the degradation of these compounds.

Different types of soil–plant systems and wetlands (Scholz and Lee 2005; Scholz *et al.* 2007) might also be used to treat log yard runoff with high concentrations of phosphorus. When the elevated level of phosphorus in the runoff is a problem, an efficient purification method must reduce the concentration of phosphorus in the water before it reaches the receiving watercourse. The methods that have been used in laboratories or field-scale trials have focused mainly on the degradation of the organic matter in the runoff and the accompanying reduction of toxicity.

2.5.3.2 Soil Infiltration

The reduction of pollutants that is achieved by soil infiltration of runoff is due to physical, chemical, and biological processes taking place in the soil–plant water ecosystems (Wang *et al.* 1999). A soil infiltration system example is shown in Figure 2.24. The degree of degradation, assimilation, sorption, exchange, and neutralization during the infiltration also depends on the velocity of the solution through the soil. Biological degradation, mineralization, and adsorption are the



Figure 2.24 Experimental soil infiltration system treating log yard runoff near Heby, Sweden

major mechanisms contributing to the reduction of organic substances in the soil column. If contaminants are predominantly retained through adsorption alone, then a breakthrough of high concentrations of organics in the leachate can occur when the soil adsorption capacities are exceeded (Wang *et al.* 1999).

An early study by Peek and Liese (1974) in Lower Saxony (Germany) concluded that the purifying capacity of the soil in the area where windthrown timber was stored and sprinkled for a year was sufficient to prevent a decline of water quality in an adjoining well or stream. More recent studies have shown that infiltration of log yard runoff in unlined wetlands planted with different plant species have been successfully used for a sawmill in central Sweden (Hedmark and Jons-son 2008), even at irrigation intensities of up to 66 mm/d. The purification efficiency of the infiltration wetland was continuously high for total organic carbon, total phosphorus, and phenols, and the purification capacity was clearly maintained after 4 years of irrigation. During the fourth year of operation, however, the concentrations of total phosphorus in the groundwater below the irrigated area increased significantly compared to previous years. This may indicate that the soil is close to the saturation point for total phosphorus (Casson *et al.* 2006).

Other reports by Jonsson *et al.* (2004, 2006) based on soil infiltration experiments in field-scale studies utilizing lysimeters have shown good purification results. There was no difference in purification efficiency for total organic carbon, distillable phenols, total phosphorus, and total nitrogen when comparing treatments by *Elytrigia repens* (L.) (couch grass), *Salix sp.* (willow), and *Alnus glutinosa* (L.) Gärtner (alder) with each other. No clear differences were found between lysimeters containing sand or clay. Lysimeters with high levels of irrigation showed greater retention than those with low levels.

2.5.3.3 Wetland Treatment

Constructed wetlands are recognized as low cost and ‘minimal’ maintenance systems that could lower the impact of wastewater or storm water discharges on natural water bodies (Scholz 2006a; Scholz and Lee 2005). Wetlands, ponds, lagoons, and trenches (Figure 2.25) are sometimes used at sawmills and log yards to enhance treatment of runoff including sedimentation of particles. For example, a vegetated trench is used for treatment of log yard runoff from a sawmill in Boxholm, Sweden. This wetland performs well in terms of organic matter removal.

In a study by Doig *et al.* (2006), the composition of the organic matter in the runoff from a log sort yard was examined before and after treatment in a lagoon. The fresh runoff had chemical oxygen demand concentrations ranging between 346 and 3690 mg/l. In a rainfall-generated sample, particulates (*i.e.*, particles greater than approx. 1.5 μm) contributed up to 52% of the total COD, and colloids (particles between 0.02 μm and approx. 1.5 μm) were associated with only 39% of the corresponding COD. Following primary treatment in a lagoon, the organic particles found were mostly of a colloidal nature. Primary treatment experiments showed that between 27 and 54% of the COD could be removed by settling, de-

pending on the initial concentration. An additional 33% of COD was removed, probably due to biological degradation during settling. The COD of the ultimately treated effluents remained relatively high (378 to 533 mg/l), indicating that not all suspended material could be removed through settling and biodegradation alone, and that the application of further treatment methods would be required.

The use of vegetated wetlands usually leads to better purification results than lagoons or ponds (Scholz 2006a). There are various reasons for the better efficiency. For example, macrophytes act as mechanical filters and provide substrate for microorganisms, which take part in degradation and absorption processes of pollutants. Furthermore, aquatic plants aerate water and the root zone, and reduce the flow speed of the water, giving particles more time to settle. Sediment adsorption is likely to affect the treatment performance in surface-flow constructed wetlands predominantly during the initial and transitional operating periods. However, the major mechanism for the removal of organic carbon from wood waste leachate during long-term operation is likely to be biological degradation (Tao *et al.* 2007).

Tao *et al.* (2006) studied the treatment performances of four vegetated surface-flow mesocosm wetlands fed with different dilutions of wood waste leachate over a period of 12 weeks. The wood waste leachate was diluted before discharge to constructed wetlands to prevent potentially adverse impacts of the raw leachate on heterotrophic bacteria and aquatic plants. During a subsequent period of 13 weeks, the effluent of a vegetated wetland fed with the raw leachate was further treated in



Figure 2.25 Wetland trench treating log yard runoff near Boxholm, Sweden

both a vegetated wetland and an open wetland. The highest reduction rates for COD as well as tannins and lignins were achieved in the wetland fed with the raw leachate. The most diluted (up to six times) wood waste leachate yielded the highest reduction efficiencies. Up to 47 mg/l volatile fatty acids in the influent were depleted through wetlands with a hydraulic retention time of 13 d. In this study, the vegetation made an insignificant difference in terms of the treatment performance for wood waste leachate. The COD as well as tannins and lignins were further removed through the wetlands operating in series, though at lower reduction rates. The performance of wetlands for the treatment of wood waste leachate was partly regulated by the dissolved oxygen concentration and the availability of bacterial substrates.

In another field experiment, Tao *et al.* (2006) studied how different hydraulic retention times affected the purification efficiency of four surface-flow mesocosm wetlands. The wetlands were fed with diluted wood waste leachate, which was acidic, of very high oxygen demand, and toxic. Mass reduction efficiencies of COD, tannins, and lignins increased significantly with hydraulic retention time when 10% leachate was diluted with tap water. When a more recalcitrant influent was fed into the wetlands, there was a slight increase of reduction efficiency with increasing hydraulic retention time. Precipitation and evapotranspiration had profound impacts on the overall performance and its variability.

Another study on the effectiveness of constructed wetlands for the treatment of wood waste leachate was conducted by Masbough *et al.* (2005) on a wood waste storage site, adjacent to the Lower Fraser River (near Mission, British Columbia, Canada). The leachate was characterized by high oxygen demand, tannins and lignins, and volatile fatty acid concentrations, but by low pH and nutrient values. Diluted leachate passed through six pilot-scale wetland cells, four planted with *Typha latifolia* L. (reedmace) and two unplanted controls, with a mean depth of 0.4 m and a hydraulic retention time of 7 d. Nutrient addition and pH adjustments were made to improve contaminant removal. Reductions in contaminants were consistently achieved, with mean removals for BOD, COD, volatile fatty acid, and tannins and lignins of 60, 50, 69, and 42%, respectively. Furthermore, aging of the constructed wetland system increased the treatment performance.

2.5.3.4 Other Treatment Methods

Where it is not possible to construct wetlands or use infiltration as a treatment method, for example due to limited land availability, more high-tech methods can be used to reduce the impact of contaminated runoff from wood handling sites. There are only a limited number of studies so far concerning runoff from log yards and sawmills. More studies have been conducted on wastewater from pulp and paper mills, and those are discussed in Section 2.5.3.5.

In a study by Woodhouse and Duff (2004) on runoff from a sawmill on Vancouver Island (British Columbia, Canada), five runoff samples were treated in a laboratory-scale attached microbial growth reactor. Six samples were acutely toxic

($EC_{50} < 100\%$) based on the Microtox assay. The samples were effectively treated in the attached microbial growth reactor. Treatment for 24 h at 34°C resulted in substantial reductions of BOD, COD, and tannins and lignins. Near complete removal of acute toxicity and color were also observed. Twenty-four-hour treatment at lower temperatures (24°C and 5°C) also substantially reduced the concentrations of organic matter and the toxicity.

In another Canadian study, samples of log yard runoff were obtained from two British Columbia coastal sawmills (Zenaitis and Duff 2002). Centrifuged samples were treated with ozone doses up to approx. 0.5 mg ozone/mg COD in a lab-scale reactor. Ozonation was found to significantly reduce toxicity, tannin and lignin, and dehydroabietic acid concentrations. However, there were only moderate reductions in COD and BOD. At slightly acidic to neutral conditions, pH had no effect on the COD. Tannin and lignin, and toxicity removal were slightly improved in neutral solutions compared to acidic ones, while dehydroabietic acid removal significantly improved.

In another study on Vancouver Island (British Columbia, Canada), ozone treatment of log yard runoff was combined with biological treatment (Zenaitis *et al.* 2002). Batch biological treatment of log yard runoff reduced the BOD, COD, and tannin and lignin concentrations considerably. Acute toxicity decreased from an initial EC_{50} of 1.83% to a value of 50.4% after 48 h of treatment.

The efficiency of ozone as a pre- and postbiological treatment stage was also assessed by Zenaitis *et al.* (2002). During ozone pretreatment, the combined tannin and lignin concentration and the acute toxicity were rapidly reduced. Preozonation had a minor reduction effect on the BOD and COD. However, a larger fraction of residual COD was non-biodegradable after ozonation.

Moreover, biologically treated effluent was subjected to ozonation to determine whether further improvements in effluent quality could be achieved. Reductions in COD and in tannin and lignin concentrations were observed during ozonation. However, no further improvement in toxicity was observed. Ozonation increased the BOD due to chemical conversion (Zenaitis *et al.* 2002).

2.5.3.5 Treatments Used for Organic Matter in Pulp and Paper Mill Wastewater

In this section, the focus will be on techniques that are applicable for the treatment of organic matter contained within log yard runoff and wood waste leachate as well as within effluents from pulp and paper mills. The effluent characteristics of pulp and paper mills can differ considerably from those of the runoff from wood handling sites because of the chemicals used in the production processes. However, a relatively large proportion of organic material in the wastewater originates from the wood handled at the sites (Wang *et al.* 1999).

The wastewaters from mechanical pulping and secondary fiber pulping and the condensates from chemical pulping are non-toxic to methanogenic degradation and contain easily degradable organic compounds. Anaerobic digestion is therefore an

attractive treatment for these effluents (Rintala and Puhakka 1994). This technology, often in combination with aeration units, is now well established in the industry.

Munoz *et al.* (2006) applied different advanced oxidation processes to remove the organic carbon content of a paper mill effluent originating from a pulp bleaching process. The oxidation processes comprised titanium-dioxide-mediated heterogeneous photocatalysis, titanium-dioxide-mediated heterogeneous photocatalysis assisted with hydrogen peroxide, titanium-dioxide-mediated heterogeneous photocatalysis coupled with Fenton, photo-Fenton, ozonation, and ozonation with ultraviolet-A light irradiation. The results showed that all selected oxidation processes considerably decreased the dissolved organic carbon content.

Moreover, a life cycle assessment indicated that heterogeneous photocatalysis coupled with the Fenton's reagent had the lowest environmental impact and a moderate to high dissolved organic carbon removal rate. On the other hand, heterogeneous photocatalysis appeared to be the worst oxidation process both in terms of dissolved organic carbon abatement rate and corresponding environmental impact. The environmental impact of the studied advanced oxidation processes is attributable to the high electrical energy consumption necessary to run a UV-A lamp or to produce ozone (Munoz *et al.* 2006).

Nair *et al.* (2007) studied the effect of phenol degrading *Alcaligenes* spp. on paper factory effluent. Three techniques were applied: use of free cells, immobilized cells, and a continuously operated packed bed reactor. An almost complete reduction of phenol was found for all techniques. When comparing them with each other, the reduction capacity for COD was the best with the continuously operated packed bed reactor, even at a much shorter treatment time. After the complete removal of phenol, the strain could still further enhance the reduction of the COD, which clearly indicated that this strain could also oxidize organic matter other than phenol in the paper factory effluent.

