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ARTICLE

Integrated Constructed Wetlands (ICW): Ecological Development in Constructed Wetlands for Manure Treatment

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Abstract In Flanders (Belgium), treatment systems based on constructed wetlands have been successfully implemented to treat the liquid fraction of separated pig manure, resulting in water that meets the stringent discharge criteria. As these systems have proven their nutrient removal performance, attention has moved towards the concept of biodiversity development from a perspective of Integrated Constructed Wetlands (ICWs). The spontaneous colonization of constructed wetlands with macroinvertebrates was examined to quantify the added biological value presented by a farmyard treatment wetland. In total, 17 taxa were found. The diversity of macroinvertebrates increased along the water treatment path in the system and was higher in summer than in autumn. Several pollution sensitive taxa like *Orthetrum*, *Ischnura*, and *Cloeon* were found at the last basins of the treatment wetland chain. Most species were insects, since their adults can easily aerially colonize these recently constructed systems. Water quality was assessed using the Multimetric Macroinvertebrate Index Flanders (MMIF), and showed a general increase towards the end of the treatment. The MMIF increased significantly with decreasing nutrient levels. ICWs adequately combine the treatment of wastewater with supporting and enhancing biodiversity in agricultural landscapes.

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Introduction

Intensive animal production in many regions in the world has resulted in considerable local overproduction of animal manure. This overproduction, if not properly managed, may lead to pressure on environmental quality resulting from excessive emission of nitrogen and phosphorus. For instance, the region in which the current study is situated (Flanders, Belgium) is characterized by a dense animal population (3000 animals per km^2; Sanitel-Pigs 2010) related to pig production and comprises an area of 450–560 km^2. The contribution of nitrogen emissions from agriculture to surface water pollution is 55% for the European Union (Kersebaum et al. 2003). Specifically in Flanders, the contribution of agriculture to total eutrophying emissions is approximately 65% (VMM 2007). The resulting pressure on the environment has resulted in a number of European and regional legislative frameworks stimulating/enforcing the implementation of corrective measures to reduce the environmental impact of intensive farming. Based on model simulations, Volk et al. (2009) determined that in order to achieve the desired European water quality targets in the Upper Ems River basin (Northern Germany), drastic measures would be required, which are unrealistic from a socio-economic point of view. The problems associated with intensive farming in relation to stringent European water quality targets have been a driving force for technological innovations to process manure. The use of wetlands for tertiary treatment of the liquid fraction of animal manure in combination with conventional secondary biological treatment has been proposed (Meers et al. 2005) and was subsequently demonstrated at the field-scale level (Meers et al. 2008). Effluent concentrations were consistently below regional legal discharge criteria: < 15 mg l^{-1} total nitrogen (N), <2 mg l^{-1} total phosphorus (P), <25 mg l^{-1} biological oxygen demand (BOD), <35 mg l^{-1} suspended solids, and < 125 mg l^{-1} chemical oxygen demand (COD) (Meers et al. 2008). Cronk (1996) and Carty et al. (2008) in their reviews about farm constructed wetlands generally concluded that they may require considerably more land than conventional treatment technologies, yet that such conventional technologies are usually associated with more maintenance and capital costs.

The conversion of liquid fraction into clean and transparent discharge water of good physico-chemical quality as presented by Meers et al. (2008) in itself presented a milestone in the treatment of excessive amounts of pig manure and reduction of the nutrient concentrations related to intensive industrial farming. A next natural step in wetland engineering involves attention for 'softer' aspects of environmental engineering, namely landscape integration and ecological habitat creation. Campbell and Ogden (1999) stated that a decentralized system of treating sewage wastes with constructed wetlands not only can provide an economical and energy efficient means of achieving treatment objectives, but also a resource in the form of reclaimed water available for landscape irrigation or creation of wildlife habitats. Although there have been numerous seminars and conferences on the technical aspects of wetland design, engineering, chemistry, and biological activity mostly relating to treatment performance, there has been a noticeable lack of attention to their aesthetic, landscape, wildlife habitat, and other multiple-use values.

