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Influence of vegetation on the removal of heavy metals and nutrients in a constructed wetland

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Abstract

A free water surface wetland was built to treat wastewater containing metals (Cr, Ni, Zn) and nutrients from a tool factory in Argentina. Water, sediment and macrophytes were sampled in the inlet and outlet area of the constructed wetland during three years. Three successive phases of vegetation dominance were developed and three different patterns of contaminant retention were observed. During the *Eichhornia crassipes* dominance, contaminants were retained in the macrophyte biomass; during the *E. crassipes* + *Typha domingensis* stage, contaminants were retained in the sediment and in the *T. domingensis* dominance stage, contaminants were retained in sediment and in the macrophyte biomass. Removal efficiency was not significantly different among the three vegetation stages, except for NH_4^+ and i-P_{diss}. Because of its highest tolerance, *T. domingensis* is the best choice to treat wastewater of high pH and conductivity with heavy metals, a common result from many industrial processes.

Keywords: Free-water surface constructed wetland; Removal efficiency; Macrophyte; Metallurgic factory; Effluent treatment

1. Introduction

Constructed wetlands (CWs) were mainly used for nutrient and organic matter retention in domestic and municipal sewage, storm water and agricultural runoff (Hammer, 1989; Moshiri, 1993; Kadlec and Knight, 1996; Vymazal et al., 1998; Kadlec et al., 2000). The application of CW for industrial wastewater treatment is a promising alternative in Argentina, since there is a large availability of marginal land around most cities with a low population density. The central and northern areas of the country have mild winters, allowing extended growing periods for plants.

The choice of plants is an important issue in CWs, as they must survive the potentially toxic effects of the effluent and its variability. Common reeds (*Phragmites australis* (Cav.) Trin.), cattails (*Typha* spp.), bulrushes (*Scirpus* spp.) and reed canary grass (*Phalaris arundinacea* L.) have been used for both domestic and industrial wastewater treatment (Shepherd et al., 2001; Mbuligwe, 2005; Vymazal, 2005; Vymazal and Krópfelová, 2005). Although regionally abundant macrophyte species are adapted to the local climatic and edaphic conditions, their performance under the environmental conditions imposed by wastewater was unknown. Bahco metallurgic industry constructed a small-scale experimental wetland to assess the feasibility of treating wastewater at its tool factory in Santo Tomé, Santa Fe (Argentina). Metals, nutrients, biological and chemical oxygen demand largely decreased in the wetland effluent (Maine et al., 2005). The success in improving wastewater quality led this industrial plant to develop a large-scale CW to treat the wastewater of the whole factory. The CW has been in operation since 2003. The wastewater received a primary treatment (precipitation, sieving and decantation); however it contained Cr, Ni and Zn and showed high pH and conductivity. An assemblage of locally common macrophytes

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was transplanted to the studied wetland. Plant growth showed three different periods, the first one dominated by *Eichhornia crassipes* (Mart.) Solms., followed by a decline of *E. crassipes* and increased *Typha domingensis* Pers. cover period, and finally a *T. domingensis* dominance period. The present contribution assesses the removal efficiency of this CW and compares it during the different macrophyte dominance stages.

2. Study site

A free water surface wetland was constructed at the Bahco Argentina metallurgic plant, located in Santo Tomé, Santa Fe, Argentina (S 31° 40′; W 60° 47′). It is 50 m long, 40 m wide and 0.5–0.8 m deep, with a central baffle dividing it in two identical sections forcing the effluent to cover double the distance, reaching a 5:1 length–width ratio (Fig. 1). Mean wastewater discharge was 100 $m^3 d^{-1}$ throughout the experiment. Water residence time ranged from 7 to 12 days. The wetland was rendered impermeable by means of a bentonite liner to reach a hydraulic conductivity of 10^{-7} m s^{-1} (5 compacted layers of bentonite − approximate total depth: 0.6 m). A layer of 1 m of soil was placed on top of the bentonite layer. Different locally available macrophyte species were transplanted into the wetland, with *E. crassipes*, *T. domingensis* and *Pontederia cordata* L. reaching the largest cover. After passage through the wetland, wastewater is discharged by a canal into a 1.5 ha pond.

This study began in March 2003 when both sewage and industrial effluent were treated together. During the first stage of the wetland operation (from October 2002 to February 2003) only diluted sewage from the factory was treated (the composition of the influent was 25 $m^3 d^{-1}$ of sewage + 75 $m^3 d^{-1}$ of pond water). Sewage consisted of wastewater from a staff of 750 employees of the factory. Later (March 2003), industrial wastewater and sewage were treated together (25 $m^3 d^{-1}$ of sewage + 75 $m^3 d^{-1}$ of industrial wastewater). The idea of treating sewage was based on the fact that sewage composition is rich in organic matter and favors macrophyte growth. High nutrient concentrations could improve the ability of macrophytes to take up heavy metals from wastewater (Manios et al., 2003). This hypothesis was corroborated by our research group (Hadad et al., 2006a).

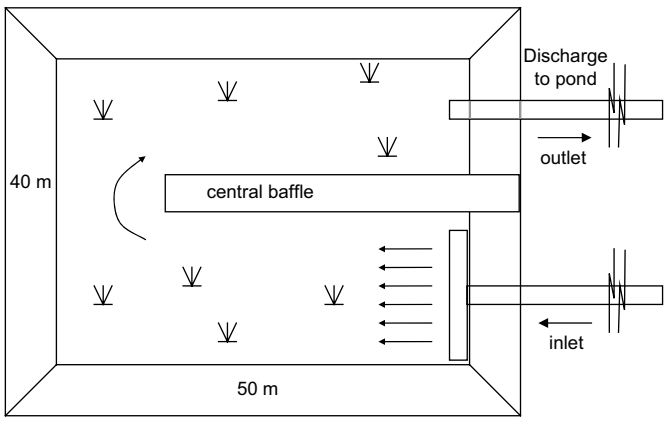

Fig. 1. Layout of the large-scale constructed wetland.