Uğurlu *et al.* (2006) studied the effectiveness of electrochemical treatment for the high concentration of organic pollutants in paper mill effluents. They used a working electrode cell consisting of a graphite electrode and powder-based activated carbon. Electrolysis time, voltage, initial pH, activated carbon, sodium chloride concentration, and air flow were selected as parameters. The removal efficiency significantly depended on the applied cell voltage, air flow, time, salt concentration, and pH. COD and lignin and phenol removal efficiencies from the paper mill effluent were just below 90% after electrolysis.

Ozonation has been evaluated for the treatment of both log yard runoff and pulp mill effluents. In a study by El-Din *et al.* (2006b), ozonation of resin and fatty acids, which were found in pulp mill effluents, was assessed using rapid-scan stopped-flow spectrophotometry. The resin and fatty acid oxidation (*i.e.*, degradation) efficiency increased with an increase of ozone and temperature. Microtox bioassay tests showed that there was an increase in toxicity for resin acid as a result of ozonation. The toxicity of fatty acid samples, however, decreased as a result of ozonation.

In a further study by El-Din *et al.* (2006a), ozonation of abietic and linoleic acids (resin and fatty acids) was assessed at various pH levels and low temperatures

(0 to 10°C) using a rapid-scan stopped-flow spectrophotometer. Degradation of abietic and linoleic acids could be enhanced by manipulating reaction conditions including ozone dose, temperature, and pH. Ozonation of abietic acid led to a 140% increase in Microtox toxicity, whereas it was reduced to half in the case of linoleic acid.

Wetlands have also been investigated for the treatment of wastewater from pulp and paper mills. The capacity of a pilot project sub-surface batch flow constructed wetland in the tropics to remove phenol from pretreated pulp and paper mill wastewater was studied under varying hydraulic retention times (Abira *et al.* 2005). Initial results based on a 15-month study indicated that mean removal efficiencies for phenol were variable but reached 60 and 77% at 5-d and 3-d hydraulic retention times, respectively. It was thought that the longer retention time might have caused oxygen and nutrient deficiencies, which may have reduced the removal performance. Although phenol was sometimes not detectable in the wetland outflow, mean values over the experimental period did not meet national guidelines.

Irrigation and infiltration of pulp mill effluents have also been investigated. For example, large barrel lysimeters containing intact soil cores were irrigated with thermomechanical pulp effluent in New Zealand for 16 months (Wang *et al.* 1999). The thermomechanical pulp mill used *Pinus radiata* D. Don (radiata pine) as its main wood source. The primary treated effluent was applied on the lysimeters at a loading rate of 30 mm/week. Analysis of the effluent showed low concentrations of nutrients and inorganic contaminants, but high concentrations of wood-derived organic contaminants such as total organic carbon, COD, BOD, and wood extractives. Removal rates through the soil core were greater than 90% for total organic carbon, COD, BOD, turbidity, resin acids, phytosterols, and phenols. No excessive nutrient and heavy metal leaching was observed during the experimental period.

2.5.4 Discussion, Conclusions, and Further Research

2.5.4.1 Discussion Concerning the Cost-effectiveness of the Treatment Methods

The literature review presented here has highlighted that runoff from wood handling sites can cause negative environmental effects in receiving watercourses. However, it is common in industry that no treatment of wastewater is done prior to its release. As knowledge regarding the content and the environmental impact of the runoff increases, environmental authorities are demanding more frequently that the pollution loads from the sites be reduced. Low-cost biotechnologies such as constructed wetlands for the treatment of runoff have gained considerably in popularity (Scholz 2006a).

The review presented here indicates that for sawmills and log yards, it is important to select a treatment method with reasonable costs for installation and maintenance. Treatment with constructed wetlands or infiltration in soil and plant systems has been shown to be effective for reducing the concentrations of organic matter and phosphorus and for reducing the toxicity of the runoff from wood handling sites. These methods do not require high technical maintenance, and the amount of labor necessary to keep them operating is low. The cost of construction, however, depends on the soil properties and the cost of land at each site.

Constructed wetlands are less costly than conventional techniques used for the treatment of wastewater and are a viable option for wood handling operators. At sites with a shortage of land for constructed wetlands or soil infiltration, it is likely to be necessary to invest in treatment methods that are more costly, energy demanding, or high-maintenance such as ozonation.

The processes used to treat the organic matter in pulp and paper mill effluents are also applicable to treat runoff from wood storage sites. However, the cost of the treatment may be an important limiting factor. Based on the author's experiences, sawmills and other wood handling sites usually have a smaller turnover than pulp and paper mills and therefore do not have the same means to invest in high-tech water treatment equipment.

Based on the author's professional experience, ozonation is generally associated with very good purification efficiencies for organic matter and can also reduce the accompanying toxicity of these compounds. Ozonation of some compounds such as resin acid can lead to increased toxicity. However, this depends on the formation of degradation products of higher toxicity than the original compounds.

The degree of purification that needs to be achieved largely depends on the condition of the receiving watercourse and demands from the environmental authorities and the public. It follows that different treatment methods alone or in combination with each other (treatment train) might be fit to a given purpose (Scholz 2006a).

Constructed wetland biotechnology could be enhanced by taking advantage of recent advances in microbiology. For example, Doradoa *et al.* (2000) indicated the potential of wood pretreatment with selected sapstain fungi for decreasing effluent toxicity in pulping (Doradoa *et al.* 2000). The combination of this pretreatment method with constructed wetlands is likely to lead to a reduction in required wetland area, making this biotechnology more attractive for areas where land is costly.

2.5.4.2 Summary of Conclusions

This section addresses the timely and urgent need of the wood and sawmill industries to assess various soft technologies including wetland systems to treat and attenuate log yard runoff. Moreover, the pollution potential of runoff from

wood handling sites has been assessed. Organic matter and phosphate are the key contaminants.

The reasons for pollution generation at wood handling sites have been highlighted, and the runoff and its effect on the receiving watercourses have been assessed. Soft treatment biotechnologies including wetland and soil infiltration have been characterized and assessed. Various wetland systems including a novel combination of a size-optimized wetland with a runoff conveyance channel (case study in Boxholm, Sweden) have been suggested for future application. The proposed system reduces phosphates and organic matter to concentrations below common wastewater treatment guidelines. The key advantage of most relevant wetland and soil infiltration systems is that they require little energy and low maintenance.

Ozonation has been shown to often effectively decrease acute toxicity, but to have little effect on the COD and BOD. Ozonation of some organic compounds (e.g., resin acid,) can also lead to increased toxicity. Therefore, this high energy-consuming advanced oxidation technology is often unsuitable for the treatment of log yard runoff.

Finally, various treatment methods used for the removal of organic matter in pulp and paper mill wastewater have been assessed. Based on this review, wetland systems similar to those treating log yard runoff are recommended for the industry.

2.5.4.3 Further Recommended Research

The environmental impact from each separate wood handling site is relatively small, but cumulatively, the overall environmental impact from the storage and handling of wood is likely to be of high significance in terms of the pollution load. This hypothesis needs to be tested in future research studies.

It is also common that irrigation water for a log yard is abstracted from a watercourse close to the yard, and in small watercourses overabstraction can lead to drying out or a significant reduction of the water level, which may have negative effects on both the biology and the chemistry of the watercourse. However, no research on this aspect of the wood handling industry's environmental impact can be found in the public domain.

This section supports the reader in choosing a sustainable and cost-effective treatment technique for runoff from wood handling sites. However, no ultimate method that gives sufficient long-term removal of phosphorus was found. Further studies on sustainable treatment methods that can effectively remove phosphorus from runoff are therefore strongly recommended.

Further research should be undertaken on the 'real' whole life costs and tangible benefits of wetland treatment systems compared with traditional methods such as the activated sludge process and percolation filtration. Moreover, the modeling of wetland performance for highly dynamic (*i.e.*, seasonal operation and fluctuating stock capacities) systems such as log yards remains a challenge.

2.6 Wetlands for Treating Hydrocarbons

2.6.1 Introduction

2.6.1.1 Constructed Treatment Wetlands

Wetlands are complex and integrated systems in which water, animals, plants, microorganisms, and the environment interact to improve the water quality (Guirguis 2004). Man has started to mimic nature by building wetlands to treat a variety of waters, wastewaters, storm waters, gully pot liquor, acid-mine drainage waters, landfill leachate, irrigation waters, agricultural wastewater, runoff waters, industrial wastewater, and produced waters (Moshiri 2000; Rew and Mulamoottil 1999; Scholz 2004; Vrhovsek *et al.* 1996; Yang *et al.* 2001). However, very little is known about the processes involved and the use of constructed treatment wetlands in the removal of petroleum hydrocarbons from processed water. This has limited the effective application of this technology in the oil and gas industries.

Vertical-flow and horizontal-flow constructed reed beds based on soil, sand, or gravel are used frequently to treat domestic and industrial wastewater (Kadlec and Knight 1996). Constructed wetlands for wastewater treatment have several advantages such as low operational costs and habitat enhancement over conventional treatment methods (Cooper *et al.* 1996; Moshiri 2000). The wetland treatment performance depends on the interactions between many different physical and biochemical components. It follows that some functions such as the role of aggregates (also called substrate or wetland media) within constructed wetlands are not completely understood (Scholz 2006a).

Vertical-flow wetland systems have a structural makeup of several layers of aggregates. Water is forced to flow perpendicular to the length of the wetland. Wetlands consist of different sizes of gravel and sand and are often planted with macrophytes, which provide oxygen to the rhizosphere, thereby creating an aerobic environment (Scholz 2006a). This in turn supports microbial communities, which can either directly biodegrade organics or catalyze chemical reactions and subsequently support biotransformation processes. Bacteria capable of degrading volatile organics such as benzene, toluene, ethylbenzene, and p-xylene have been found in the rhizosphere (Hiegel 2004; Sugai *et al.* 1997). Numerous benzene-degrading aerobic microorganisms have been identified; the most notable are the *Pseudomonas* spp., which may account for up to 87% of the gasoline-degrading microorganisms in contaminated aquifers (Ridgeway *et al.* 1990). Petroleum wastes have been documented to degrade in natural wetland environments (Wallace and Knight 2006; Wemple and Hendricks 2000). Benzene is biodegradable, particularly in the presence of oxygen (Wemple and Hendricks 2000). Benzene degradation has also been demonstrated in the presence of nitrate–nitrogen (Burland and Edwards 1999). Many studies have shown that microbial degradation of petroleum hydrocarbons in the environment is strongly influenced by physical and chemical factors such as temperature, oxygen, nutrients, salinity, pressure, water

activity, and pH and the chemical composition, physical state, and concentration of the contaminant, as well as by biological factors such as the composition and adaptability of the microbial population (Burland and Edwards 1999).

Kadlec (2001) states that aeration is an important component of sub-surface-flow wetland design because an active aeration system enhances both volatilization and aerobic degradation of hydrocarbons. The transfer of oxygen to contaminated aquifers to stimulate aerobic degradation is a common bioremediation practice (Lovley and Lloyd 2000). The biodegradability of the most water-soluble components of gasoline such as benzene (Paje *et al.* 1997; Solano-Serena *et al.* 1999), toluene (Leahy and Olsen 1997), ethylbenzene (Di Lecce *et al.* 1997), and xylene (Di Lecce *et al.* 1997) compounds has been clearly established.

Because biodegradation and evaporation processes compete in removing petroleum hydrocarbons, biodegradative losses cannot be differentiated clearly from volatility losses (Zhou and Crawford 1995). Mitsch and Gosselink (1993) revealed that freshwater wetlands are typically considered to be nutrient-limited due to the heavy demand for nutrients by the plants, and they could also be nutrient traps, as a substantial amount of nutrients may be trapped in biomass. Hence the addition of nutrients is necessary to enhance the biodegradation of oil pollutants (Choi *et al.* 2000; Kim *et al.* 2005). However, studies in the past (Chaillan *et al.* 2006) have shown that excessive nutrient concentrations can inhibit biodegradation activity, and several authors have reported the negative effect of high nitrogen, phosphorus, and potassium levels on the biodegradation of hydrocarbons (Chaineau *et al.* 2005; Oudot *et al.* 1998), and particularly on the aromatics (Carmichael and Pfander 1997).