Various authors have stated that wetlands fed with nutrient rich influents can be significantly more biologically diverse than other treatment wetlands (van der Valk 2006; Scholz et al. 2007; Jurado et al. 2008). The current study wishes to examine the concept of ecological development in treatment systems specifically designed for manure processing, from a perspective of Integrated Constructed Wetlands (ICWs). ICWs is a concept promoted by the Irish National Parks and Wildlife Service, in which ICWs are free surface flow wetlands, which closely resemble natural wetlands and are based on the holistic and interdisciplinary use of land to control water quality. The ICW concept is based on the following principles (Harrington et al. 2005): (1) the containment and treatment of influents within shallow vegetated ponds using local soil material, wherever possible, (2) the aesthetic placement of the containing wetland structure into the local landscape ('landscape fit') to enhance a site's ancillary values, and (3) the enhancement of habitat and biodiversity.

Whereas most research papers focus on the first principle, namely the treatment aspect of such wetlands (Scholz et al. 2010), only little attention has been paid to the latter two. Current work was aimed to assess the spontaneous colonization by invertebrate taxa of wetlands designed for treatment of liquid fraction of animal manure in order to make a quantified appreciation of the added biological value presented by a farmyard treatment wetland. Moreover, the role of the macroinvertebrates as ecological indicators and their impact on the treatment efficiency is analysed. Several methods on how the treatment performance and ecological value as well as the 'landscape fit' can be improved are discussed.

Methods

Study Site and Experimental Design

The investigated constructed wetland is situated in Ichtegem (Belgium). The system involves a multi-bed system, including eight beds/lagoons, with both vertical and horizontal flow helophyte beds and ponds. The system includes two horizontal flow fields, two vertical percolation fields, two underwater systems, and two ponds systems (Fig. 1). The constructed wetland is only fed by process water and the final effluent is discharged into a nearby river. The total surface area of the pilot-scale system is 4500 m^2. A more detailed description of the system and manure treatment (seasonal performance, economic cost, …) is available in Meers et al. (2008).

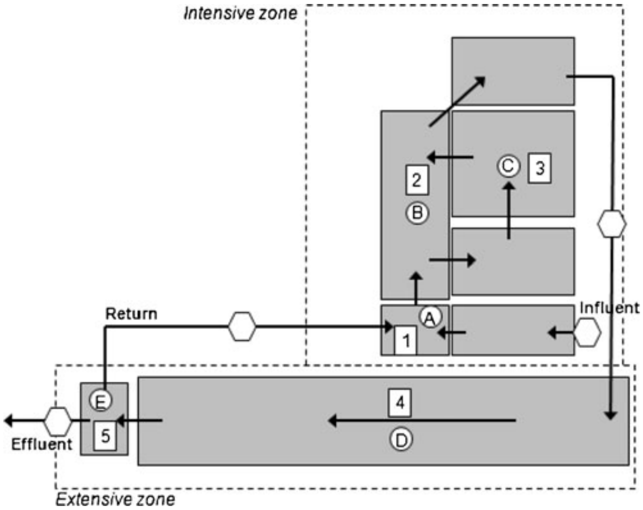

Fig. 1 Schematic overview of the integrated constructed wetland, consisting of an intensive zone and an extensive zone. The white squares represent the five different sampling locations for biological analysis (1–5), whereas the white circles represent the five sampling locations for chemical analysis (A–E)

In our study, macroinvertebrates were monitored to assess the spontaneous colonization and the improvement of the water quality in the different basins of a wetland used to treat the thin fraction of pig manure. It was expected that the species richness and the sensitivity of the species would increase with improving water quality. To test this hypothesis the physico-chemical water quality and quantity at the in- and out-flow of the system was determined over an extended period of time (September 2007–July 2008) at five different sampling locations (A–E; Fig. 1). The first location was situated at the mixing basin, the second at the middle of the first lagoon, the third at the horizontal flow reed bed, the fourth in the rectangular helophyte bed, and the last in the pond receiving the effluent wastewater (Fig. 1). These different locations were chosen along a water quality gradient, where the first basin received high loads of nutrients from incoming wastewater, whereas the last pond contained the purified water. Subsequently these physico-chemical measurements were correlated with the biological quality (assessed in the same basins at the physico-chemical measurements) and ecological development assessed in two sampling campaigns, once in autumn (November 2007) and once in summer (July 2008), along this water quality gradient.