3. Materials and methods

3.1. Water

Forty-seven samplings of the influent and effluent were performed from March 2003 until March 2006. Sampling frequency was approximately every two weeks from March to October 2003 and monthly since then. Samples were collected in triplicate. Conductivity was measured with a YSI 33 conductimeter, dissolved oxygen (DO) with a Horiba OM-14 portable meter and pH with an Orion pH-meter. Water samples were filtered through Millipore membrane filters (0.45 μm) for soluble P and N determinations. Chemical analyses were performed following APHA (1998). NO_2^- was determined by coupling diazotation followed by a colorimetric technique. NH_4^+ and NO_3^- by potentiometry (Orion ion selective electrodes, sensitivity: 0.01 mg l^{-1} of N, reproducibility: ±2%). Dissolved inorganic phosphate (i-P_{diss}) was determined by the colorimetric molybdenum blue method (Murphy and Riley, 1962). Ca^{2+} and Mg^{2+} were determined by EDTA titration. Na^+ and K^+ were determined by flame emission photometry. Alkalinity (carbonate and bicarbonate) was measured by HCl titration. Cl^- was determined by the argentometric method. SO_4^{2-} was assessed by turbidimetry. Chemical oxygen demand (COD) was determined by the open reflux method and biochemical oxygen demand (BOD) by the 5-Day BOD test (APHA, 1998). Total Fe, Cr, Ni and Zn concentrations were determined in water samples by atomic absorption spectrometry (by flame or electrothermal atomization, according to the sample concentration, Perkin Elmer 5000), following APHA (1998).

Statistical significance between inlet and outlet water concentrations was assessed using a mean comparison test ($p < 0.05$). ANOVA analysis was performed to compare the removal efficiency of the measured parameters during the different macrophyte dominance stages. A level of $p < 0.05$ was used in all comparisons. Duncan's test was used to differentiate means when appropriate. A level of $p < 0.05$ was used in all comparisons.

3.2. Sediment

Sediment total phosphorous (TP), total Kjeldahl nitrogen (TKN), Ni, Cr and Zn concentrations were determined monthly or bimonthly in the inlet and outlet areas of the wetland. Sediment samples were collected in triplicate using a 4-cm diameter PVC corer. All the samples were transported to the laboratory in the cold. TP was determined after acid digestion with $HClO_4$:HNO_3:HCl (7:5:2) mixture followed by i-P_{diss} determination in the digested samples (Murphy and Riley, 1962). Cr, Ni and Zn were determined in the same digests by atomic absorption spectrometry (Perkin Elmer 5000) and TKN was determined by the Kjeldahl method (APHA, 1998).

P fractionation in sediment was performed following the EDTA method (Golterman, 1996). This sequential extraction uses Ca-EDTA + dithionite to extract iron-bound phosphate ($Fe(OOH) \approx P$) and then Na_2-EDTA to extract calcium-bound phosphate ($CaCO_3 \approx P$). For the organically bound P-fractions, acid soluble organic phosphate and alkali soluble organic phosphate (org $P_{\rightarrow acid}$ and org $P_{\rightarrow alk}$), this sequence is followed by an extraction with H_2SO_4 at 20 °C and then by NaOH at 90 °C, respectively. All extractions were carried out under rotational continuous shaking. The duration of each extraction stage is described in Golterman (1996). i-P_{diss} was measured in the extracts using the blue-molybdate method with some modifications suggested by Golterman (1996). For i-P_{diss} determination in the Ca-EDTA extract, 5 ml of the extract was used and the solution was heated (75 °C).

TP, TKN and metal concentrations were compared between initial and successive samplings during the study period at inlet and outlet by a mean comparison test ($p < 0.05$).

3.3. Macrophytes

Macrophytes were sampled monthly with a 0.50 × 0.50 m square sampler following Vesk and Allaway (1997). Four replicates were taken randomly at the inlet in each sampling. The macrophytes were then harvested and sorted by species in the laboratory, washed, separated between above (stems and leaves) and belowground parts (roots of floating species and roots plus rhizomes of emergent macrophytes). TP, TKN, Cr, Ni and Zn in roots and leaves were determined in the same way as in the sediment samples. To measure dry weight, plants were dried at 105 °C until constant weight was reached (APHA, 1998). Plant cover was estimated measuring the area occupied by each species in the wetland.

4. Results and discussion

4.1. Wetland efficiency

The composition of the influent showed an important variation range during the study period (Table 1). The concentrations of the parameters measured at the effluent of the wetland were found to be below the guide levels for industrial wastewater according to national regulations. The effluent showed not only lower concentrations but lower variability of the parameters measured regarding the influent, proving the buffer capacity of the wetland.

The DO concentration at the inlet and outlet showed a high variability, being anoxic at several samplings. At the outlet, DO concentrations at the surface layer were different than concentrations the bottom, the latter being lower. Differences in DO concentrations induced differences between surface and bottom concentrations in most other parameters (Maine et al., 2006). Since the outlet pipe was placed at the bottom, it was decided to build a wall at the outlet to force the effluent into forming a small waterfall, which in turn favored the increase in DO concentration. Slight modifications in design may remarkably improve the efficiency of contaminant removal.

