Effects of vegetation, season and temperature on removal of pollutants in experimental floating treatment wetlands

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Abstract

The research and interest towards the use of constructed floating wetlands for (waste)water treatment is emerging as more treatment opportunities are marked out and the technique is applied more often. To evaluate the effect of a floating macrophyte mat and the influence of temperature and season on physico-chemical changes and removal, two constructed floating wetlands (CFWs), including a floating macrophyte mat, and a control, without emergent vegetation, were build. Raw domestic wastewater from a wastewater treatment plant was added on day 0. Removal of total nitrogen, NH_4-N, NO_3-N, P, COD, TOC and heavy metals (Cu, Fe, Mn, Ni, Pb and Zn) was studied during 17 batch-fed testing periods with a retention time of 11 days (February-March 2007 and August 2007-September 2008). In general the CFWs performed better than the control. Average removal efficiencies for NH_4-N, total nitrogen, P and COD were respectively 35%, 42%, 22% and 53% for the CFWs, and 3%, 15%, 6% and 33% for the control. The pH was significantly lower in the CFWs (7.08 ± 0.21) than in the control (7.48 ± 0.26) after 11 days. The removal efficiencies of NH_4-N, total nitrogen and COD were significantly higher in the CFWs as the presence of the floating macrophyte mat influenced positively their removal. Total nitrogen, NH_4-N and P-removal was significantly influenced by temperature with the highest removal between 5 and 15°C. At lower and higher temperatures, removal relapsed. In general, temperature seemed to be the steering factor rather than season. The presence of the floating macrophyte mat restrained the increase of the water temperature when air temperature was >15°C. Although the mat hampered oxygen diffusion from the air towards the water column, the redox potential measured in the rootmat was higher than the value obtained in the control at the same depth, indicating that the release of oxygen from the roots could stimulate oxygen consuming reactions within the root mat and root oxygen release was higher than oxygen diffusion from the air.

Keywords: wastewater treatment, combined sewer overflow, nitrogen, phosphorous, COD, heavy metals

INTRODUCTION

Constructed floating treatment wetlands (CFWs) form the link between pond systems and conventional substrate based systems. Similar as in constructed wetland systems, emergent plants are used, yet the presence of a free water compartment is in common with pond systems. In contrast with classical constructed wetlands, such as surface and subsurface flow constructed wetlands, the vegetation is not rooted in a substrate or soil, but grows in a matrix floating on the water surface. The treatment potential of this technology has been already evaluated for different wastewater streams (Revitt et al., 1997; Smith and Kalin, 2000, Hubbard et al., 2004, Todd et al., 2003) and CFWs seem to be a valuable alternative when dealing with systems that are subject to high and/or highly variable water levels as drowning of the vegetation is prevented.

In recent years the main mechanisms for removal of organic matter, nitrogen, phosphorous and heavy metals in constructed wetlands have been clearly identified (Kadlec & Knight, 1996; Sundaravadivel and Vigneswaran, 2001; Vymazal, 2007). The effect of the presence of vegetation on the performance of constructed wetlands has been evaluated by different authors, pointing out their added value in the removal of various pollutants (Tanner et al., 1995; Allen et al., 2002; Riley et al., 2005; Akratos and Tsihrintzis, 2007; Iamchaturapatr et al., 2007; Headley and Tanner, 2008). Plants are able to modify the wetland environment by rhizosphere oxidation and the excretion of H^+, organic acids and CO_2 (Armstrong et al., 1990, Tanner et al., 1995, Coleman et al., 2001). Next to the effect of plant uptake, the presence of the vegetation results in more stable year-round temperatures which, in turn, may promote enhanced pollutant removal in constructed wetlands (Hill and Payton, 2000). However, the knowledge of the effect of the vegetation in CFWs on removal performance is still very limited. Some have investigated for CFWs the effect of different plant species on the removal of pollutants in CFWs and the effect of their presence compared to a system without vegetation. However, studies comprising a control without floating macrophyte mat were mostly aiming at evaluating algae removal, rather than pollutant removal (Nakamura and Shimatani, 1997, Oshima et al. 2001). A significant contribution to the removal of Cu from stormwater by the presence of a floating mat was found by Headley and Tanner (2006, 2008). Also the removal of N was positively affected by the presence of the vegetation in CFWs (Oshima et al., 2001).