Physico-Chemical Quality

Over the entire course of the current study (September 2007–July 2008) the loading and discharge rate of the wetland was monitored by means of flow meters at inlet and outlet points. Using these data the hydraulic loading rate was calculated as the volume per day applied over the wetland surface area ($l\ m^{-2}\ d^{-1}$). In addition, the corresponding masses of ingoing and outgoing N, P, and COD were determined. Starting two months prior to the period of the biological sampling campaigns, water samples for chemical analysis were collected to assess the water quality gradient throughout the wetland (September–November 2007 and May–July 2008). Samples were collected once per 2–4 weeks at five locations within the system and subsequently analysed for conductivity (EC), N content, P content, and COD.

Conductivity was measured using an LF 537 conductivity electrode (Wissenschaftlich Technischen Werkstäten, Weilheim, Germany). Total N was determined using a modified Kjeldahl procedure (Van Ranst et al. 1999) and total P content was determined using the colorimetric method of Scheel (Van Ranst et al. 1999). Two standards (25 ppm; 50 ppm) were used to calibrate absorbance in function of P concentration. The absorbance at 700 nm of samples and standards was determined using a Jenway 6400 spectrophotometer (Barloworld Scientific T/As Jenway, Felsted, United Kingdom). COD was determined photometrically using standardized cuvette tests (Dr. Bruno Lange, GmbH & Co, KG Düsseldorf, GE).

Biological Analysis

Biological sampling was carried out with artificial substrates (AS) in order to assess the water quality based on the presence of macroinvertebrates. These AS consisted of polypropylene bags, with a volume of 5 l, filled with substrate similar to the one used in the different basins. Three substrates were placed at five different locations (1–5) in the constructed wetland (Fig. 1) and retrieved after a minimum of 3 weeks of colonisation according to the guidelines for assessment of macroinvertebrates in Flanders (Gabriels et al. 2010). One sampling campaign was conducted in autumn (November 2007) and the second one in summer (July 2008). All collected material was thoroughly examined for the presence of macroinvertebrates, sorted, and preserved in Norvanol D. All species were identified according to the taxonomic levels defined by Gabriels et al. (2010) needed for the calculation of the Multimetric Macroinvertebrate Index Flanders (MMIF). Identification to the family or genus level was done using the key of De Pauw and Vannevel (1990). After identification, the total number of individuals of each taxon was recorded. The MMIF was calculated to determine the water quality of the different ponds (Gabriels et al. 2010). The index calculation is a type-specific multimetric system based on five equally weighted metrics: taxon richness, number of Ephemeroptera, Plecoptera, and Trichoptera taxa, number of other sensitive taxa, the Shannon-Wiener diversity index, and the mean tolerance score. The tolerance score ranges from 10 for taxa that are very pollution sensitive to 1 for taxa that are very pollution tolerant. The final index value is expressed as an Ecological Quality Ratio ranging from zero for bad ecological quality to one for very good ecological quality. Since these constructed wetlands can be classified as lentic systems with slightly brackish water ($Na>250\ mg\ l^{-1}$), the latter was used to calculate the MMIF. Based on the collected measurements for conductivity, total P, total N and COD, the relationship with the MMIF was tested using Spearman Rank Correlation Coefficient. Statistical analyses were performed using Statistica 7.0 (Statsoft Inc. 2004).