2.6.1.2 Benzene Removal

There is a gap in knowledge concerning the optimization of those process conditions that would foster a more efficient application of the constructed wetland technology particularly for the removal of hydrocarbons under different climatic and other variable environmental conditions. However, the number of research projects concerning constructed wetlands for hydrocarbon treatment has considerably increased (Baris *et al.* 2001; Eke and Scholz 2006; Myers and Jackson 2001; Omari 2003; Xia 2003).

Since 1995, journal articles and symposia proceedings indicate the petroleum industry's interest in using constructed wetlands to manage process wastewater and storm water at a variety of installations including refineries, oil and gas wells, and pumping stations (Knight 1999). The area of emphasis in this research is the use of constructed wetlands for the treatment of dissolved petroleum hydrocarbon compounds, hence the need to have an understanding of what constitutes petroleum and its degradation products. There are four basic petroleum compounds (Harayama *et al.* 2004):

- saturated hydrocarbons, the primary component;
- aromatic hydrocarbons (at least one aromatic ring within the molecule);

- resins, which are mostly unknown; and
- asphaltenes.

These compounds are relatively large molecules and tend to possess small amounts of nitrogen, sulfur, or oxygen (Harayama *et al.* 2004). Untreated petroleum industry wastewaters contain phenolics (Knight 1999), sulfides, and various trace metals (Knight 1999). Hydrocarbons are highly soluble and neurotoxic and cause cancer (Hiegel 2004). Monoaromatic for benzene, toluene, ethylbenzene, and xylenes (BTEX), hydrocarbons (*i.e.*, benzene, toluene, ethylbenzene, and xylene) are commonly found in gasoline and are highly volatile substances (Coates *et al.* 2002). Due to their relatively high solubility and toxicity, they represent a significant health risk in contaminated environments. Of all the BTEX compounds, benzene poses the gravest concern because it is the most toxic and a well-known human carcinogen. The benzene ring (Figure 2.26) is a chemical structure that is common in nature. Moreover, the thermodynamic stability of the benzene ring increases its persistence in the environment; therefore, many aromatic compounds are major environmental pollutants (Dagley 1986; Diaz Eduardo 2004).

Their major industrial source is petroleum and natural gas, formed geochemically from biomass under high pressure and temperature (Heider *et al.* 1998). Aromatic hydrocarbons are one of the most abundant class of organic compounds and constituents of petroleum and its refined products. Monocyclic aromatic hydrocarbons are of major concern, because of their toxicity, high solubility, and ability to migrate within groundwater. These BTEX compounds are of primary discharge concern for the water quality of receiving waters (Caswell *et al.* 1992). The BTEX fraction of total volatile hydrocarbons is primarily responsible for most of the total toxicity in gasoline-contaminated groundwater. Hence, an attempt to reduce toxicity requires targeting these compounds for destruction. Many components of hydrocarbon mixtures are toxic and relatively soluble in water. In natural gas, benzene concentrations typically range from about 0 to 1,000 mg/l; in crude oils from virtually zero to 10,000 mg/l (Janks and Cadena 1991). Benzene has relatively high water solubility (1,780 mg/l). Water contamination by oil exploration and production operations, tank farms, underground storage tank leakage, and refineries has become a concern to the oil and gas industry.

Hydrocarbon degradation is less dependent on the actual reactions taking place than it is on the processes occurring in the surrounding ecosystem (Sugai 1997). Aerobic biodegradation and volatilization constitute a coupled pathway that contributes significantly to the natural attenuation of hydrocarbons (Lahvis *et al.* 1999).

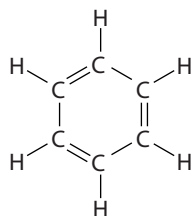


Figure 2.26 Chemical structure of benzene

Achieving high treatment performances within a short time is critical, so the design can be amended to allow manipulations of environmental conditions to enhance dissolved hydrocarbon treatment. Environmental conditions to be taken into consideration include dissolved oxygen (DO), pH, temperature, and nutrient requirements (*i.e.*, nitrogen and phosphorus) of the wetland plants and microbes. Associated contamination is therefore a major environmental problem due to the manufacture, transportation, and distribution of petroleum (Atlas and Cerniglia 1995).

2.6.1.3 Novelty, Aim, and Objectives

The research uses data gathered from experimental small-scale wetlands to assess the efficiency of benzene removal in each wetland and to compare different operational conditions. The research covers the assessment of environmental, physical, and microbial processes. This enhances operational knowledge and understanding of treatment wetlands and provides data that could be used to design full-scale wetland systems for efficient hydrocarbon treatment, and to model biodegradation and operational processes. Improved system control should include knowledge and understanding concerning environmental requirements such as oxygen availability, water inundation duration and temperature variability, fertilizer requirements for wetland microbes, and characterization of microbes capable of degrading petroleum hydrocarbons.

The overall aim is therefore to advance understanding of the application of constructed treatment wetlands for benzene removal. The objectives are to assess:

- the current literature on benzene removal with constructed wetland systems;
- investigation of the main benzene removal pathways;
- variables and boundary conditions impacting operation and treatment performance (*e.g.*, temperature level and variability, macrophytes, and aggregates);
- the efficiency of different wetland set-ups in removing benzene, COD, BOD, and nutrients;
- the effect of nutrient concentration increases on benzene removal within wetlands; and
- the impact of seasonal change and environmental control on the treatment efficiency of benzene and other water quality variables such as COD, BOD, DO, redox potential, turbidity, and nutrients.

2.6.2 Materials and Methods

2.6.2.1 Experimental System Design and Operation

Two experimental small-scale constructed wetland rigs with six wetlands each are located on The King's Buildings campus in Edinburgh, UK (Table 2.1; Fig-

ures 2.27 and 2.28). The systems have been in operation since April 2005. The wetlands were designed to simulate physical, chemical, and microbiological processes occurring in full-scale semi-natural wetlands. One experimental rig was situated outdoors to assess seasonal changes, while the other system was placed indoors to allow better control over the environmental changes. The word ‘control’ indicates in this case that the data variability of most environmental variables was relatively low due to the relative absence of seasonal impacts.

Since summer 2006, a temperature and humidity control unit was in operation to reduce the variability of temperature even further. The indoor rig was located below three plant grow lights (Sylvania 15000 Hour, 36 W, 1200 mm, T8 GroLux Fluorescent Tube; supplied by Lyco Direct Limited (Bletchely, Milton Keynes, England), product code number: EV1768) simulating day and night conditions.

The wetland rigs were designed to optimize the chemical, physical, and microbiological processes naturally occurring within wetlands. Phenomena studied include biomass manipulations, loading rate variations, and changes to the cycle of filling and emptying of the wetlands.

Round grey polyvinyl chloride drainage pipes (height: 75 cm; diameter: 10 cm), which are resistant to hydrocarbons, were used to construct vertical-flow wetlands. The outlet valves are located at the center of the bottom plate of each wetland and are used for the regulation of flow and sampling. Passive aeration is encouraged with 1.3-cm-diameter ventilation pipes reaching down to 10 cm above the bottom of each wetland. The wetlands are very small in comparison to large-scale systems used in industry, but previous findings based on similar column experiments have been fully accepted by the scientific community (Eke and Scholz 2006; Hiegel 2004; Scholz 2004, 2006a; Omari 2003; Wallace and Knight 2006).

Table 2.1 Packing order of the experimental constructed wetland set-up for inside and outside wetlands. All wetlands were alternatingly inundated and subsequently fully drained two times per week

Height (cm)	Wetland 1	Wetland 2	Wetland 3	Wetland 4	Wetland 5	Wetland 6
61–75 (top)	W+B+F	W+F	W+B+F	W+F	W+B+F	W+F
56–60	5+P+W+B+F	5+P+W+F	5+W+B+F	5+W+F	W+B+F	W+F
51–55	5+P+W+B+F	5+P+W+F	5+W+B+F	5+W+F	W+B+F	W+F
36–50	4+P+W+B+F	4+P+W+F	4+W+B+F	4+W+F	W+B+F	W+F
26–35	3+W+B+F	3+W+F	3+W+B+F	3+W+F	W+B+F	W+F
11–25	2+W+B+F	2+W+F	2+W+B+F	2+W+F	W+B+F	W+F
0–10 (bottom)	1+W+B+F	1+W+F	1+W+B+F	1+W+F	W+B+F	W+F

W: water; B: benzene; F: fertilizer (8 g of N-P-K Miracle-Gro fertilizer were added to all wetlands every 2 weeks until 29 May 2006 when the concentration was increased to 30 g. From 26 June 2006 onwards, the concentration was lowered to 15 g every 2 weeks.); P: *Phragmites australis* (Cav.) Trin. ex Steud. (nine plants of roughly equal biomass and strength per wetland); 1: stones (37.5–75 mm); 2: large gravel (10–20 mm); 3: medium gravel (5–10 mm); 4: small gravel (1.2–5 mm); 5: sand (0.6–1.2 mm).

The wetland systems have different wetland bed volumes depending on the composition of layers of aggregates such as stones, gravel, and sand. Selected wetlands were planted with *Phragmites australis* (Cav.) Trin. ex Steud. (Table 2.1). Nine plants of roughly equal biomass and strength obtained from a local supplier (Alba Trees Public, Lower Winton, Gladsmuir, East Lothian, Scotland) were planted in each ‘planted wetland’.

Wetlands 5 and 6 are strictly speaking rather pond systems (extended storage) considering that they do not contain any aggregates. Moreover, wetland 6, con-

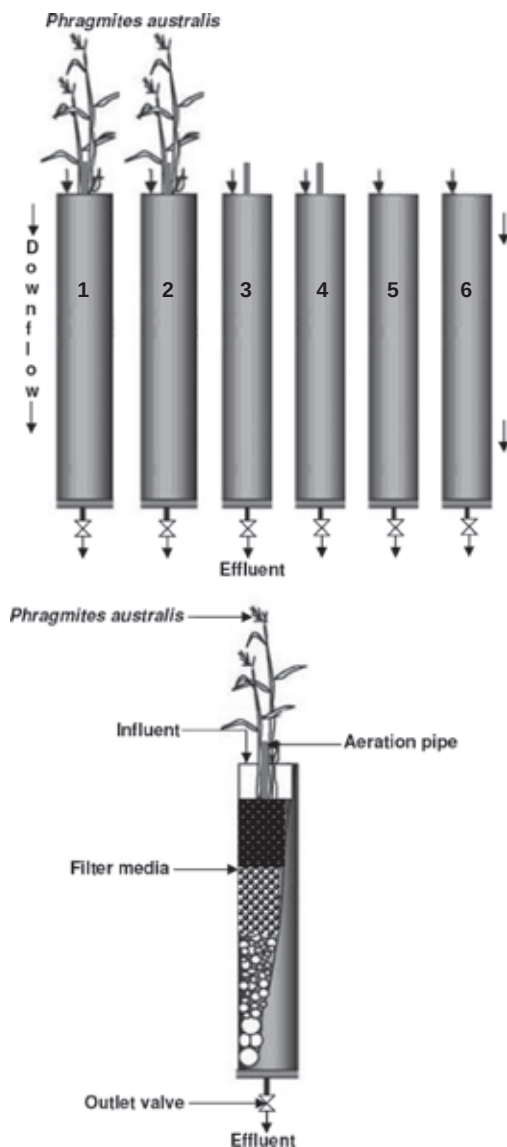


Figure 2.27 Schematic representation showing wetland set-up and internal structure of experimental constructed treatment wetland cell 1

taining only water and fertilizer, can be considered a 'blank.' Wetlands 1, 3, and 5 of both rigs received tap water mixed with a concentration of 1 g/l benzene two times per week. Benzene was used as an example volatile hydrocarbon to assess the removal of low-molecular-weight petroleum compounds. Benzene (BDH analytical reagent, C_6H_6 (99.7%)), supplied by VWR International Ltd (Hunter Boulevard, Lutterworth, UK) was used. Benzene was chosen for various reasons:

- It is a common constituent of liquid fuels.
- It can be used as a surrogate for a mixture of hydrocarbons to allow for easy interpretation of the data and subsequent modeling.
- The traditional treatment technologies used by the oil industry such as hydrocyclones and separators predominantly remove heavy hydrocarbons but not aromatic components in the dissolved water phase.
- The thermodynamic stability of the benzene ring increases its persistence in the environment; therefore, many aromatic compounds are major environmental pollutants.