The influence of temperature and season on the treatment performance of constructed wetlands, however, remains less clear for substrate-based wetlands (Stein and Hook, 2005) and very little research has been conducted towards those influencing factors for CFWs. A study of the removal of ortho-phosphate from a reservoir used for drinking-water production and supplied with floating macrophyte mats showed highly variable removal efficiencies throughout the year (0-99.6%) with lower efficiencies during the period October-December and higher efficiencies during April-August (Garbett, 2005). Similarly, Nakamura and Shimatani (1997) found that with CFWs (*Typha* spp. and *Scirpus triangulates*) used for the treatment of eutrophic lake water, a good removal could be obtained during summer, but not in winter. Also Wong (2000) observed seasonal variations at the study site of Witches Oak Waters where floating *Phragmites*-wetlands were used. However, all the references cited above concerned water with low nutrient concentrations, information on the influence of season and temperature for higher loaded wastewater was not found.

In general, wetlands are affected by solar radiation and air temperature which both cycle on a daily and annual basis, affecting plant activity and microbial processes (Kadlec, 1999). The removal of pollutants from the water column is the end result of a combination of reactions, all of which are possibly influenced by vegetation, temperature and season or a combination of these parameters. Because of interfering reactions, the effect of each individual variable is not always readily distinguishable and the net effect of these reactions may be counter intuitive (Kadlec and Reddy, 2001; Stein and Hook, 2005). Kadlec and Reddy (2001) indicated that the effect of the vegetation on the treatment performance could not be based on temperature only, as temperature is not unique to season. The effect of plant growth on the removal of pollutants may vary over the different seasons with enhanced plant growth in spring followed by senescence and plant decay in fall and winter.

Nitrogen removal is more strongly influenced by season and temperature compared to removal of P because it is mainly effectuated by microbial activity. For P, the physico-chemical process of sorption to the sediment is of main importance (Spieles and Mitsch, 2000). Picard et al. (2005) described that there was less variance in seasonal phosphorous removal when compared to nitrogen. This difference in variance is demonstrated in many treatment wetlands and may be due to the year-round sedimentary and substrate binding of phosphorous (Kadlec and Knight, 1996; Wittgren and Maehlum, 1997).

In this article results are reported from a small-scale experiment investigating the removal of nitrogen, phosphorous, organic material and heavy metals (Cu, Fe, Mn, Ni, Pb and Zn) in constructed floating wetlands. Samples were gathered during the period February-March 2007 and August 2007- September 2008. The study aimed at the determination of factors influencing the removal of pollutants in constructed floating treatment wetlands. It was presumed that the presence of the floating macrophyte mats influenced both the physico-chemical conditions and removal performance when comparing with a system without floating macrophyte mat. Furthermore, the effect of temperature was evaluated as removal processes can be influenced by temperature. Because plant growth, plant uptake and indirect plant influences may differ over the different seasons, also the evaluation of seasonal changes in removal performance was incorporated.

MATERIAL AND METHODS

Experimental setup

Three outdoor test installations of 1.44 m² (length: 1.5m; width: 0.8m; height: 1.2m; water level: 0.9m) were installed at the site of a wastewater treatment plant in Drongen (Belgium). They consisted of 18 mm plywood panels sealed with a liner. Two were covered with a floating macrophyte mat, mimicking constructed floating wetlands. The third one served as a control as the water surface was uncovered. The floating mat consisted of two plastic pipes filled with foam to enhance its buoyancy and covered with rough-messed wire netting. On top of the wire netting, a coconut coir was present in which the vegetation was rooted. The dominant plant species were *Carex* spp. (> 95% of the surface area) (Luc Mertens Ltd, Loenhout, Belgium). Less than 5% of the vegetation on both mats consisted out of *Lythrum salicaria, Phragmites australis* and *Juncus effusus*. The surface area of the floating mats was 0.77 m² (0.7m x 1.1m) but due to overhanging plant material, the effective area shaded by the vegetation was about 100%. Prior to the start of the experiments, the floating macrophyte mats were grown in a greenhouse from April 2006 till January 2007 and fed with tap water, containing very low nutrient concentrations. Both floating mats were treated as replicates as their plant composition and plant and root development was comparable. From August 2007 till December 2007 and from March till September 2008 the water surface of the control was spontaneously colonized by *Lemna* spp. During winter the *Lemna* evanesced from the water column by sedimentation and went in hibernation at the bottom of the system.

The installations were batch loaded with raw domestic wastewater from the treatment plant in Drongen (Belgium). The raw wastewater was taken before the coarse grids and sand catcher, just after the screws that bring the water up to the wastewater treatment plant. After each testing period of 11 days, approximately half of the wastewater was discarded by lowering the initial level to 45 cm and fresh raw wastewater was successively added up to a total level of 0.9 m in all three installations. This way of addition management was applied to preclude (i) the addition of peak concentrations entering the treatment system of Drongen as the remaining water in the system resulted in dilution of the added water and (ii) the difference in concentrations of the raw wastewater during the addition time as the three test installations were not fed simultaneously but successively. The overall time needed for adding the water to all three installations was ± 1h.