Results

Physico-Chemical Water Quality

Daily loading rate of the treatment wetland, expressed as $l\ m^{-2}\ d^{-1}$ manure fed to the system, over the course of the current study was 1.51 $lm^{-2}\ d^{-1}$ in autumn 2007, 0.62 l m^{-2} d^{-1} in winter 2007–2008, 2.02 l m^{-2} d^{-1} in spring 2008, and 2.47 l m^{-2} d^{-1} in summer 2008. During the overall period covered in the current study (September 2007–July 2008), the system received and treated 2450 m³ manure into dischargeable water, corresponding with 759 kg N, 25 kg P, and 1718 kg COD. The overall removal efficiency of the treatment wetland amounted to 95.4, 95.9, and 79.5% for N, P, and COD, respectively (Table 1). The minimum, maximum, and average observed values of N, P, COD, and conductivity decreased throughout the treatment wetland during both monitoring campaigns (September–November 2007 (Table 2A) and May–July 2008 (Table 2B)).

Biological Assessment

In total, 17 taxa were recorded for all ponds during the two seasons. During autumn, the highest number of eight taxa was found at the end basin (location 5), whereas during summer the highest numbers of 9 and 10 taxa were found at locations 3 and 4, respectively (Fig. 2a). A higher number of taxa per location was observed during summer when compared to autumn, especially at the inflow, where no taxa were recorded during autumn. At the last basin (location 5), the number of taxa did not change between seasons (Fig. 2a). At the end of the treatment, several pollution sensitive taxa could be observed (Table 3). In terms of abundance, *Corixa* and *Chironomus* group *thummi-plumosus* were the dominant taxa at the different locations in the wetland (Table 3). There was an increase in MMIF observed along the system during both winter and summer, although the increase was less pronounced during summer (Fig. 2b). The maximum MMIF value was obtained at the end basin during autumn as well as summer with MMIF values of 0.5 and 0.35, respectively (Fig. 2b). MMIF values had significant negative correlations with conductivity (R=0.67), total P (R=0.80), total N (R=0.81), and COD (R=0.86) (Fig. 3).

Discussion

Treatment Performance

A key factor in the use of (integrated) constructed wetlands is the operational performance as evidenced by sufficient nutrient removal. In our study, N and P were the target components of the manure treatment process and needed to be efficiently removed to safeguard the environment, particularly the waterway that received the treated effluent. Physico-chemical analysis demonstrated that, notwithstanding high loading rates, more than 90% of N and P was removed in the wetland (Table 1). Previous studies have already demonstrated extensively that nutrients such as N and P are effectively removed by microbial processes (N; nitrification/denitrification) as well as by plant uptake (N, P) in wetlands (Vymazal 2007). Taking into account that nutrient removal was our primary goal we suggest annual biomass harvest and removal to avoid nutrient build up within the system. This is not in line with the suggestions stated by Harrington and McInnes (2010), in their outlines for Integrated Constructed Wetlands (ICWs) for livestock wastewater management. They propose allowing the organic matter to build up at a rate of 2–4 cm per annum, and specifically design the systems to allow for long-term detritus accumulation. However, by not removing plant biomass that is fed high nutrient concentrations from the system, an annual accumulation of up to 120 kg P ha^{-1} could result. It would cause a P saturated substrate to be established, constituting a potential leaching of nutrients. The decrease in nutrients occurred gradually through the wetland, until the point that both N and P were effectively reduced to levels below the legal discharge criteria of 2 mg l^{-1} for P and 15 mg l^{-1} for N

Table 1 Monthly hydraulic loading rate (HLR, l m^{-2} d^{-1}), in and outflow rate (m^3) and loading and discharge rate (kg) for total nitrogen (N), total phosphorus (P) and chemical oxygen demand (COD) for the constructed wetland from September 2007 until July 2008

	Loading				Discharge				
	In (m^3)	N (kg)	P (kg)	COD (kg)	Out (m^3)	N (kg)	P (kg)	COD (kg)	HLR (l m^{-2} d^{-1})
Sep	193	86	1.4	196	176	1.1	0.0	18	1.4
Oct	144	72	1.4	108	233	1.7	0.1	19	1.0
Nov	281	104	9.0	311	480	12.0	0.2	60	2.1
Dec–Feb	250	130	3.0	129	412	7.3	0.1	30	0.6
Mar	242	74	1.0	84	393	2.6	0.1	29	1.7
Apr	312	93	2.1	134	341	3.2	0.3	44	2.3
May	284	56	2.3	219	182	1.7	0.1	37	2.0
Jun	227	16.3	1.7	154	100	0.7	0.0	21	1.7
Jul	517	128	3.5	383	507	4.4	0.2	95	3.7