While hydrocarbons dissolved in water are a suitable source of carbon and energy for microbes, they are an 'incomplete food source' since they do not contain nutrients such as nitrogen, phosphorus, and potassium required for microbial growth. In order to stimulate the growth of microbiological biomass and *P. australis*, 8 g of the well-balanced nitrogen–phosphorus–potassium Miracle-Gro (formerly Osmocote, produced by Scot Europe B. V., The Netherlands) fertilizer were added to all wetlands every 2 weeks until 29 May 2006, when the concentra-



Figure 2.28 Indoor experimental wetland filters treating benzene

tion was increased to 30 g to assess the effect of nutrient concentration increases on benzene removal. From 26 June 2006 onwards, the concentration was lowered to 15 g every 2 weeks, which was seen as a more realistic fertilizer concentration to reduce the fertilizer budget and to avoid too high outflow concentrations, which would have been detrimental to the environment (Table 2.1). The system was designed to operate in batch flow mode to reduce pumping costs.

All wetlands were alternately inundated and subsequently fully drained two times per week to encourage rapid air penetration through the soil (Table 2.1) (Cooper *et al.* 1996). During draining, the water flowed rapidly out of the system, and air was subsequently drawn deeply into the lower part of the wetland. This aerated the system and speeded up the rate of hydrocarbon degradation during the 2 or 3 d when the wetlands were not inundated.

Water samples were tested twice per week for COD, pH, DO, turbidity, redox potential, conductivity, ammonia–nitrogen, nitrate–nitrogen, ortho-phosphate–phosphorus, and temperature. The BOD was analyzed in all water samples using a respirometric method with the help of the OxiTOP IS 12-6 system, supplied by the Wissenschaftlich-Technische Werkstätten (WTW), Weilheim, Germany. This equipment uses the principle of piezoresistive measure of pressure differences. Samples were analyzed for benzene removal once per month until January 2007. Samples were tested twice per month afterwards. Benzene content was determined with the Perkin Elmer GC-FID and headspace sampler (models 9700 and HS-101 respectively) instrument.

All water quality variables were determined according to the American Standard Methods for the Examination of Water and Wastewater (1998) unless stated otherwise. Statistical differences were assessed by ANOVA and Tukey's Honestly Significantly Different test ($p < 0.05$), which is based on the "studentized range distribution" (Rice Virtual Lab, Lane D; <http://davidmlane.com/hyperstat/B95118.html>).

2.6.2.2 Biodegradation and Volatilization Determination

Biodegradation and volatilization were also tested in separate experiments. Two extra wetlands (heights: 24 cm; diameters: 5 cm) were set up under controlled environmental conditions; one wetland comprised aggregates and detritus containing mature microbial biomass (284 g detritus was taken from the upper layer of the contaminated parent wetland 3 located indoors) and the other wetland was left empty. The small wetlands were constructed in the same way as the large wetlands except for the absence of ventilation pipes. The purpose of this auxiliary experiment was to assess the main removal pathways of benzene (combined biodegradation and adsorption versus volatilization) in constructed treatment wetlands. Samples were taken after 1, 2, 3, 6, and 9 d, and benzene content was subsequently determined using headspace and gas chromatography.

2.6.3 Results and Discussion

2.6.3.1 Treatment Performance Comparisons

Considering the benzene inflow concentration of approx. 1.3 g/l, the findings indicate high percentage mean removal efficiencies (85 to 95%) for wetlands containing *P. australis* and aggregates (wetlands 1 and 3 of the indoor and outdoor rig; Table 2.1) and are comparable to data published elsewhere (Myers and Jackson 2001). The findings indicate a high variability of the benzene outflow concentrations. In contrast, wetlands without aggregates and *P. australis* (*i.e.*, wetland 5 of the indoor and outdoor rig; Table 2.1) had slightly lower mean treatment performances (86 to 93%). The removal rate achieved in wetland 5 has been predominantly attributed to the presence of fertilizer (Table 2.1) enhancing the biodegradation rate, because some microbial communities are able to utilize the nitrogen component (*i.e.*, nitrate–nitrogen) of the fertilizer.

The relatively low removal rate for wetland 5 is an indication that aggregates play an indirect role in the treatment of benzene by providing habitat for nitrifying microorganisms (Eke and Scholz 2008). In comparison, the treatment efficiencies for wetlands planted with *P. australis* were similar to the efficiency of the corresponding unplanted wetlands. The treatment performances reduced during the winter of the second year (2006/7) with increasing hydrocarbon accumulation within the corresponding wetlands. This finding is in accordance with data presented previously (Cooper *et al.* 1996; Scholz 2006a).

Concentrations of DO were variable (0 to 12 mg/l). The low DO concentrations (0 to 1 mg/l) in the system during inundation could be due to the lasting effects of rapid aerobic degradation processes observed after drawing air into the system, thus causing a sudden change of the microbiology and subsequent imbalance promoting predominantly aerobic microorganisms in the wetlands. High DO variability was expected considering the relatively high influent concentrations of benzene, low retention time, small size of the experimental wetlands, and sudden environmental changes due to rapid operational changes (see above and Table 2.1).

Control of the variability of environmental variables such as temperature, pressure, and light resulted in improved overall treatment performances of the wetlands. The temperature data (standard deviation: 3.7°C) of the indoor rig were relatively stable, particularly after the temperature was fully controlled (Figure 2.29). Benzene treatment performances were better for the indoor rig versus the outdoor rig. The overall removal efficiencies were lower for the outdoor experimental rig (*e.g.*, benzene, 85 to 86%) in comparison to the experimental rig placed indoors (*e.g.*, benzene, 93 to 95%).

2.6.3.2 Impact of Volatilization

Figure 2.30 shows a comparison of benzene removal for wetlands with and without biomass. The impacts of volatilization, biodegradation, and adsorption on the

benzene removal efficiency are often difficult to separate quantitatively from each other. Preliminary findings indicate that biodegradation and adsorption dominate for a retention time of 1 d. However, volatilization becomes the major removal mechanism afterwards. Water and oil are likely to separate if the inflow is not in motion (Eke and Scholz 2008).

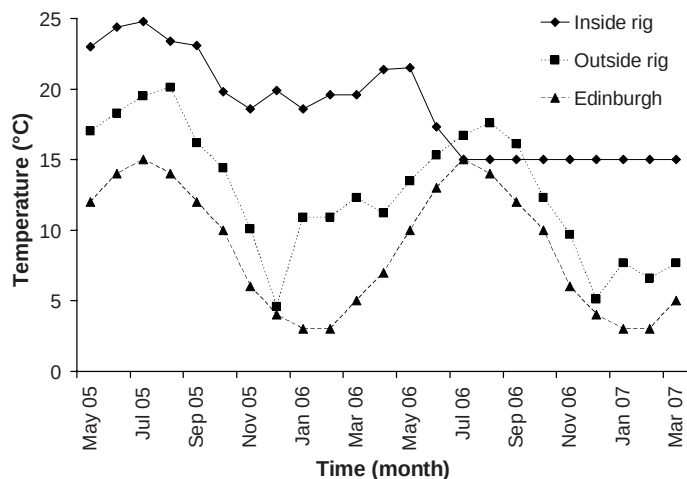


Figure 2.29 Comparison of monthly mean temperatures for inside and outside rigs, and Edinburgh (Met Office (<http://www.metoffice.gov.uk/education/secondary/teachers/ukclimate.html#3.2>))

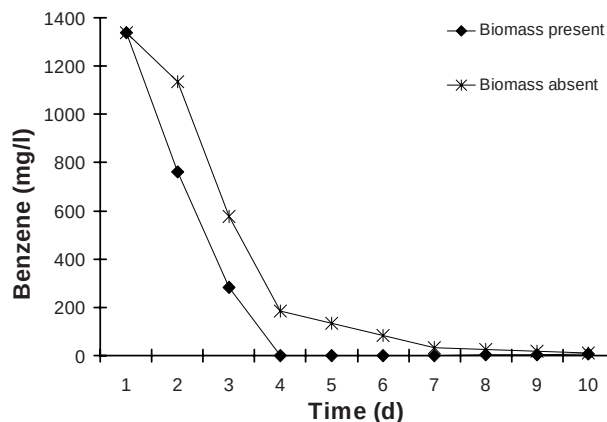


Figure 2.30 Comparison of benzene removal for wetlands with and without biomass

2.6.4 Conclusions

The findings suggest that intermittently flooded vertical-flow constructed wetlands treat benzene effectively in the presence of sufficient oxygen and fertilizer, which provides nitrate as used as an alternative electron acceptor during anaerobic periods of full inundation. Relatively high and stable temperatures (*i.e.*, no seasonal variations) lead to improved hydrocarbon treatment efficiencies.

As benzene and its degradation products started to accumulate in the wetlands, removal efficiencies subsequently diminished. Findings show also that benzene removal was highest in wetlands with aggregates and biomass providing habitat for hydrocarbon-degrading microbes. However, further studies on estimating the microbial biomass are encouraged.

Metabolic processes of microorganisms are likely to play an important role in removing hydrocarbon compounds in both controlled and semi-natural wetlands. The results show also that *P. australis* does not play a significant role (despite providing additional oxygen via its rhizomes) in removing benzene, unless sufficient nutrients (including fertilizer) are available.

Findings indicate also that both biodegradation and volatilization support treatment. Volatilization is the dominant mechanism for benzene removal after 1 d of retention time. However, optimizing environmental conditions such as locating wetlands in areas with relatively high temperatures enhances the biodegradation rate. Further research is required to quantify volatilization, aerobic and anaerobic biodegradation, adsorption, absorption, mineralization, and other removal mechanisms in large-scale constructed treatment wetlands.

References

- Abira MA, van Bruggen JJA, Denny P (2005) Potential of a tropical subsurface constructed wetland to remove phenol from pre-treated pulp and paper mill wastewater. *Wat Sci Technol* 51:173–176
- Akratos CS, Tsihrintzis VA (2007) Effect of temperature, HRT, vegetation and porous media on removal efficiency of pilot-scale horizontal subsurface flow constructed wetlands. *Ecol Eng* 29:173–191
- APHA (1998) Standard methods for the examination of water and wastewater, 20th edn. American Public Health Association (APHA), American Water Works Association and Water and Environmental Federation, Washington, DC
- Arimoro FO, Osakwe EI (2006) The influence of sawmill wood wastes on the distribution and population of macroinvertebrates at Benin River, Niger Delta area, Nigeria. *Chem Biodiv* 3:578–592
- Arvidsson S (2006) En ekotoxikologisk studie av lakvatten från timmerbevattning (An ecotoxicological study of runoff from sprinkling of roundwood). Master's thesis, LITH-IFM-EX-05/1508-SE, Linköping University, Linköping, Sweden
- Atlas RM, Cerniglia CE (1995) Bioremediation of petroleum pollutants. *BioSci* 45:332–338
- Babatunde AO, Zhao YQ, O'Neill M, O'Sullivan B (2008) Constructed wetlands for environmental pollution control: a review of developments, research and practice in Ireland. *Environ Int* 34:116–126