NO_3^- and NO_2^- were removed (Table 1), while NH_4^+ was not efficiently removed. i-P_{diss} concentration of the effluent

Table 1
Mean loading rates (g m^{-2} d^{-1}), mean and range of temperature (°C), conductivity (μS cm^{-1}), concentrations (mg l^{-1}) of the influent and effluent and removal percentages in the constructed wetland along the studied period

Parameter	Influent			Effluent			Mean removal (%)
	Loading rate (g m^{-2} d^{-1})	Concentration		Loading rate (g m^{-2} d^{-1})	Concentration		
		Mean	Range		Mean	Range	
Temperature	–	19.1	10.2–28	–	16.7	6.6–26	–
pH	–	9.01[a]	6.5–12.3	–	7.66[b]	6.9–9.1	–
Conductivity	–	3249[a]	480–8500	–	1799[b]	470–5000	–
DO	–	1.53[a]	0–7.1	–	0.898[b]	0–7.5	–
Suspended solids	137.9	2758.4[a]	699–8550	72.5	1450.9[b]	524–3693	36
Ca^{2+}	7.90	157.9[a]	24–651	3.33	66.7[b]	17.3–268	34
Mg^{2+}	0.81	16.2[a]	0.5–59	0.77	15.5[a]	3.3–62	5
Alkalinity ($CaCO_3$)	18.5	369.1[a]	71.2–1187	14.0	280.1[b]	95.2–475	33
SO_4^{2-}	62.4	1247.1[a]	98.1–3598	30.5	609.1[b]	158–2238	34
Cl^-	13.4	268.4[a]	70.4–778	7.70	154.0[b]	38.6–320	34
Na^+	35.4	708.0[a]	200.2–1680	20.3	405.4[b]	135–1136	34
K^+	0.878	17.6[a]	7.3–38	0.854	17.1[a]	2.4–39	5
Fe	0.387	7.73[a]	0.05–73.9	0.012	0.237[b]	0.05–1.22	74
Cr	0.0009	0.018[a]	0.001–0.164	0.0002	0.004[b]	0.001–0.015	53
Ni	0.001	0.028[a]	0.002–0.2	0.0006	0.013[b]	0.003–0.10	39
i-P_{diss}	0.005	0.097[a]	0.001–0.51	0.005	0.104[a]	0.005–0.43	−3
TP	0.020	0.405[a]	0.046–1.39	0.015	0.302[a]	0.032–1.51	6
N-NO_2^-	0.014	0.285[a]	0.001–1.6	0.001	0.016[b]	0.001–0.30	75
N-NO_3^-	0.225	4.53[a]	0.018–16	0.044	0877[b]	0.07–7.0	68
N-NH_4^+	0.101	2.02[a]	0.05–9.65	0.097	1.94[a]	0.04–13.8	7
BOD	3.53	70.7[a]	6.5–360	1.07	21.4[b]	5.0–83.4	66
COD	10.6	211.7[a]	21.8–1082	2.87	57.5[b]	11.2–172.5	72

Different letters represent statistical significant differences.

had an overall mean concentration 3% greater than in the influent. COD and BOD reductions suggested a large mineralization of incoming organic matter. Organic matter mineralization increased the CO_2 concentration in water, which, in turn, reduced water pH range from 6.5 to 12.3 in the influent to 6.9–9.1 in the effluent. Mean calcium concentrations decreased 34% and alkalinity 33%. Removal percentages of calcium and alkalinity were greater when the pH of the incoming water was higher (9.2–12.3), suggesting that calcium carbonate precipitation within the wetland represents an important pathway governed by the pH of the incoming water.

Metal concentrations were significantly lower in the effluent than in the influent. The overall mean retention throughout the study period was 74%, 53% and 39% for Fe, Cr and Ni, respectively. The removal percentages of each metal remained almost constant during the experimental period. Consequently, the higher the incoming loads, the larger the removed metal amounts. Zn concentration was below 50 $\mu g\,l^{-1}$ (detection limit of the analytical method) both in influent and effluent throughout the study period. Simultaneous SO_4^{2-} and Fe removal and DO depletion in the water column suggest insoluble FeS formation. Because of the high SO_4^{2-} concentration in the incoming wastewater, most of the organic matter mineralization took place at the expense of biological SO_4^{2-} reduction as observed in coastal marine sediment where SO_4^{2-} reduction is responsible for 25–79% of the total organic matter mineralization (Giblin, 1988). S^{2-} released by SO_4^{2-} reduction subsequently reacts with iron to form FeS minerals (Giblin, 1988). Several monosulphide minerals early precipitated are later converted to pyrite (Giblin, 1988). FeS formation depends on the rate of Fe(II) and S^{2-} supply. The lower concentrations of Fe than SO_4^{2-} in the incoming wastewater and the almost complete Fe retention within the wetland (Table 1) suggest that Fe availability limits FeS formation in the CW.

The large-scale wetland showed a retention efficiency for metals similar to that of the small-scale prototype studied earlier (Maine et al., 2005, 2006). The incoming wastewater was different; the influent of the small-scale wetland presented greater pH, conductivity, nutrient and metal concentrations than the larger wetland, causing the earlier disappearance of the floating macrophytes. In both wetlands, vegetation development showed a similar pattern, being *T. domingensis* the dominant species after a year of operation (Hadad et al., 2006b).