Table 2 Minimum (min), maximum (max), and average (avg) observed values of conductivity (EC), total phosphorus (P), chemical oxygen demand (COD), and total nitrogen (N) throughout the treatment wetland in A) Autumn (September–November 2007) and B) Summer (May–July 2008). Sampling locations (Influent, A–E) are indicated in Fig. 1

		EC (mS cm^{-1})	P (mg l^{-1})	COD (mg l^{-1})	N (mg l^{-1})
A) Autumn					
Influent	Max	20	110	1500	540
	Min	12	4.3	535	324
	Avg	14	36	1071	419
A	Max	12	12	1116	356
	Min	11	11	116	112
	Avg	12	11	632	255
B	Max	9.6	5.6	432	180
	Min	8.9	1.1	240	123
	Avg	9.3	1.6	314	141
C	Max	10	<0.3	292	132
	Min	8.5	<0.3	164	57
	Avg	9.3	<0.3	179	86
D	Max	9.3	<0.3	292	49
	Min	5.3	<0.3	69	9
	Avg	7.2	<0.3	145	30
E	Max	6.3	0.4	146	11
	Min	4.1	< 0.3	69	5.8
	Avg	5.6	< 0.3	112	7.8
B) Summer					
Influent	Max	13	13.2	930	300
	Min	11	7.6	660	72
	Avg	12	10	748	167
A	Max	12	10.3	1008	180
	Min	5.3	1	405	48
	Avg	8.7	3.2	640	83
B	Max	9.3	4.3	600	96
	Min	7.6	1.2	296	42
	Avg	8.6	2.7	467	72
C	Max	10.4	1.6	332	62
	Min	9.3	0.4	196	20
	Avg	9.7	0.9	276	43
D	Max	10	1.6	332	25
	Min	6.9	0.3	136	8
	Avg	8.8	0.8	240	13.5
E	Max	9.3	0.9	244	11
	Min	4.4	0.3	136	6.8
	Avg	7.2	0.6	204	9

(Table 2). These data are in line with previous conducted research by Meers et al. (2008) and demonstrate that the wetland was representative to assess the spontaneous development of biological diversity in manure-treating wetlands.

Biological Gradient: Ecological Assessment

In total, 17 taxa were recorded for all ponds during the two seasons. A general increase in the number of taxa could be observed along the treatment pathway in the system in autumn as well as in summer. The general increase in diversity during summer was mainly due to an increase in several taxa belonging to Diptera, whereas the abundance and number of taxa belonging to Odonata and Ephemeroptera decreased. Many species of these taxa mature and emerge during spring or early summer and consequently were not found in the samples at the end of July. Moreover, the growth of several plant species and an increase in organic material during spring and summer led to an increase in abundance of detrivorous and algivorous taxa, like *Corixa* and Culicidae. Although the