- Bailey HC, Elphick JR, Potter A, Chao E, Konasewich D, Zak JB (1999) Causes of toxicity in storm water runoff from sawmills. *Environ Toxicol Chem* 18:1485–1491
- Baris AJ, Eifert WH, Klotzer K, McGuckin CJ (2001) Use of a sub-surface flow constructed wetland for collection and treatment of water containing BTEX. Roux Associates, Inlandia, New York
- Bastviken SK, Eriksson PG, Premrov A, Tonderski KS (2005) Potential denitrification in wetland sediments with different plant species detritus. *Ecol Eng* 25:183–190
- Bonomo L, Pastorelli G, Zambon N (1997) Advantages and limitations of duckweed-based wastewater treatment systems. *Wat Sci Technol* 35:239–246
- Borgå P, Elowson T, Liukko K (1996a) Environmental loads from water-sprinkled softwood timber: 1. Characteristics of an open and a recycling watering system. *Environ Toxicol Chem* 15:856–867
- Borgå P, Elowson T, Liukko K (1996b) Environmental loads from water-sprinkled softwood timber: 2. Influence of tree species and water characteristics on wastewater discharges. *Environ Toxicol Chem* 15:1445–1454
- Bowmer KH, Laut P (1992) Waste-water management and resource recovery in intensive rural industries in Australia. *Wat Res* 26:201–208
- Braskerud BC (2002) Factors affecting phosphorus retention in small constructed wetlands treating agricultural non-point source pollution. *Ecol Eng* 1:41–61
- Brix H (1994) Functions of macrophytes in constructed wetlands. *Wat Sci Technol* 29:71–78
- Brix H, Schierup H-H, Arias CA (2007) Twenty years experience with constructed wetland systems in Denmark – what did we learn? *Wat Sci Technol* 56:63–68
- Burland SB, Edwards EA (1999) Anaerobic benzene biodegradation linked to nitrate reduction. *Appl Environ Microbiol* 65:529–533
- Buss SR, Herbert AW, Morgan P, Thorton SF, Smith JWN (2004) A review of ammonium attenuation in soil and groundwater. *Q J Eng Geo Hydrogeol* 37:347–359
- Cameron K, Madramootoo C, Crolla A, Kinsley C (2003) Pollutant removal from municipal sewage lagoon effluents with a free-surface wetland. *Wat Res* 37:2803–2812
- Carmichael LM, Pfaender FK (1997) The effect of inorganic and organic supplements on the microbial degradation of phenanthrene and pyrene in soils. *Biodegradation* 8:1–13
- Carty A, Scholz M, Heal K, Gouriveau F, Mustafa A (2008) The universal design, operation and maintenance guidelines for farm constructed wetlands (FCW) in temperate climates. *Biores Technol* 99:6780–6792
- Carroll P, Harrington R, Keohane J, Ryder C (2005) Water treatment performance and environmental impact of integrated constructed wetlands in the Anne valley watershed, Ireland. In: Dunne EJ, Reddy KR, Carton OT (eds) *Nutrient management in agricultural watersheds: a wetlands solution*. Wageningen Academic Publishers, Wageningen, The Netherlands, pp. 207–217
- Casson JP, Bennett DR, Nolan SC, Olson BM, Ontkian GR (2006) Degree of phosphorus saturation thresholds in manure-amended soils of Alberta. *J Environ Qual* 35:2212–2221
- Caswell PC, Gelb D, Marinello SA, Emerick JC, Cohen RR (1992) Evaluation of constructed surface-flow wetlands systems for the treatment of discharged waters from oil and gas operations in Wyoming. In: *Proceedings of the SPE Rocky Mountain regional conference*, Paper SPE 24331, Casper, WY
- Chaillan F, Chameau CH, Point V, Salot A, Oudot J (2006) Factors inhibiting bioremediation of soil contaminated with weathered oils and drill cuttings. *Environ Pollut* 144: 255–265
- Chameau CH, Rougeux G, Yepremian C, Oudot J (2005) Effects of nutrient concentration on the biodegradation of crude oil and associated microbial populations in the soil. *Soil Biol Biochem* 37:1490–1497
- Chalkias C, Petrakis M, Psiloglou B, Lianou M (2006) Modelling of light pollution in suburban areas using remotely sensed imagery and GIS. *J Environ Manag* 79:57–63
- Chen SW, Kao CM, Jou CR, Fu YT, Chang YI (2008) Use of a constructed wetland for post-treatment of swine wastewater. *Environ Eng Sci* 25:407–417
- Choi S-C, Kwon KK, Sohn JH, Kim S-J (2000) Evaluation of fertilizer additions to stimulate oil biodegradation in sand seashore mesocosms. *J Microbiol Biotechnol* 12:431–436

- Ciria MP, Solano ML, Soriano P (2005) Role of macrophyte *Typha latifolia* in a constructed wetland for wastewater treatment and assessment of its potential as biomass fuel. *Biosyst Eng* 92:535–544
- Clarke E, Baldwin AH (2002) Responses of wetland plants to ammonia and water level. *Ecol Eng* 18:257–264
- Cleneghan C (2003) Phosphorus regulations; national implementation report. Environmental Protection Agency, Wexford, Ireland
- Coates JD, Chakraborty R, McNerney MJ (2002) Anaerobic benzene biodegradation – a new era. *Res Microbiol* 153:621–628
- Conte G, Martinuzzi N, Giovannelli L, Pucci B, Masi F (2001) Constructed wetlands for wastewater treatment in central Italy. *Wat Sci Technol* 44:339–343
- Cook MG, Hunt PG, Stone KC, Canterberry JH (1996) Reducing diffuse pollution through implementation of agricultural best management practices: a case study. *Wat Sci Technol* 33:191–196
- Cooper PF, Job GD, Green MB, Shutes RBE (1996) Reed beds and constructed wetlands for wastewater treatment. Water Research Centre, Swindon, UK
- Cronk JK (1996) Constructed wetlands to treat wastewater from dairy and swine operations: a review. *Agric Ecosyst Environ* 58:97–114
- Dagley S (1986) Biochemistry of aromatic hydrocarbon degradation in *Pseudomonas*. In: So-katch J, Ornston LN (eds) *The bacteria*. Academic, Orlando, FL, USA, vol 10, pp. 527–555
- Daly K (2000) Phosphorus desorption from Irish soils. In: Tunney H (ed) Quantification of phosphorus loss from soil to water. Environmental Protection Agency, Wexford, UK, pp. 59–101
- DeHoop CF, Einsel DA, Ro KS, Chen S, Gibson MD, Grozdits GA (1998) Storm water runoff quality of a Louisiana log storage and handling facility. *J Environ Sci Health A Toxic Hazard Subst Environ Eng* 33:165–177
- Di Lecce C, Accarino M, Bolognese F, Galli E, Barbieri P (1997) Isolation and metabolic characterization of a *Pseudomonas stutzeri* mutant able to grow on the three isomers of xylene. *Appl Environ Microbiol* 63:3279–3281
- Díaz E (2004) Bacterial degradation of aromatic pollutants: a paradigm of metabolic versatility. *Int Microbiol* 7:173–180
- Doig P, van Poppelen P, Baldwin SA (2006) Characterization of particles in fresh and primary-treated log sort yard runoff. *Wat Qual Res J Can* 41:37–46
- Dorado J, Claassena FW, Lenonc G, van Beek TA, Wijnberg JBPA, Sierra-Alvarez R (2000) Degradation and detoxification of softwood extractives by sapstain fungi. *Biores Technol* 71:13–20
- Drizo A, Comeau Y, Forget C, Chapuis RP (2002) Phosphorus saturation potential: a parameter for estimating the longevity of constructed wetland systems. *Environ Sci Technol* 36:4642–4648
- Dunne EJ, Culleton N, O'Donovan G, Harrington R, Olsen AE (2005a) An integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water. *Ecol Eng* 24:221–234
- Dunne EJ, Culleton N, O'Donovan G, Harrington R, Daly K (2005b) Phosphorus retention and sorption by constructed wetland soils in Southeast Ireland. *Wat Res* 39:4355–4362
- EEC (1991) European Economic Community (EEC) Council Directive 91/676/EEC of 12 December 1991. Concerning the protection of waters against pollution caused by nitrates from agricultural sources. *Official J L* 375, 1–8
- Eke PE, Scholz M (2006) Hydrocarbon removal with constructed treatment wetlands for the benefit of the petroleum industry. In: Dias V, Vymazal J (eds) 10th international conference on wetland systems for water pollution control (23–29/09/2006). International Water Association, Lisbon, Portugal, 3:1707–1714
- Eke PE, Scholz M (2008) Benzene removal with vertical-flow constructed treatment wetlands. *J Chem Technol Biotechnol* 83:55–63
- El-Din MG, Smith DW, Al Momani F, Ikehata K (2006a) Ozone treatment for the degradation of resin and unsaturated fatty acids at low temperatures. *J Environ Eng Sci* 5:95–102

- El-Din MG, Smith DW, Al Momani F, Wang W (2006b) Oxidation of resin and fatty acids by ozone: kinetics and toxicity study. *Wat Res* 40:392–400
- Environmental Protection Agency (1998) Irish Phosphorus Regulations 1998. Water quality standards for phosphorus. Local Government (Water Pollution) Act 1977. Irish Government, Dublin, Ireland. S.I. No. 258 1998: 1998; <http://www.epa.ie/whatwedo/enforce/pa/phosphorus/>. Accessed 15 Jun 2009
- EPA (2002) Interim report: the biological survey of river quality 2001. Environmental Protection Agency, Johnston Castle, Wexford, UK
- Eriksson PG, Weisner SEB (1996) Functional differences in epiphytic microbial communities in nutrient-rich freshwater ecosystems: an assay of denitrifying capacity. *Freshwat Biol* 36: 552–562
- Erskine AD (2000) Transport of ammonium in aquifers: retardation and degradation. *Q J Eng Geo Hydrogeol* 33:161–170
- Field JA, Leyendeckers, MJH, Alvarez RS, Lettinga G, Habets LHA (1988) The methanogenic toxicity of bark tannins and the anaerobic biodegradability of water-soluble bark matter. *Wat Sci Technol* 20:219–240
- Feist WC, Springer EL, Hajny GJ (1971) Viability of parenchyma cells in stored green wood. *Tech Assoc Pulp Paper Ind (TAPPI, now TAPPI Journal)* 54:1295–1297
- Froneman A, Mangnall MJ, Little RM, Crowe TM (2001) Water bird assemblages and associated habitat characteristics of farm ponds in the Western Cape, South Africa. *Biodiv Conserv* 10:251–270
- Gottschall N, Boutin C, Crolla A, Kinsley C, Champagne P (2007) The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. *Ecol Eng* 29:154–163
- Guirguis M (2004) Treatment of waste water: a reed bed environmental case history. In: SPE International conference on health, safety, and environment in oil and gas exploration and production, Paper SPE86673, Calgary, Alberta, Canada, 29–31 March 2004
- Haberl H, Erb KH, Krausmann F, Adensam H, Schulz N (2003) Land-use change and socio-economic metabolism in Austria: II. Land-use scenarios for 2020. *Land Use Policy* 20:21–39
- Hamouri B, El Nazih J, Lahjouj J (2007) Subsurface-horizontal flow constructed wetland for sewage treatment under Moroccan climate conditions. *Desalination* 215:153–58
- Harayama S, Kasai Y, Hara AA (2004) Microbial communities in oil contaminated seawater. *Curr Opin Biotechnol* 15:205–214
- Harrington A (2005) The relationship between plant vigour and ammonium concentrations in surface waters of constructed wetlands used to treat meat industry wastewaters in Ireland. In: Dunne EL, Reddy KR, Carton OT (eds) *Nutrient management in agricultural watersheds: a wetlands solution*. Wageningen Academic Publishers, Wageningen, The Netherlands, pp. 219–223
- Harrington R, Ryder C (2002) The use of integrated constructed wetlands in the management of farmyard runoff and waste water. In: *Proceedings of the national hydrology seminar on water resource management: sustainable supply and demand*, Irish National Committees of the International Hydrological Programme and International Commission on Irrigation and Drainage. Tullamore, Offaly, UK, pp. 55–63
- Harrington R, Dunne EJ, Carroll P, Keohane J, Ryder C (2005) The concept, design and performance of integrated constructed wetlands for the treatment of farmyard dirty water. In: Dunne EL, Reddy KR, Carton OT (eds) *Nutrient management in agricultural watersheds: a wetlands solution*. Wageningen Academic Publishers, Wageningen, The Netherlands, pp. 179–188
- Harrington R, Carroll P, Carty A, Keohane J, Ryder C (2007) Integrated constructed wetlands: concept, design, site evaluation and performance. *Int J Wat* 3:243–256
- Harrison JJ, Turner RJ, Marques LLR, Ceri H (2005) Biofilms: a new understanding. *Am Sci* 93(6) 508–515
- He LS, Liu HL, Xi BD, Zhu YB (2006) Enhancing treatment efficiency of swine wastewater by effluent recirculation in vertical-flow constructed wetland. *J Environ Sci China* 18:221–226
- Head IM, Saunders JR, Pickup RW (1998) Microbial evolution, diversity, and ecology; a decade of ribosomal RNA analysis of uncultured microorganisms. *Microbiol Ecol* 35:1–21