4.2. Macrophyte dominance stages

E. crassipes became dominant and covered about 80% of the surface from March 2003 to January 2004, decreasing progressively until its disappearance over the following six months (Fig. 2). In September 2003 roughly 20% of the biomass was harvested. In January 2004 the wetland was emptied for a few days and the plants remained anchored in the mud, without apparent damage. *E. crassipes* decreased progressively since then to attain a very small cover in July 2004. In August 2004, the wetland was emptied again, the few remaining floating macrophytes were harvested and soil was added in strips, perpendicularly to the water circulation, where *T. domingensis* was transplanted. Once the wetland was refilled, water depth was reduced to 0.5 m and 0.3 m at the strips with added soil, respectively, in order to favour *T. domingensis* growth and the oxygenation of water. *E. crassipes* was transplanted again in August 2004, although it showed some initial growth, it soon decreased its cover. Metal concentrations in water were not sufficiently high to cause its disappearance. *E. crassipes* tolerance thresholds for pH and conductivity lay between 9–10 and 3–4 mS cm^{-1}, respectively (Hadad et al., 2006b). In many samplings, conductivity and pH were higher than the tolerance thresholds, possibly being the cause of *E. crassipes* disappearance. *T. domingensis* developed along the banks, covering roughly 4–13% of the wetland surface during 2003 (Fig. 2). After soil strips were added in August 2004 plant cover steadily increased to attain roughly 30% of the surface by the end of 2004. In October 2005 the wetland was emptied again and the organic matter remaining on the bottom was extracted. As floating macrophytes showed low tolerance to the conditions and caused DO depletion, only new specimens of *T. domingensis* were planted. *T. domingensis* covered 65% of the surface by the end of the study.

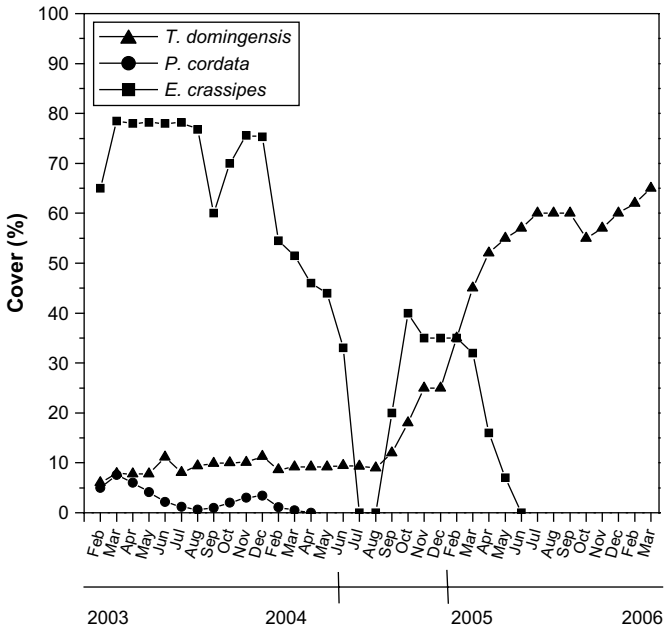

Fig. 2. Macrophyte cover in the CW throughout the study period. The line below the figure indicates the different vegetation dominance stages.

The experimental period was divided in the three successive stages described for the vegetation development: *E. crassipes* dominance stage (March 2003 to June 2004); *E. crassipes* + *T. domingensis* dominance stage (September 2004 to February 2005); and *T. domingensis* dominance stage (March 2005 to March 2006).

4.3. Contaminant concentrations in macrophyte and sediment

Large temporal variations were observed in metal and nutrient concentrations (Figs. 3 and 4) along the studied period,