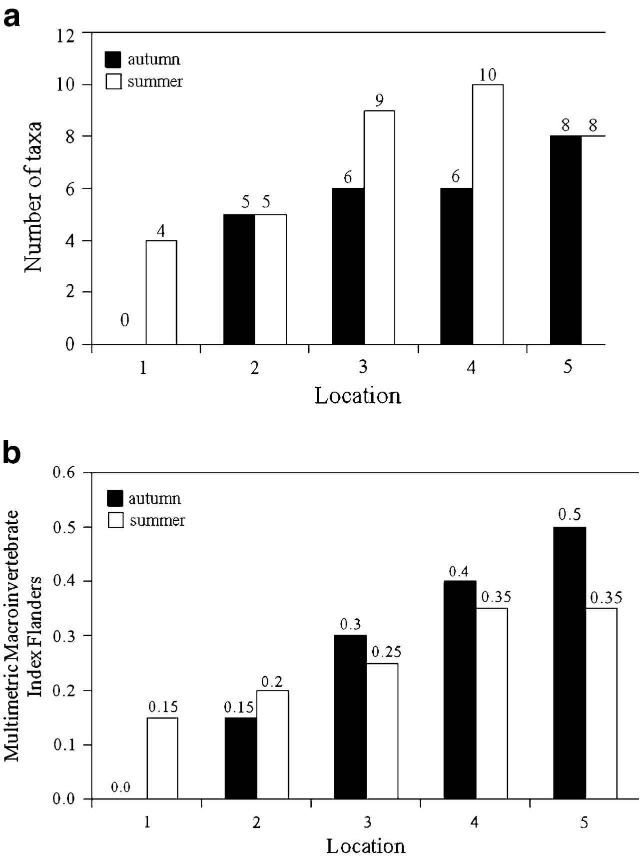

Fig. 2 **a** Total number of taxa per location for autumn (*black*) and summer (*white*) and **b** the Multimetric Macroinvertebrate Index Flanders (MMIF) per location for autumn (*black*) and summer (*white*)

chemical water quality was best in the last basin, this was—during summer—not reflected in the number of taxa. This could be due to differences in substrate, pond depth, and vegetation. Numerous studies have observed a positive relation between macroinvertebrate diversity and aquatic vegetation (Balcombe et al. 2005; Jurado et al. 2009). Since the end basin was a rather deep pond with no emergent vegetation, steep edges, and only a few submersed plants and algae, macroinvertebrate diversity could be lower. According to Hansson et al. (2005), at least four years are needed to reach maximum species richness of benthic macroinvertebrates in constructed wetlands. Because our system was operational for only two years, an increase in the number of taxa might be expected with the ageing of the system. Moreover, several studies indicate that macrophyte species richness increases with time, which could positively influence macroinvertebrate diversity as well.

Besides a higher diversity towards the end of the treatment system, more pollution sensitive taxa were found in the rectangular horizontal flow reed bed and the end basin. In autumn, the most sensitive taxon was *Orthetrum* (Odonata) with a tolerance score of 7, whereas during summer the most sensitive taxa *Ischnura* (Odonata) and *Cloeon* (Ephemeroptera) had tolerance scores of 6, somewhat less sensitive to pollution (Table 3). At the beginning of the wetland, *Chironomus* group *thummi-plumosus* and *Corixa* occurred almost exclusively; *C.* group *thummi-plumosus* is a complex of species that are very pollution tolerant, allowing them to survive in nutrient rich waters, with low oxygen availability, and rapid temperature changes (Rosenberg 1992). Therefore, it was not surprising to find them in high numbers in the initial ponds receiving influent. In terms of abundance, the macroinvertebrate community was mainly dominated by *Chironomus* group *thummi-plumosus* during autumn at all locations. During summer, several taxa including *Corixa*, *Chironomus* group *thummi-plumosus* and Culicidae reached high abundances (> 500 individuals/sample) at different locations in the system.

Macroinvertebrates that aerially colonize dominated the studied constructed wetlands. This was expected since there was no direct connection to nearby rivers or ponds. However, indirect connections with ponds situated nearby could act as sources for colonizers. Connectivity between ponds can have an important effect on pond community structure since some macroinvertebrates require the presence of other waterbodies in the surrounding area for

Table 3 Overview of the most sensitive taxa and the dominant taxa found in the samples at each location (1–5) for autumn and summer, with relative tolerance scores (1 = not sensitive to pollution, 10 = very sensitive to pollution)