- Healy MG, Rodgers M, Mulqueen J (2007) Treatment of dairy wastewater using constructed wetlands and intermittent sand filters. *Biores Technol* 98:2268–2281
- Hedmark Å, Jonsson M (2008) Treatment of log yard runoff in a couch grass infiltration wetland in Sweden. *Int J Environ Stud* 65:273–278
- Hedmark Å, Scholz M (2008) Review of environmental effects and treatment of runoff from storage and handling of wood. *Biores Technol* 99:5997–6009
- Hefting MM, Bobbink R, de Caluwe H (2003) Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. *J Environ Qual* 32:1194–1203
- Heider J, Spormann AM, Beller HR, Widdel F (1998) Anaerobic bacterial metabolism of hydrocarbons. *FEMS Microbiol Rev* 22:459–473
- Hiegel T (2004) Analysis of pilot scale constructed wetland treatment of petroleum contaminated groundwater. MSc thesis, Department of Civil Engineering, University of Wyoming, Laramie, WY, USA
- Hill VR, Sobsey MD (2001) Removal of Salmonella and microbial indicators in constructed wetlands treating swine wastewater. *Wat Sci Technol* 44:215–222
- Hill DT, Payne VWE, Rogers JW, Kown SR (1997) Ammonia effects on the biomass production of five constructed wetland plant species. *Biores Technol* 62:109–113
- Hill CM, Duxbury J, Geohring L, Peck T (2000) Designing constructed wetlands to remove phosphorus from barnyard runoff: a comparison of four alternative substrates. *J Environ Sci Health A Toxic Hazard Subst Environ Eng* 35:1357–1375
- Hilton J (2003) Reducing diffuse pollution to rivers: a dictionary of Best Management Practices. Conserving Natura 2000 Rivers Conservation Techniques Series No. 10, English Nature, Peterborough, UK
- Hubbard RK, Ruter JM, Newton GL, David JG (1999) Nutrient uptake and growth response of six wetland/riparian plant species receiving swine lagoon effluent. *Trans Am Soc Agric Eng* 42:1331–1341
- Hubbard RK, Gascho GJ, Newton GL (2004) Use of floating vegetation to remove nutrients from swine lagoon wastewater. *Trans Am Soc Agric Eng* 47:1963–1972
- Hunt PG, Szogi AA, Humenik FJ, Rice JM, Matheny TA, Stone KC (2002) Constructed wetlands for treatment of swine wastewater from an anaerobic lagoon. *Trans Am Soc Agric Eng* 45:639–647
- Hunt PG, Matheny TA, Stone KC (2004) Denitrification in a coastal plain riparian zone contiguous to a heavily loaded swine wastewater spray field. *J Environ Qual* 33:2367–2374
- Humenik FJ, Szogi AA, Hunt PG, Broome S, Rice M (1999) Wastewater utilization: a place for managed wetlands – review. *Asian-Australas J Animal Sci* 12:629–632
- Ibekwe AM, Grieve CM, Lyon SR (2003) Characterization of microbial communities and composition in constructed dairy wetland wastewater effluent. *Appl Environ Microbiol* 69:5060–5069
- Irish Phosphorous Regulations (1998) Water quality standards for phosphorous. Local Government (Water Pollution) Act 1977. S.I. No. 258 1998. Irish Government, Dublin, Ireland
- Janks JS, Cadena F (1991) Identification and properties of modified zeolites for the removal of benzene, toluene and xylene from aqueous solutions, paper SPE 22833. In: 1991 Society of Petroleum Engineers (SPE) annual technical conference and exhibition, Dallas, TX, 6–9 October 1991
- Jiang CL, Fan XQ, Cui GB, Zhang YB (2007) Removal of agricultural non-point source pollutants by ditch wetlands: implications for lake eutrophication control. *Hydrobiology* 581:319–327
- Jonsson M (2004) Wet storage of round wood. PhD thesis. Acta Universitatis Agriculturae Sueciae – Silvestria, vol 319. Swedish University of Agricultural Sciences, Uppsala, Sweden
- Jonsson M, Dimitriou I, Aronsson P, Elowson T (2004) Effects of soil type, irrigation volume and plant species on treatment of log yard run-off in lysimeters. *Wat Res* 38:3634–3642
- Jonsson M, Dimitriou I, Aronsson P, Elowson T (2006) Treatment of log yard runoff by irrigation of grass and willows. *Environ Pollut* 139:157–166
- Kadlec RH (1999) The limits of phosphorus removal in wetlands. *Wetland Ecol Manag* 7:165–175

- Kadlec RH (2001) Thermal environments of subsurface treatment wetlands. *Wat Sci Tech* 44:251–258
- Kadlec RH, Knight RL (1996) Treatment wetlands. CRC, Boca Raton, FL
- Kadlec RH, Knight RL, Vymazal J, Brix H, Cooper P, Haberl R (2000) Constructed wetlands for pollution control. Scientific and technical report No. 8. International Water Association Publishing, London, UK
- Kantawanichkul S, Somprasert S (2005) Using a compact combined constructed wetland system to treat agricultural wastewater with high nitrogen. *Wat Sci Technol* 51:47–53
- Kantawanichkul S, Neamkam P, Shutes RBE (2001) Nitrogen removal in a combined system: vertical vegetated bed over horizontal sand bed. *Wat Sci Technol* 44:137–142
- Kantawanichkul S, Somprasert S, Aekasin U, Shutes RBE (2003) Treatment of agricultural wastewater in two experimental combined constructed wetland systems in a tropical climate. *Wat Sci Technol* 48:199–205
- Karim MR, Manshadi FD, Karpiscak MM, Gerba CP (2004) The persistence and removal of enteric pathogens in constructed wetlands. *Wat Res* 38:1831–1837
- Karrasch B, Parra O, Cid H, Mehrens M, Pacheco P, Urrutia R, Valdovinos C, Zaror C (2006) Effects of pulp and paper mill effluents on the microplankton and microbial self-purification capabilities of the Biobio River, Chile. *Sci Total Environ* 359:194–208
- Kauppinen T, Vincent R, Liukkonen T, Grzebyk M, Kauppinen A, Welling I, Arezes P, Blacks N, Bochmann F, Campelo F, Costa M, Elsigan G, Goerens R, Kikemenis A, Kromhout H, Miguel S, Mirabelli D, McEneany R, Pesch B, Plato N, Schlunssen V, Schulze J, Sonntag R, Verougstraete V, De Vincente MA, Wolf J, Zimmermann M, Husgafvel-Pursiainen K, Savolainen K (2006) Occupational exposure to inhalable wood dust in the member states of the European Union. *Ann Occupat Hyg* 50:549–561
- Kayranli B, Scholz M, Mustafa A, Hofmann O, Harrington R (2010) Performance evaluation of integrated constructed wetlands treating domestic wastewater. *Wat Air Soil Pollut* 210:435–451
- Kellner E, Price S, Waddington JM (2004) Pressure variations in peat as a result of gas bubble dynamics. *Hydrol Proc* 18:2599–2605
- Kent DM (2000) Applied wetlands science and technology, 2nd edn. CRC, Boca Raton, FL
- Keohane J, Carroll P, Harrington R, Ryder C (2005) Integrated constructed Wetlands for farm-yard dirty water treatment: a site suitability assessment. In: Dunne EJ, Reddy KR, Carton OT (eds) Nutrient management in agricultural watersheds: a wetlands solution. Wageningen Academic Publishers, Wageningen, The Netherlands, p 196–206
- Kim S, Choi DH, Sim DS, Oh Y (2005) Evaluation of bioremediation effectiveness on crude oil-contaminated sand. *Chemosphere* 59:845–852
- Kivaisi AK (2001) The potential for constructed wetlands for wastewater treatment and reuse in developing countries: a review. *Ecol Eng* 16:545–560
- Knight RL (1999) The use of treatment wetlands for petroleum industry effluents. *Environ Sci Technol* 33:973–980
- Knight RL, Payne Jr WE, Borer RE, Clarke Jr RA, Pries JH (2000) Constructed wetlands for livestock wastewater management. *Ecol Eng* 15:41–55
- Kowalchuk GA, Stephen JR, De Boer W, Prosser JI, Embley TM, Woldendorp JW (1997) Analysis of ammonia-oxidizing bacteria of the β subdivision of the class Proteobacteria in coastal sand dunes by denaturing gradient gel electrophoresis and sequencing PCR-amplified 16S ribosomal DNA fragments. *Appl Environ Microbiol* 63:1489–1497
- Kraus H (1985) Noise abatement in sawmills. *Holz-Zentralblatt* 111:1833–1834
- Lahvis MA, Baehr AL, Baker RJ (1999) Quantification of aerobic biodegradation and volatilization rates of gasoline hydrocarbons near the water table under natural attenuation conditions. *Wat Resour Res* 35:753–765
- Lansing SL, Martin JF (2006) Use of an ecological treatment system (ETS) for removal of nutrients and solids from dairy wastewater. *Ecol Eng* 28:235–245
- Leahy JG, Olsen RH (1997) Kinetics of toluene degradation by toluene-oxidizing bacteria as a function of oxygen concentration, and the effect of nitrate. *FEMS Microbiol Ecol* 23:23–30

- Lee CY, Lee CC, Lee FY, Tseng SK, Laio CJ (2004) Performance of subsurface flow constructed wetland taking pre-treated swine effluent under heavy loads. *Biores Technol* 92:173–179
- Lee B-H, Scholz M, Horn A (2006) Constructed wetlands for the treatment of concentrated stormwater runoff (Part A). *Environ Eng Sci* 23:191–202
- Leinweber P, Meissner R, Eckhardt K-U, Seeger J (1999) Management effects on forms of phosphorus in soil and leaching losses. *Eur J Soil Sci* 50:413–424
- Li L, Li Y, Kumar Biswas D, Nian Y, Jiang G (2008) Potential of constructed wetlands in treating the eutrophic water: evidence from Taihu Lake of China. *Biores Technol* 99:1656–1663
- Lindholm EL, Berg S (2005) Energy requirement and environmental impact in timber transport. *Scand J For Res* 20:184–191
- Liukko K. (1997) Climate-adapted wet storage of saw timber and pulpwood: An alternative method of sprinkling and its effect on freshness of roundwood and environment. PhD thesis. *Acta Universitatis Agriculturae Sueciae – Silvestria*, No. 51. Swedish University of Agricultural Sciences, Uppsala, Sweden
- Liukko K, Elowson T (1995) Climate controlled sprinkling of saw timber. In: Web Proceedings of the International Union of Forestry Research Organizations (IUFRO) XX World Congress, P3.07 Meeting, Tampere, Finland, 6–12 August, pp. 44–53. Oregon State University, Corvallis, OR, USA (<http://www.metla.fi/iufro/iufro95/index.htm>)
- Liukko K, Elowsson T (1999) The effect of bark condition, delivery time and climate-adapted wet storage on the moisture content of *Picea abies* (L.) Karst. pulpwood. *Scand J For Res* 14:156–163
- Lovley DR, Lloyd JR (2000) Microbes with a mettle for bioremediation. *Nat Biotechnol* 18: 600–601
- Machate T, Noll BHH, Kettrup A (1997) Degradation of Phenanthrene and hydraulic characteristics in a constructed wetland. *Wat Res* 31:554–560
- Maguire RO, Edwards AC, Wilson MJ (1997) Comparison of chemical forms and distribution of phosphorus within cultivated and uncultivated soils; some implications for losses. In: Tunney H, Carton OT, Brooks PC, Johnson AE (eds) *Phosphorus losses from soil to water*. CAB Wallingford, UK, pp. 427–430
- Mantovi P, Marmiroli M, Maestri E, Tagliavini S, Piccinni S, Marmiroli N (2003) Application of a horizontal subsurface flow constructed wetland on treatment of dairy parlor wastewater. *Biores Technol* 88:85–94
- Masbough A, Frankowski K, Hall KJ, Duff SJB (2005) The effectiveness of constructed wetland for treatment of woodwaste leachate. *Ecol Eng* 25:552–566
- McCuskey SA, Conger AW, Hillestad HO (1994) Design and implementation of functional wetland mitigation – case studies in Ohio and South Carolina. *Wat Air Soil Pollut* 77: 513–532
- McMaster ME, Hewitt LM, Parrott JL (2006) A decade of research on the environmental impacts of pulp and paper mill effluents in Canada: field studies and mechanistic research. *J Toxicol Environ Health B Crit Rev* 9:319–339
- Met Éireann (2007) <http://www.met.ie/climate/climate-data-information.asp>. Accessed 15 Jan 2010
- Mitsch WJ, Gosselink JG (1993) *Wetlands*. 2nd edn. Wiley, New York
- Mitsch WJ, Gosselink JG (2007) *Wetlands*. 4th edn. Wiley, New York
- Molle P, Prost-Boucle S, Lienard A (2008) Potential for total nitrogen removal by combining vertical flow and horizontal flow constructed wetlands: a full-scale experiment study. *Ecol Eng* 34:23–29
- Moshiri GA (2000) *Constructed wetland for water quality improvement*. CRC, Boca Raton, FL, USA
- Munoz I, Rieradevall J, Torrades F, Peral J, Domenech X (2006) Environmental assessment of different advanced oxidation processes applied to a bleaching Kraft mill effluent. *Chemosphere* 62:9–16
- Mustafa A, Scholz M, Harrington R, Carrol P (2009) Long-term performance of a representative integrated constructed wetland treating farmyard runoff. *Ecol Eng* 35:779–790