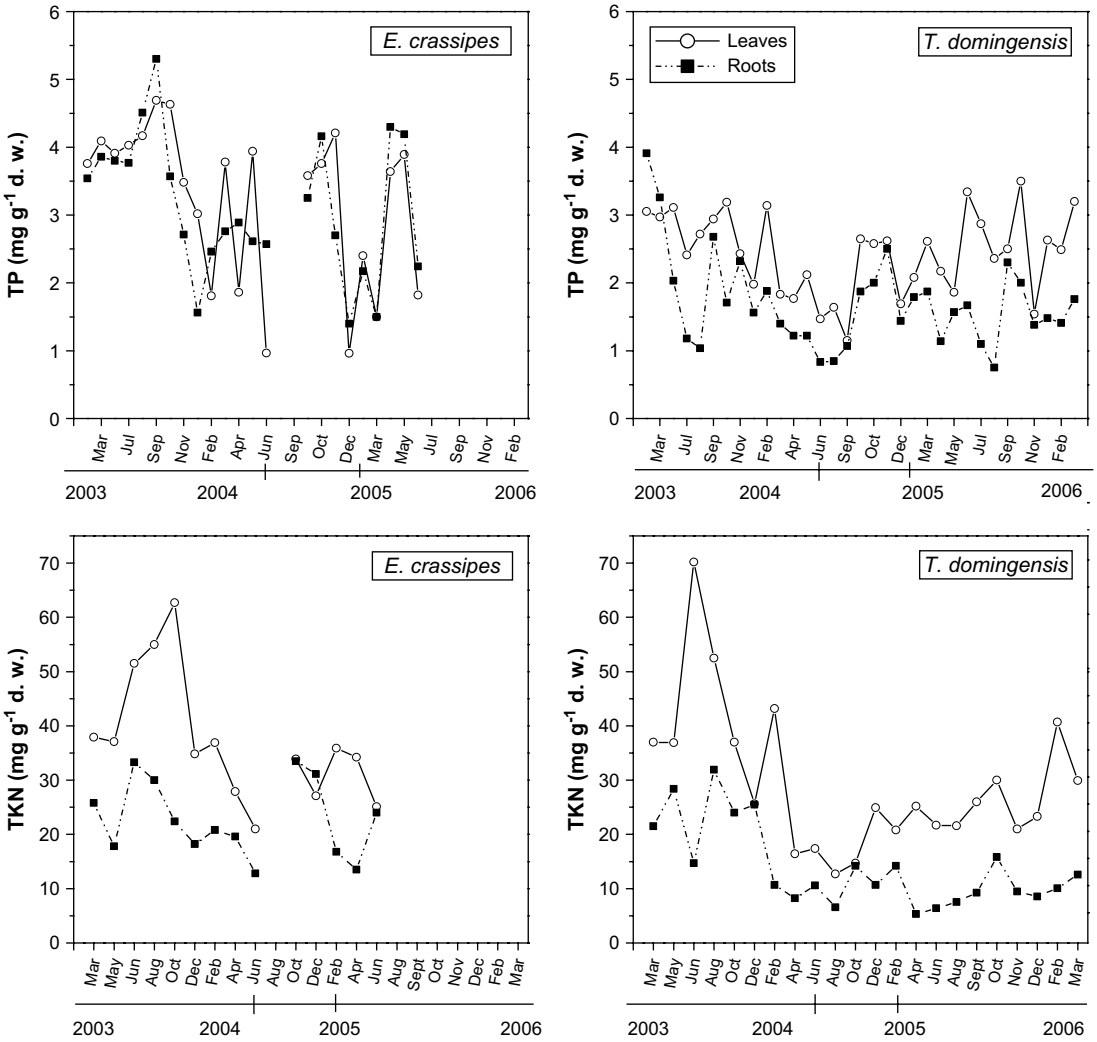

Fig. 3. Mean TP and TKN concentrations in tissues (*E. crassipes* and *T. domingensis*). The line below the figure indicates the different vegetation dominance stages.

probably due to biomass temporal variation. Metal concentration in plant tissue was higher in roots than in leaves (Fig. 4). TP, Cr, Ni and Zn concentrations in sediment (Fig. 5) did not increase significantly through the *E. crassipes* stage in spite of the fact that the concentrations between influent and effluent revealed wetland retention. *E. crassipes* was responsible for the P, Cr, Ni and Zn removal; mass balance (Table 2) corroborated these results. Co-precipitation with iron determined metal retention in sediment, but almost permanent DO depletion in the *E. crassipes* stage prevented precipitation to the bottom and metals were sorbed by macrophytes. Since June 2004 (*E. crassipes* + *T. domingensis* and *T. domingensis* dominance stages) significant increases in TP, Cr, Ni and Zn sediment concentration at the inlet area were registered. During the *E. crassipes* + *T. domingensis* stage, Cr, Ni and Zn were retained by the sediment (Table 2 and Fig. 5). During the *T. domingensis* dominance stage, Cr, Ni, Zn and P were accumulated both in sediment and macrophytes, but mainly in sediment (Table 2 and Fig. 5). Due to the fact that this species is not in a direct contact with residual water, it retains metals in a lower proportion than the free-floating species. However, emergent plants contribute to wastewater treatment processes in a number of ways, such as favoring the settlement of suspended solids, providing surface area for microorganisms, carrying oxygen from the aerial parts to the roots, creating the proper environment in the rhizosphere for the proliferation of microorganisms and promoting a variety of chemical and biochemical reactions which enhance metal retention by the sediment (Brix, 1994, 1997; Kadlec et al., 2000).

Although Zn concentration in the influent and effluent remained below detection limits in most samplings, the increase in plant tissues and in sediment concentrations during the last two stages suggested that the wetland effectively retained Zn at the later stages.

There were no significant differences between the initial and final TKN concentrations in sediment, both in the inlet and the outlet areas. Mass balance of N suggested that the biomass N pool accounted for 16–23% of the N removed from the incoming wastewater (Table 2). Given the observed DO depletion it could be proposed that denitrification is the major removal process.