Location	Most sensitive taxon		Dominant taxon	
	Autumn	Summer	Autumn	Summer
1	No species	*Corixa* (5)	No species	*Corixa* (5)
2	*Corixa* (5), *Notonecta* (5), *Lymnaea* (5)	Dytiscidae (5), Hydrophilidae (5), *Corixa* (5)	*C.* group *thummi-plumosus* (2)	Culicidae (3)
3	*Ischnura* (6)	Dytiscidae (5), Hydrophilidae (5), *Corixa* (5), *Lymnaea* (5)	*C.* group *thummi-plumosus* (2)	*C.* group *thummi-plumosus* (2)
4	*Orthetrum* (7)	*Ischnura* (6)	*C.* group *thummi-plumosus* (2)	*Corixa* (5)
5	*Orthetrum* (7)	*Cloeon* (6)	*C.* group *thummi-plumosus* (2)	*C.* group *thummi-plumosus* (2), *Corixa* (5)

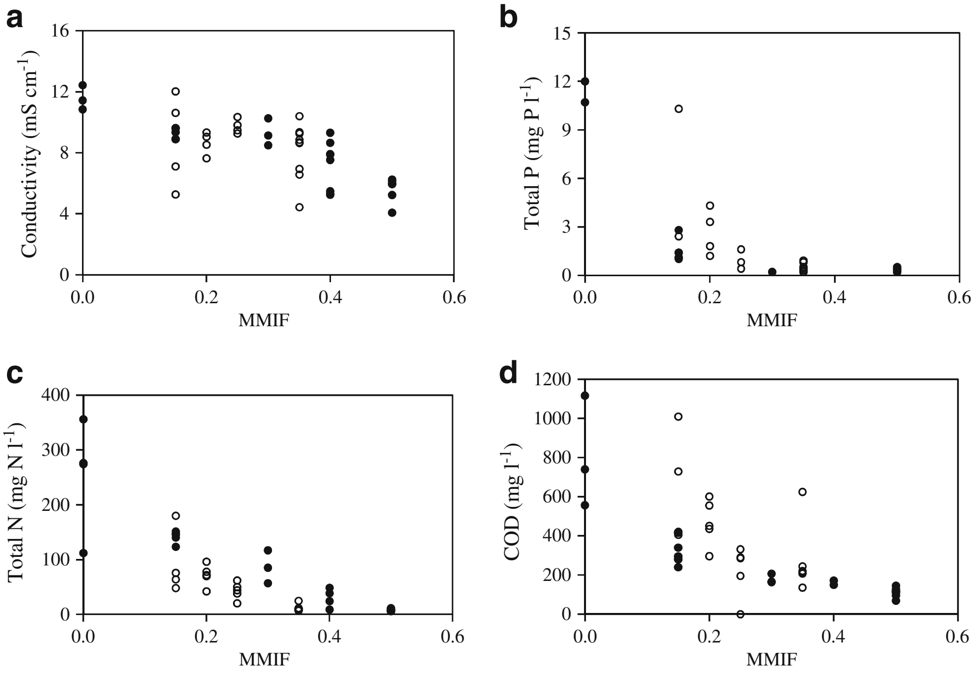

Fig. 3 Relationship between Multimetric Macroinvertebrate Index Flanders (MMIF) and **a** conductivity, **b** total phosphorus concentration (Total P), **c** total nitrogen concentration (Total N), and **d** chemical oxygen demand (COD) for autumn (*black circle*) and summer (*white circle*)

dispersal (Anderson and Smith 2004; Jurado et al. 2009). Taxa that were absent included Hirudinea and Crustacea. Hirudinea can easily be spread via birds (Davies et al. 1982), and their absence might be due to relatively recent construction (see also Balcombe et al. 2005 and Spieles and Mitsch 2000). Crustacea occur in most fresh, brackish, and marine environments, regardless of water quality (De Pauw and Vannevel 1990). Some of the missing taxa, especially crustaceans, have a detritivorous diet and might become important decomposers of organic material. Inoculating treatment wetlands with tolerant macroinvertebrates could enhance decomposition processes, benefitting the systems. *Asellus aquaticus* seems a good candidate species for this approach because it has a broad natural distribution in Flanders and can withstand high levels of nutrients (Messiaen et al. 2010). Attention should be paid to salinity tolerance because the water in the treatment system is often more saline compared to natural rivers.