- Myers JE, Jackson LM (2001) An evaluation of the Department of Energy Naval Petroleum Reserve No. 3 – Produced water bio-treatment facility. In: Proceedings of SPE/EPA/DOE exploration and production environmental conference, SPE Paper 66522, San Antonio, TX
- Nair I, Jayachandran K, Shankar S (2007) Treatment of paper factory effluent using a phenol degrading *Alcaligenes sp.* under free and immobilized conditions. *Biores Technol* 98: 714–716
- Newman JM, Clausen JC, Neafsey JA (1999) Seasonal performance of a wetland constructed to process dairy milkhouse wastewater in Connecticut. *Ecol Eng* 14:181–198
- Nicholson BC (1987) Changes in volatiles composition of *Pinus radiata* on wet storage. *Holzfor-schung* 41:209–213
- Nielsen LP, Christensen PB, Revsbech NP, Sørensen J (1990) Denitrification and oxygen respi-ration in biofilms studied with a microsensor for nitrous oxide and oxygen. *Microb Ecol* 19:63–72
- NIWA (1997) Guidelines for constructed wetland treatment of farm dairy wastewaters in New Zealand. Science and Technology Series No. 48, National Institute of Water and Atmospheric Research Limited (NIWA), <http://www.niwasience.co.nz/pubs/st/st48.pdf>. Accessed 25 Sep 2007
- Omari K (2003) Hydrocarbon removal in an experimental gravel bed constructed wetland. *Wat Sci Technol* 48:275–281
- Orban JL, Kozak RA, Sidle RC, Duff SJB (2002) Assessment of relative environmental risk from log yard runoff in British Columbia. *For Chr* 78:146–151
- Oudot J, Merlin FX, Pinvidic P (1998) Weathering rates of oil components in a bioremediation experiment in estuarine sediments. *Mar Environ Res* 45:113–125
- Paje MLF, Neilan BA, Couperwhite IA (1997) *Rhodococcus* species that thrives on medium saturated with liquid benzene. *Microbiology* 143:2975–2981
- Pant HK, Reddy KR, Lemon E (2001) Phosphorus retention capacity of root bed media of sub-surface flow constructed wetlands. *Ecol Eng* 17:345–355
- Pease BC (1974) Effects of log dumping and rafting on the marine environment of southeast Alaska. USDA Forest Service General Technical Report PNW-22, Pacific Northwest Forest and Range Experiment Station, US Department of Agriculture, Forest Service, Portland, OR
- Peek RD, Liese W (1974) The effect of wet storage of windthrown timber on water quality. *Forstwissenschaftliches Zentralblatt* 96:348–357
- Picard C, Fraser HL, Steer D (2005) The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. *Biores Technol* 96:1039–1047
- Picek T, Cizkova H, Dusek J (2007) Greenhouse gas emissions from a constructed wetland – plants as important sources of carbon. *Ecol Eng* 31:98–106
- Poach ME, Hunt PG, Sadler EJ, Matheny TA, Johnson MH, Stone KC, Humenik FJ, Rice JM (2002) Ammonia volatilisation from constructed wetlands that treat swine wastewater. *Trans Am Soc Agric Eng* 45:619–627
- Poach ME, Hunt PG, Vanotti MB, Stone KC, Matheny TA, Johnson MH, Sadler EJ (2003) Improved nitrogen treatment by constructed wetlands receiving partially nitrified liquid swine manure. *Ecol Eng* 20:183–197
- Poach ME, Hunt PG, Reddy GB, Stone KC, Matheny TA, Johnson MH, Sadler EJ (2004) Ammonia volatilization from marsh-pond-marsh constructed wetland treating swine wastewater. *J Environ Qual* 33:844–851
- Poe AC, Pichler MF, Thompson SP, Paerl HW (2003) Denitrification in a constructed wetland receiving agricultural runoff. *Wetlands* 23:817–826
- Prantner SR, Kanwar RS, Lorimor JC, Pederson CH (2001) Soil infiltration and wetland micro-cosm treatment of liquid swine manure. *Appl Eng Agric* 17:483–488
- Puustinen M, Jormola J (2005) Constructed wetlands for nutrient retention and landscape diver-sity. International Commission on Irrigation and Drainage 21st European Regional Confer-ence 2005. Frankfurt an der Oder, Germany
- Ragusa SR, McNevin D, Qasem S, Mitchell C (2004) Indicators of biofilm development and activity in constructed wetlands microcosms. *Wat Res* 38:2865–2873

- Ran N, Agami M, Oron G (2004) A pilot study of constructed wetlands using duckweed (*Lemna gibba* L.) for treatment domestic primary effluent in Israel. *Wat Res* 38:2241–2248
- Reddy KR, D'Angelo EM (1997) Biogeochemical indicators to evaluate pollutant removal efficiency in constructed wetlands. *Wat Sci Technol* 35:1–10
- Reddy KR, Kadlec RH, Flaig E, Gale PM (1999) Phosphorus retention in streams and wetlands. *Crit Rev Environ Sci Technol* 29:83–146
- Reddy GB, Hunt PG, Phillips R, Stone K, Grubbs A (2001) Treatment of swine wastewater in marsh-pond-marsh constructed wetlands. *Wat Sci Technol* 44:545–550
- Rew S, Mulamootil G (1999) A cost comparison of leachate treatment alternatives. In: Mulamootil G, McBean EA, Rovers F (eds) *Constructed wetlands for the treatment of landfill leachates*. Lewis, Boca Raton, FL, USA
- Richardson CJ, Marshall PE (1986) Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecol Monogr* 56:279–302
- Rice RW, Erich MS (2006) Estimated VOC losses during the drying of six eastern hardwood species. *For Prod J* 56:48–51
- Rice RW, Izuno FT, Garcia RM (2002) Phosphorus load reductions under best management practices for sugarcane cropping systems in the Everglades Agricultural Area. *Agric Wat Manag* 56:17–39
- Ridgeway HF, Safarik J, Phipps D, Carl P, Clark D (1990) Identification and catabolic activity of well-derived gasoline-degrading bacteria and a contaminated aquifer. *Appl Environ Microbiol* 56:3565–3575
- Rintala JA, Puhakka JA (1994) Anaerobic treatment in pulp- and paper-mill waste management: a review. *Biores Technol* 47:1–18
- Sakadevan K, Bavor H (1998) Phosphate adsorption characteristics of soils, Slags and zeolite to be used as substrates in constructed wetland systems. *Wat Res* 32:393–399
- Samis SC, Liu SD, Wernick BG, Nassichuk MD (1999) Mitigation of fisheries impacts from the use and disposal of wood residue in British Columbia and the Yukon. Canadian Technical Report of Fisheries and Aquatic Sciences 2296. Fisheries and Oceans Canada, and Environment Canada. Ottawa, Ontario, Canada
- Schaafsma JA, Baldwin AH, Streb CA (2000) An evaluation of a constructed wetland to treat wastewater from a dairy farm in Maryland, USA. *Ecol Eng* 14:199–206
- Scholz M (2004) Treatment of gully pot effluent containing nickel and copper with constructed wetlands in a cold climate. *J Chem Technol Biotechnol* 79:153–162
- Scholz M (2006a) *Wetland systems to control urban runoff*. Elsevier, Amsterdam, The Netherlands
- Scholz M (2006b) Practical sustainable urban drainage system decision support tools. *Inst Civ Eng Eng Sustain* 159:117–125
- Scholz M (2007) Classification methodology for sustainable flood retention basins. *Landsc Urban Plann* 81:246–256
- Scholz M, Xu J (2001) Comparison of vertical-flow constructed wetlands for treatment of wastewater containing lead and copper. *J Chart Inst Wat Environ Manag* 15:287–293
- Scholz M, Xu J (2002) Performance comparison of experimental constructed wetlands with different filter media and macrophytes treating industrial wastewater contaminated with lead and copper. *Biores Technol* 83:71–79
- Scholz M, Lee B-H (2005) Constructed wetlands: a review. *Int J Env Stud* 62:421–447
- Scholz M, Höhn P, Minall R (2002) Mature experimental constructed wetlands treating urban water receiving high metal loads. *Biotechnol Progr* 18:1257–1264
- Scholz M, Sadowski AJ, Harrington R, Carroll P (2007a) Integrated constructed wetlands assessment and design for phosphate removal. *Biosyst Eng* 97:415–423
- Scholz M, Harrington R, Carroll P, Mustafa A (2007b) The integrated constructed wetlands (ICW) concept. *Wetlands* 27:337–354
- Seo DC, Cho JS, Lee HJ, Heo JS (2005) Phosphorus retention capacity of filter media for estimating the longevity of constructed wetland. *Wat Res* 39:2445–2457