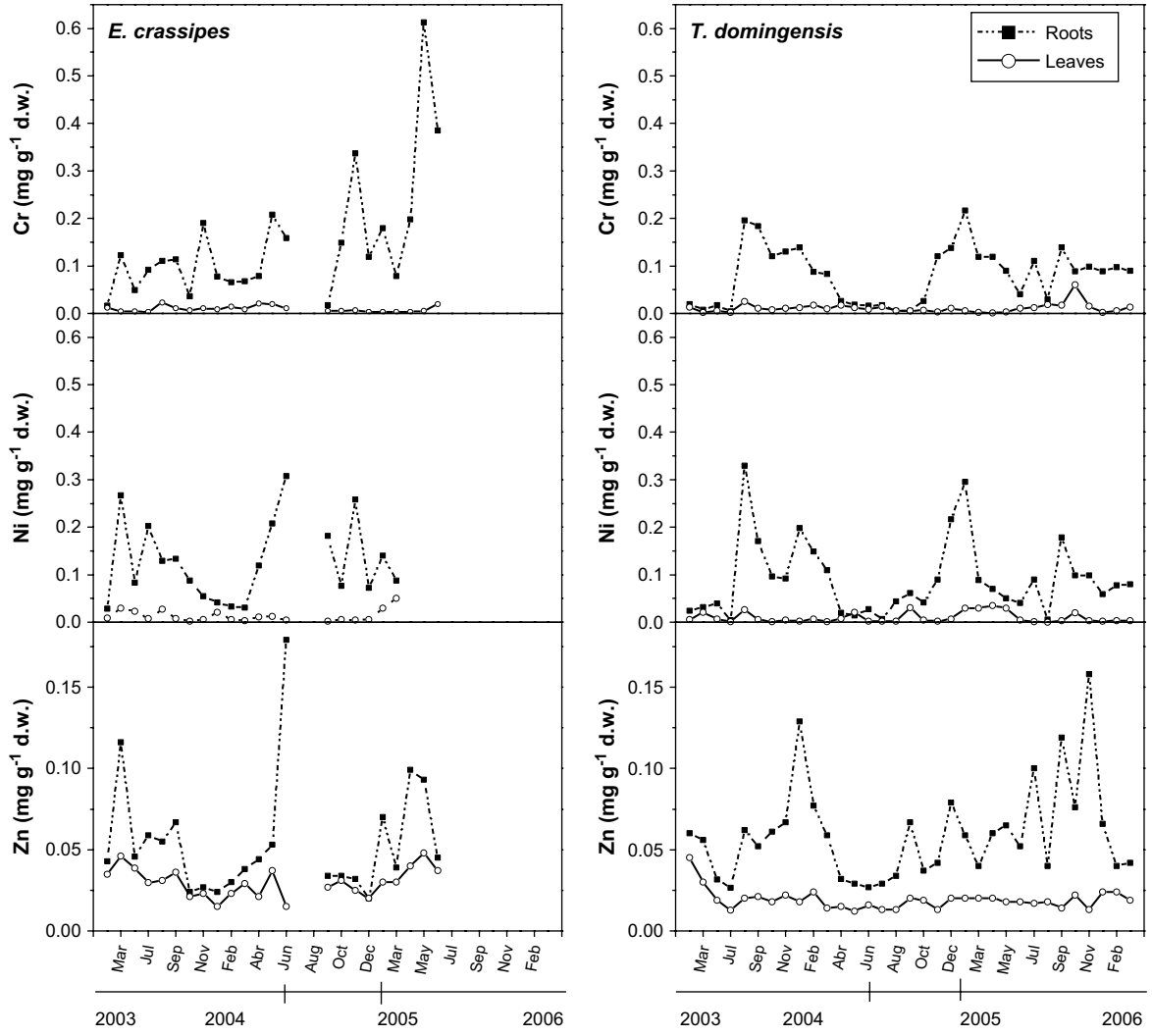

Fig. 4. Mean Cr, Ni and Zn concentrations in tissues (*E. crassipes* and *T. domingensis*). The line below the figure indicates the different vegetation dominance stages.

4.4. Comparison of the removal efficiencies during the three macrophyte dominance stages

As the retention mechanisms at the three vegetation stages were different, the removal efficiency at each stage of vegetation dominance defined from plant cover was evaluated (Table 3). It was observed that there were no significant differences in the removal efficiency of the parameters measured among the three stages, except for NH_4^+, TP and i-P_{diss}. NH_4^+ showed a different behaviour at the different phases of vegetation development. *E. crassipes* produces a large amount of detritus, which decomposes in the anoxic bottom thus reducing the redox potential in the water column. Organic matter mineralization represents a source of NH_4^+, which is not nitrified because of DO depletion and therefore NH_4^+ was often higher in the effluent than in the influent. During the *E. crassipes* + *T. domingensis* period, decreased *E. crassipes* cover likely allowed a higher oxygen contribution from the atmosphere to the water column. Besides, emergent macrophytes release oxygen from the roots, increasing aerobic degradation of organic matter and producing a strong positive effect on nitrifying bacteria in the rhizosphere (Bodelier et al., 1996; Brix, 1997). The simultaneous occurrence of partial oxidation of ammonium and denitrification account for the ammonium removal observed during the last two stages. Ammonium removal related to the increase of *T. domingensis* cover is consistent with the removal of ammonium observed in the constructed small-scale wetland in which *T. domingensis* was the dominant species (Hadad et al., 2006b).

In the first stage, i-P_{diss} concentration of the effluent had an overall mean concentration 14% higher than in the influent. The low i-P_{diss} concentrations at the inlet coincidentally with the samplings of high pH, calcium and carbonate might be caused by phosphorous co-precipitation with calcium carbonate. The analysis of P fractionation in sediment (Fig. 6) showed that the $CaCO_3 \approx$P fraction increased significantly at the inlet. The concentrations of i-P_{diss}, higher at the outlet than at the inlet, may be due to the mineralization of organic matter. As pH decreases, P sorption to carbonates decreases, while the adsorption to Fe^{3+} oxy hydroxides increases