Besides their function as litter decomposers, macroinvertebrates can contribute other wetland functions by assisting in nutrient cycling (Cummins 1973) and regulating plant communities (Weller 1994). Macroinvertebrates can contribute to the transfer of nutrients from sediments, detritus, and water column to higher trophic levels, including fish and waterfowl (Balcombe et al. 2005).

Other measures could be taken to enhance ecosystem functioning and food web stability. An increase in the complexity of the vegetation as well as a decrease in shore slope could enhance abundance and diversity of aquatic insects (Nilsson et al. 1994; Hansson et al. 2005). Introducing fish could enhance biodiversity and food web structure. Three-spined stickleback (*Gasterosteus aculeatus*) is a species that could be introduced in these systems because the species tolerates low water quality (Belpaire et al. 2000), although temperature fluctuations could be a problem. Besides the added value of increasing the complexity of the food web, fish species could be used as an overall biological assessment (and surveillance) tool (see Karr et al. 1986; Breine et al. 2004).

The European Water Framework Directive states that all natural surface waters should have good ecological water quality by the end of 2015. If this is applied to constructed wetlands, which can be seen as slightly brackish lentic waters, an MMIF score of 0.7 should be reached (Gabriels et al. 2010). Although these requirements were not met in the constructed wetland (MMIF=0.5) the biological water quality improved considerably from the beginning of the wetland towards the end. Moreover, we should keep in mind that evaluating this system based on the MMIF can give a biased impression since this method was developed to evaluate natural surface waters. Based on the MMIF, the effluent reached a moderate water quality during autumn, but quality during summer was poor. Low water quality in summer could be due to higher nutrient loads or algal blooms causing oxygen depletion at night. Moreover, several insects emerged during summer and were not encountered in samples, reducing MMIF values. Nevertheless, the results obtained were satisfactory because the chemical quality of the effluent wastewater was below Flemish discharge criteria (Meers et al. 2008). Low nutrient concentrations and low conductivity coincided, in autumn as well as in winter, with high MMIF values. This would be expected since better chemical water quality often leads to a higher biological water quality. The lower nutrient levels at the end of the treatment system allowed pollution sensitive species to establish, which was reflected in the higher MMIF. Relatively high P and N concentrations in winter might have negatively affected macroinvertebrates in the first basin. However, results indicate marked improvements in key water quality properties as well as in the biological index throughout the treatment wetland. Besides removing P and N, the wetland also fulfils a biological function, by providing habitat for wetland species and contributing to the increase of the total catchment taxon richness (Jurado et al. 2010).

Landscape Fit

Harrington et al. (2005) describe the 'aesthetic placement' of the containing wetland structure into the local landscape to enhance a site's ancillary values (the landscape fit) as one of the three underlying principles of ICWs. In the design of the wetland under study, the vegetated ponds with emergent species (helophyte systems) provide a green buffer adjacent to the existing industrialized pig farm. The elongated (70 m) piggery stables are thus obscured from view by the green barrier provided by the standing vegetation (up to 2–3 m height) during the growing season. Embedding the farm in this green surrounding provides a more aesthetically pleasing view from the surrounding open space. The current wetlands were rectangular in shape due to practical limitations involved with construction. However, in the future, designs could be developed to use more natural lines to further improve the natural look of systems. When practical, local materials were used during construction of the wetlands. Specific attention was paid to lining the wetlands. Respecting the 'prevention principle', the upstream intensive zone, where nutrient concentrations are substantially higher, should use plastic foil lining. However, lower concentrations downstream may justify switching to more natural, cheaper, and more environmentally sustainable products such as local clays. In the upstream part of the study system, waterproofing was provided by a synthetic low-density polyethylene liner, whereas more downstream, local clay materials were used for waterproofing.

Conclusion

In the future, embedding intensive pig farms in a green wetland buffer will add to the ecosystem services that such systems can provide and will improve the landscape in agro-industrial areas (which are currently determined to a large extent by monotonous monocultural land-use and industrialized farming). These wetlands will help to further reduce the environmental pressures originating from industrial farming activities.

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