- Sezerino PH, Reginatto V, Santos MA, Kayser K, Kunst S, Philippi LS, Soares HM (2003) Nutrient removal from piggery effluent using vertical flow constructed wetlands in southern Brazil. *Wat Sci Technol* 48:129–135
- Shappell NW, Billey LO, Forbes D, Matheny TA, Poach ME, Reddy GB, Hunt PG (2007) Estrogenic activity and steroid hormones in swine wastewater through a lagoon constructed wetland system. *Environ Sci Technol* 41:444–450
- Sharples AN (1995) Identifying sites vulnerable to phosphorus losses in agricultural runoff. *J Environ Qual* 24:947–951
- Shipin O, Koottatep T, Khanh NTT, Polprasert C (2005) Integrated natural treatment systems for developing communities: low-tech N-removal through the fluctuating microbial pathways. *Wat Sci Technol* 51:299–306
- Sievers DM (1997) Performance of four constructed wetlands treating anaerobic swine lagoon effluents. *Trans Am Soc Agric Eng* 40:769–775
- Skarda SM, Moore JA, Niswander SF, Gamroth MJ (1994) Preliminary results of a wetland for treatment of dairy farm wastewater. In: DuBow PJ, Reaves RP (eds) *Constructed wetlands for animal wastewater management*. Proceedings of workshop, 4–6 Apr 1994, Purdue University, West Lafayette, IN, pp. 34–42
- Solano-Serena F, Marchal R, Ropars M, Lebeault J-M, Vandecasteele J-P (1999) Biodegradation of gasoline: kinetics, mass balance and fate of individual hydrocarbons. *J Appl Microbiol* 86:1008–1016
- Solano ML, Soriano P, Ciria MP (2003) Constructed wetlands as a sustainable solution for wastewater treatment in small villages. *Biosyst Eng* 87:109–118
- Sommer SG, Christensen BT, Nielsen NE, Schjorring JK (1993) Ammonia volatilization during storage of cattle and pig slurry – effect of surface cover. *J Agric Sci* 121:63–71
- Soukup A, Williams RJ, Cattell FCR, Krough MH (1994) The function of a coastal wetland as an efficient remover of nutrients from sewage effluent: a case study. *Wat Sci Technol* 29:295–304
- Souza SM, Araújo OQF, Coelho MAZ (2008) Model-based optimization of a sequencing batch reactor for biological nitrogen removal. *Biores Technol* 99:3213–3223
- Stone KC, Hunt PG, Szogi AA, Humenik EJ, Rice JM (2002) Constructed wetland design and performance for swine lagoon wastewater treatment. *Trans Am Soc Agric Eng* 45:723–730
- Stone KC, Poach ME, Hunt PG, Reddy GB (2004) Marsh-pond-marsh constructed wetland design analysis for swine lagoon wastewater treatment. *Ecol Eng* 23:127–133
- Ström L, Christensen TR (2007) Below ground carbon turnover and greenhouse gas exchanges in a sub-arctic wetland. *Soil Biol Biochem* 39:1689–1698
- Sugai SF, Lindstrom JE, Braddock JF (1997) Environmental influences on the microbial degradation of Exxon Valdez oil on the shorelines of Prince William Sound, Alaska. *Environ Sci Technol* 31:1564–1572
- Sun G, Zhao Y, Allen S, Cooper D (2006) Generating “tide” in pilot-scale constructed wetlands to enhance agricultural wastewater treatment. *Eng Life Sci* 6:560–565
- Sundberg C, Tonderski K, Lindgren P-E (2007) Potential nitrification and denitrification and the corresponding composition of the bacterial communities in a compact constructed wetland treating landfill leachates. *Wat Sci Technol* 56:159–166
- Szogi AA, Hunt PG (2001) Distribution of ammonium-N in the water-soil interface of a surface-flow constructed wetland for swine wastewater treatment. *Wat Sci Technol* 44:157–162
- Szogi AA, Hunt PG, Humenik EJ (2000) Treatment of swine wastewater using a saturated-soil-culture soybean and flooded rice system. *Trans Am Soc Agric Eng* 43:327–335
- Szogi AA, Hunt PG, Humenik FJ (2003) Nitrogen distribution in soils of constructed wetlands treating lagoon wastewater. *Soil Sci Soc Am J* 67:1943–1951
- Szogi AA, Hunt PG, Sadler EJ, Evans DE (2004) Characteristics of oxidation-reduction processes in constructed wetlands for swine wastewater treatment. *Appl Eng Agric* 20:189–200
- Tanner CC (1996) Plants for constructed wetland treatment systems – a comparison of the growth and nutrient uptake of eight emergent species. *Ecol Eng* 7:59–58

- Tanner CC, Clayton JS, Upsdell MP (1995) Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands: II. Removal of nitrogen and phosphorus. *Wat Res* 29:27–34
- Tao W, Hall KJ, Masbough A, Frankowski K, Duff SJB (2005) Characterization of leachate from a woodwaste pile. *Wat Qual Res J Can* 40:476–483
- Tao W, Hall KJ, Duff SJB (2006) Treatment of woodwaste leachate in surface flow mesocosm wetlands. *Wat Qual Res J Can* 41:325–332
- Tao W, Hall K, Hall E (2007) Laboratory study on potential mechanisms for treatment of woodwaste leachate in surface flow constructed wetlands. *J Environ Eng Sci* 6:85–94
- Taylor BR, Carmichael NB (2003) Toxicity and chemistry of aspen wood leachate to aquatic life: field study. *Environ Toxicol Chem* 22:2048–2056
- Taylor BR, Goudey JS, Carmichael NB (1996) Toxicity of aspen wood leachate to aquatic life: laboratory studies. *Environ Toxicol Chem* 15:150–159
- Thomas PR, Glover P, Kalaroopan T (1995) An evaluation of pollutant removal from secondary treated sewage effluent using a constructed wetland system. *Wat Sci Technol* 32:87–93
- Thorén A-K, Legrand C, Tonderski KS (2004) Temporal export of nitrogen from a constructed wetland: influence of hydrology and senescing submerged plants. *Ecol Eng* 23:233–249
- Throbäck NI, Enwall K, Jarvis Å, Hallin S (2004) Reassessing PCR primers targeting *nirS*, *nirK* and *nosZ* genes for community surveys of denitrifying bacteria with DGGE. *FEMS Microbiol Ecol* 49:401–417
- Tokida T, Miyazaki T, Mizoguchi M, Seki K (2005) *In situ* accumulation of methane bubbles in a natural wetland soil. *Eur J Soil Sci* 56:389–395
- Trias M, Hu Z, Mortula MM, Gordon RJ, Gagnon GA (2004) Impact of seasonal variation on treatment of swine wastewater. *Environ Technol* 25:775–781
- Tunney H, Breeuwsma A, Withers PJA, Ehlert PAI (1997) Phosphorus fertiliser strategies: past, present and future. In: Tunney H, Carton OT, Brookes PC, Johnson AE (eds) *Phosphorus losses from soil to water*. CAB International, New York, pp. 358–361
- Uğurlu M, Karaoğlu MH, Kula (2006) Experimental investigation of chemical oxygen demand, lignin and phenol removal from paper mill effluents using three-phase three-dimensional electrode reactor. *Pol J Environ Stud* 15:647–654
- UNEP (2003) Convention on biological diversity. In: 9th Meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (10–14 November 2003), Montreal, United Nations Environment Programme (UNEP)/CBD/SBSTTA/9/INF/4, New York, USA
- USEPA (1995a) Constructed wetlands for wastewater treatment and wildlife habitat: 17 case studies. United States of America Environmental Protection Agency (USEPA) Number: 832R93005, <http://www.epa.gov/owow/wetlands/pdf/hand.pdf>. Accessed 1 Jan 2010
- USEPA (1995b) Constructed wetlands for animal waste treatment: manual on performance, design, and operation with case histories. United States of America Environmental Protection Agency (USEPA) Number: 855B97001, <http://www.epa.gov/owow/wetlands/pdf/hand.pdf>. Accessed 1 Jan 2010
- USEPA (1988) Constructed Wetlands and Aquatic Plant Systems for Municipal Wastewater Treatment. EPA/625/1-88/022. United States Environmental Protection Agency (USEPA), Office of Research and Development, Cincinnati, OH, USA
- USEPA (2000). Constructed Wetlands Treatment of Municipal Wastewaters. EPA/625/R-99/010. United States Environmental Protection Agency (USEPA), Office of Research and Development, Cincinnati, OH, USA
- USDA (1991) A Guide to Creating Wetlands for Agricultural Wastewater, Domestic Wastewater, Coal Mine Drainage Stormwater. Volume 1, General Considerations. United States Department of Agriculture. <http://www.epa.gov/OWOW/wetlands/pdf/hand.pdf>. Accessed 1 Jul 2009
- Vacca G, Wand H, Nikolaus M, Kusch P, Kastner M (2005) Effect of plants and filter materials on bacteria removal in pilot-scale constructed wetlands. *Wat Res* 39:1361–1373
- Vrhovsek D, Kukanja V, Bulc T (1996) Constructed wetland (CW) for industrial wastewater treatment. *Wat Res* 30:2287–2292

- Vymazal J (2007) Removal of nutrients in various types of constructed wetlands. *Sci Total Environ* 380:48–65
- Wallace SD, Knight RL (2006) Small-scale constructed treatment systems: feasibility, design criteria, and O&M requirements. Final report, Project 01-CTS-5. Water Environment Research Foundation, Alexandria, VA, USA
- Wang H, Gielen GJ, Judd ML, Stuthridge TR, Blackwell BG, Tomer MD, Pearce SH (1999) Treatment efficiency of land application for thermomechanical pulp mill effluent constituents. *Appita J* 52:383–386
- WDOE (1995) Best management practices to prevent storm water pollution at log yards. Publication 95-53, Washington State Department of Ecology (WDOE), Olympia, WA, USA
- Webber J, Gibbs J (1996) Water storage of timber: experience in Britain. Bulletin 117, Forestry Commission, Her Majesty's Stationery Office, London, UK
- Weishampel P, Kolka R, King JY (2009) Carbon pools and productivity in a 1-km² heterogeneous forest and peatland mosaic in Minnesota, USA. *For Ecol Manag* 25:747–754
- Welling I, Mielo T, Raisanen J, Hyvarinen M, Liukkonen T, Nurkka T, Lonka P, Rosenberg C, Peltonen Y, Svedberg U, Jappinen P (2001) Characterization and control of terpene emissions in Finnish sawmills. *Am Ind Hyg Assoc J* 62:172–175
- Wemple C, Hendricks L (2000) Documenting the recovery of hydrocarbon-impacted wetlands: a multi-disciplinary approach. In: Means JL, Hinchey RE (eds) *Wetlands and remediation: an international conference*. Battelle, Columbus, OH, USA, pp. 73–78
- Werker AG, Dougherty JM, Mchenry JL, Van Loon VA (2002) Treatment variability for wetland wastewater treatment design in cold climates. *Ecol Eng* 19:1–11
- Wheeler EF, Topper PA, Graves RE, Bruns MA, Wysocki CJ (2007) Odour-reduction performance of constructed wetlands treating diluted swine manure. *Appl Eng Agric* 23:621–630
- Woodhouse C, Duff JBS (2004) Treatment of log yard runoff in an aerobic trickling filter. *Wat Qual Res J Can* 39:230–236
- Wu Y, Chung A, Tama NFY, Pia N, Wong MH (2008) Constructed mangrove wetland as secondary treatment system for municipal wastewater. *Ecol Eng* 34:137–146
- Xia H (2003) Ecological effectiveness of constructed wetlands in treating oil-refined wastewater. In: *Proceedings of the 3rd international conference in Guangzhou, China, 6–9 Oct 2003*
- Yang L, Chang H, Huang ML (2001) Nutrient removal in gravel- and soil-based wetland microcosms with and without vegetation. *Ecol Eng* 18:91–105
- Zedler JB (2003) Wetlands at your service: reducing impacts of agriculture at the watershed scale. *Front Ecol Env* 1:65–72
- Zenaitis MG, Duff SJB (2002) Ozone for removal of acute toxicity from log yard run-off. *Ozone Sci Eng* 24:83–90
- Zenaitis MG, Sandhu H, Duff SJB (2002) Combined biological and ozone treatment of log yard runoff. *Wat Res* 36:2053–2061
- Zelikoff JT, Chen LC, Cohen MD, Schlesinger RB (2002) The toxicology of inhaled wood-smoke. *J Toxicol Environ Health B Crit Rev* 5:269–282
- Zhang L, Scholz M, Mustafa A, Harrington R (2009) Application of the self-organizing map as a prediction tool for an integrated constructed wetland agroecosystem treating agricultural runoff. *Biores Technol* 100:539–565
- Zhao YQ, Sun G, Allen SJ (2004) Anti-sized reed bed system for animal wastewater treatment: a comparative study. *Wat Res* 38:2907–2917
- Zheng J, Nanbakhsh H, Scholz M (2006) Case study: design and operation of sustainable urban infiltration ponds treating storm runoff. *J Urban Plann Develop Am Soc Civ Eng* 132:36–41
- Zhou E, Crawford RL (1995) Effects of oxygen, nitrogen, and temperature on gasoline biodegradation in soil. *Biodegradation* 6:127–140



<http://www.springer.com/978-1-84996-458-6>

Wetland Systems

Storm Water Management Control

Scholz, M.

2011, XVIII, 235 p., Hardcover

ISBN: 978-1-84996-458-6