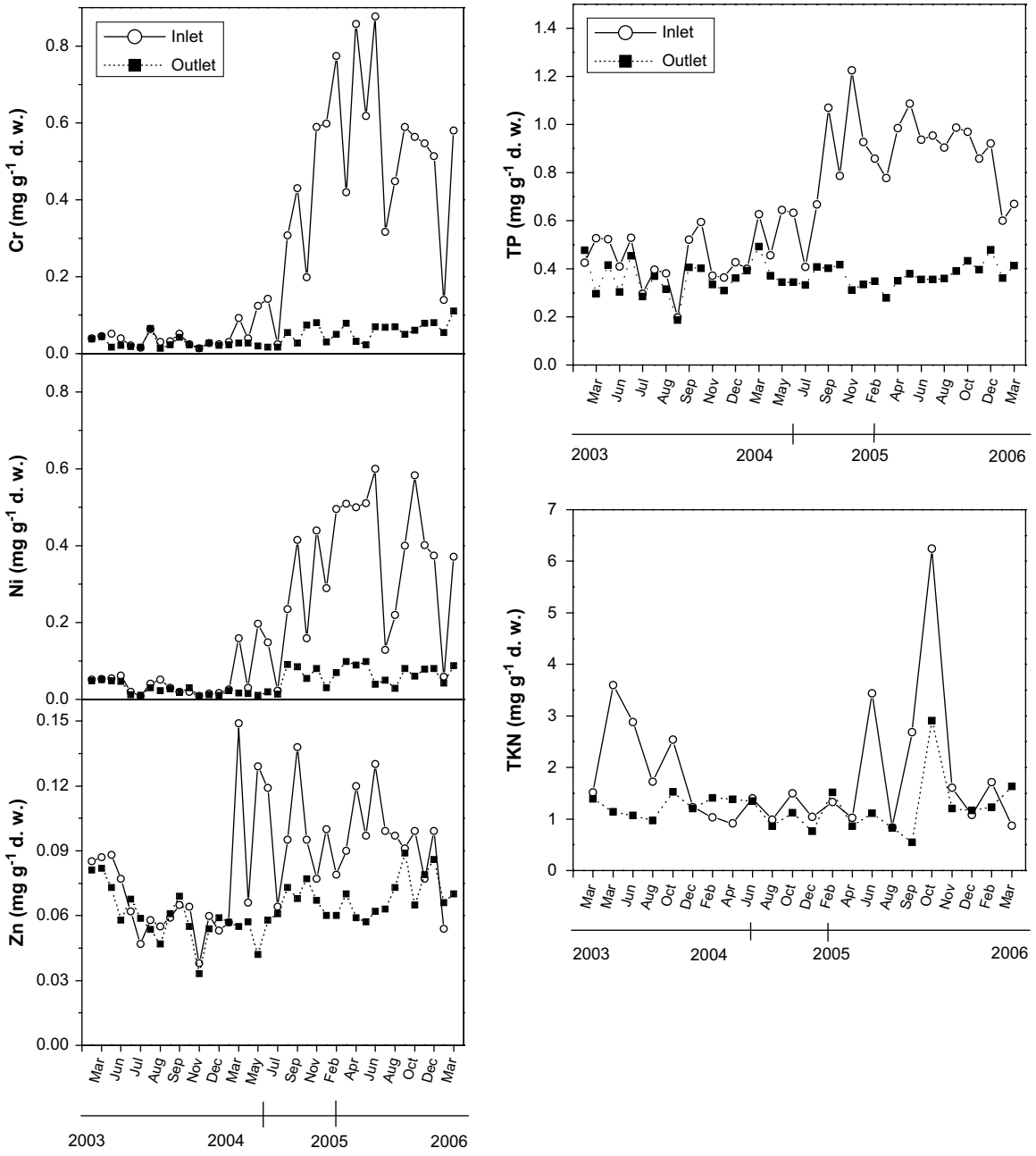

Fig. 5. Mean TP, TKN, Cr, Ni and Zn concentrations in sediment in the inlet and outlet areas. The line below the figure indicates the different vegetation dominance stages.

(Golterman, 1995). However, DO depletion prevented adsorption to iron, resulting in often higher i-P_{diss} concentrations in the effluent throughout the period of *E. crassipes* dominance. During the second and third stages, i-P_{diss} removal percentage was 15% and 22%, respectively. During these stages, i-P_{diss} continued co-precipitating with $CaCO_3$ at the wetland inlet. The increase of this fraction was remarkable in the inlet sediment (Fig. 6). At the outlet there was a significant increase of the Fe(OOH)≈P fraction at the expense of $CaCO_3$≈P fraction (Fig. 6). Therefore, it seems that $CaCO_3$≈P represents the main precipitating mechanism. However, mineralization of organic matter maintained the outlet sediment at a pH range lower than the high values prevailing in the influent. CO_3^{-2} could undergo partial dissolution and the released i-P_{diss} could be readsorbed onto the Fe(OOH)≈P fraction (Maine et al., 2007). Emergent macrophytes influence the biogeochemical cycles of the sediment through the effects on the redox status of the sediment, due to their capacity to transport oxygen from roots into the rhizosphere (Barko et al., 1991; Sorrell and Boon, 1992). Quantitatively this is easily visualized by the reddish colour associated with the oxidized forms of iron on the surface of the roots and the surrounding sediment.

A complete root–rhizome development for a newly CW may require 3–5 years. The CW performance improves with

Table 2
Distribution of the removed metals between macrophytes and sediment during the three stages (%)

Stages	Cr		Ni		Zn		P		N	
	Sediment	Macrophyte	Sediment	Macrophyte	Sediment	Macrophyte	Sediment	Macrophyte	Sediment	Macrophyte
E. crassipes	12	88	8	93	2	98	2	98	–	25
E.c. + *T.d.*	93	7	95	5	89	11	79	21	–	16
T. domingensis	70	30	87	13	59	41	62	38	–	23

wetland maturity (Kadlec et al., 2000). Vymazal and Krópfelová (2005) reported that for *Phragmites* sp., three to four seasons are usually needed to reach maximum standing crop but in some systems it may take even longer. This CW has been dominated by *T. domingensis* for only about a year. Its performance is expected to improve with time.

Plant harvest and extraction of organic matter from the bottom is periodically carried out. These materials are used in the manufacture of compost for growing ornamental plant species in a greenhouse located on the same grounds. Metal concentration in compost and tissues of the ornamental species are analyzed on a regular basis. Concentrations are below the levels permitted by national regulations.

5. Conclusions

- The CW efficiently decreased mean concentrations and variability of the parameters analyzed in water, except for the case of i-P_{diss} and NH_4^+ due to anoxia. The regulating capacity demonstrated by the CW implies an important advantage if the primary treatment failed and there were an accidental loading of high concentrations of metals, in which case the CW would retain them.
- During *E. crassipes* dominance, contaminants were retained in the macrophyte biomass; during *E. crassipes* + *T. domingensis* stage, contaminants were retained in sediment and in the *T. domingensis* dominance stage, contaminants were retained in sediment and in the macrophyte biomass.
- Even though retention mechanisms were different, removal efficiencies did not show significant differences among the three vegetation stages. Therefore, the choice of the most suitable species to use shall depend on the tolerance of the macrophytes to the conditions of the wastewater to be treated.
- *T. domingensis* was the best-adapted species to the studied wastewater in terms of growth and propagation. This species produced high biomass which could be used after harvesting in the manufacture of compost. As a consequence, it is the best choice to treat wastewater of high pH and conductivity with heavy metals, a common result from many industrial processes. Its development was favored by regulating the level of water, attaining its best growth at approximately 0.3 m depth.
- Since the conditions for i-P_{diss} removal (high pH, Fe, Ca and ionic concentrations) are largely provided by the influent, the CW would be expected to continue retaining i-P_{diss} as far as the composition of the influent remains the same. Sediment will continue retaining metals, favored by the presence of *T. domingensis,* while there are

Table 3
Mean water composition of the influent and the effluent of the wetland and removal percents during the three vegetation stages

Stages	*E. crassipes*			*E. c.* + *T. d.*			*T. domingensis*		
Parameter	Influent	Effluent	Removal (%)	Influent	Effluent	Removal (%)	Influent	Effluent	Removal (%)
pH	8.8	7.3	–	8.8	7.7	–	9.60	8.13	–
Conductivity	2803	1372	–	2371	1664	–	3949	2352	–
DO ($mg\,l^{-1}$)	2.6	0.2	–	0.5	2.3	–	0.8	4.4	–
Ca^{2+} ($mg\,l^{-1}$)	144.1	50.4	$41^{(a)}$	154.8	92.2	$22^{(a)}$	179.8	77.1	$44^{(a)}$
Mg^{2+} ($mg\,l^{-1}$)	17.4	12.6	$11^{(a)}$	17.2	19.3	$5^{(a)}$	11.3	10.6	$10^{(a)}$
Alkalinity ($mg\,l^{-1}$)	422.4	263.2	$32^{(a)}$	325.9	305.7	$12^{(a)}$	332.5	233.6	$34^{(a)}$
SO_4^{2-} ($mg\,l^{-1}$)	904.0	423.6	$37^{(a)}$	1338.5	755.4	$25^{(a)}$	1624.0	812.3	$42^{(a)}$
NO_3^- ($mg\,l^{-1}$)	3.80	0.72	$65^{(a)}$	2.51	0.43	$86^{(a)}$	6.43	1.20	$81^{(a)}$
NO_2^- ($mg\,l^{-1}$)	0.240	0.023	$78^{(a)}$	0.098	0.008	$63^{(a)}$	0.413	0.006	$91^{(a)}$
NH_4^- ($mg\,l^{-1}$)	1.51	1.76	$-24^{(a)}$	6.65	2.41	$19^{(b)}$	2.75	2.59	$11^{(b)}$
i-P_{diss} ($mg\,l^{-1}$)	0.162	0.177	$-14^{(a)}$	0.058	0.037	$15^{(b)}$	0.042	0.032	$22^{(b)}$
TP ($mg\,l^{-1}$)	0.251	0.269	$-8^{(a)}$	0.182	0.128	$17^{(b)}$	0.429	0.321	$14^{(b)}$
Fe ($mg\,l^{-1}$)	12.2	0.305	$72^{(a)}$	0.432	0.096	$65^{(a)}$	4.76	0.206	$73^{(a)}$
Cr ($\mu g\,l^{-1}$)	12.0	2.33	$66^{(a)}$	3.6	3.0	$55^{(a)}$	32	6.0	$65^{(a)}$
Ni ($\mu g\,l^{-1}$)	13.4	7.20	$48^{(a)}$	47.0	30.4	$40^{(a)}$	44	15	$52^{(a)}$
BOD ($mg\,l^{-1}$)	45.7	12.8	$62^{(a)}$	40.5	15.1	$55^{(a)}$	111.4	33.1	$68^{(a)}$
COD ($mg\,l^{-1}$)	194.9	38.1	$68^{(a)}$	95.9	33.3	$54^{(a)}$	272.9	88.9	$70^{(a)}$

Different letters represent statistically significant differences in the removal percentages among the periods.

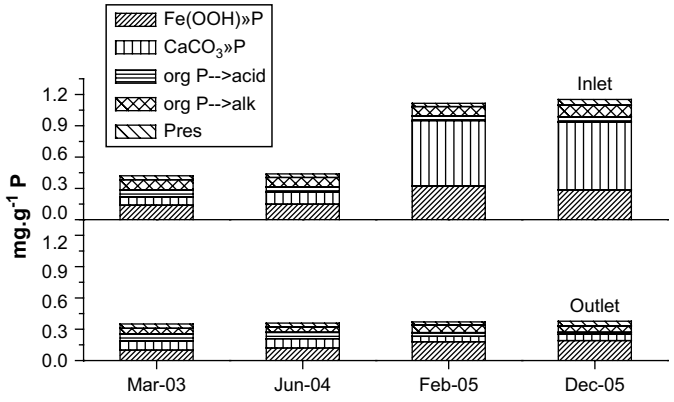

Fig. 6. P-fractions in sediment at the inlet and outlet at the different vegetation dominance stages.

available adsorption sites in the sediment. Denitrification is the principal process for N removal.

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