One measurement campaign was conducted in February-March 2007, a second one ran from August 2007 till September 2008. A total of 17 batches were performed during the two measurement campaigns. Water samples were taken across the width of the systems during the first campaign, or across the width at a depth of 5 and 60 cm below the water surface during the second campaign. These samples were combined into one composite sample before further processing. Sampling was performed at day 0, 1 (only first campaign), 2, 4, 7 and 11. In between two testing periods no fresh water was supplied to the system. Average concentrations as measured on day 0 (start of the testing period) for all 17 testing periods are presented in Table 1.

Physico-chemical analysis

Water temperature was measured in situ at a depth of 5 and 60 cm below the water surface each time water samples were taken. Average daily air temperatures were obtained from www.hydronet.be, a website with validated data from the Flemish Environment Agency (VMM). The water level in the three test installations was measured each time at the moment of sampling.

Redox potential was measured with combined platina/gel reference electrodes (HI 3090 B/5; HI 9025, Hanna Instruments, Ann Arbor, USA) which were permanently installed at a depth of 5 and 60 cm below the water surface. For the CFWs, an electrode was also installed at a depth of 5 cm in the floating macrophyte mat (CFW mat). The measured value was corrected with respect to the Standard Hydrogen Electrode as a reference by adding the difference between the redox potential measured in a ZoBells solution (0.033 M $K_3Fe(CN)_6$ and 0.033 M $K_4Fe(CN)_6$ in 0.1 M KCl) and the theoretical value of +428 mV.

Conductivity of the water samples was determined immediately upon arrival at the laboratory using an LF 537 conductivity measuring unit and a Tetracon 96 conductivity cell with integrated temperature measurement (Wissenschaftlich Technischen Werkstäten, Weilheim, Germany). Conductivity was automatically compensated for temperature with respect to a reference temperature of 25°C. For pH-measurements an Orion model 520 pH meter (Orion, Boston, MA, USA) was used. The water samples were subsequently filtered over white band filter paper (Machery-Nagel MN 640 m). Both NH_4-N and total nitrogen (Ntot) were determined following standard procedures (Eaton et al., 1995) by means of a steam distillation (Tecator Kjeltec System 1002 Distilling Unit). Part of the wastewater was filtered over 0.45µm and the remainder was acidified with H_2SO_4 (0.3 mL per 100 mL) for preservation. The 0.45 µm filtrate was analysed for nitrates, sulphates and total organic carbon (TOC). Nitrates and sulphates were determined with a Metrohm configuration consisting of a 761 Compact Ion Chromatograph equipped with a 788 IC Filtration Sample Processor and anion exchange column (IC-AN-Column Metrosep, A supp 4, Metrohm Ion Analysis, Switzerland). A Total Organic Carbon Analyser (TOC-5000, Shimadzu, Kyoto, Japan) was used for the determination of TOC. The amount of organic nitrogen (Norg) was calculated as the difference between Ntot, NO_3-N and NH_4-N.

Total phosphorus content of the water was determined by colorimetry using a Jenway 6500 spectrophotometer (Barloworld Scientific T/As Jenway, Felsted, United Kingdom) after digestion with potassiumpersulphate as described in Eaton et al. (1995). For the determination of heavy metals, 40 mL of the filtered samples with 2 mL 65% HNO_3 and 2 mL 20% H_2O_2 was heated for 20 min at 150°C after which 1 mL 20% H_2O_2 was added. This procedure was repeated after 40 min and a total destruction time of 60 min was applied. The samples were filtered and diluted with 1% HNO_3 to 50 mL followed by determination with ICP-OES (Varian Vista MPX, Varian, Palo Alto, CA) (Fe, Mn) and ICP-MS (Cu, Ni, Pb, Zn).

Chemical oxygen demand (COD) was determined using Nanocolor test kits (Machery-Nagel, Düren, Germany). The COD-analyses were performed during 9 testing periods only (5/2 – 19/2 – 5/3 – 8/10 – 12/11/2007 – 7/1 – 11/2 – 10/3/2008). A fractionation of the COD was performed for the first three periods. To that aim, 100 mL fresh wastewater was filtered over 0.45 µm and COD was determined in both the fresh wastewater and the filtrate. Distinction was made between COD_{tot} (before filtration), COD_{dissol} (after filtration) and COD_{SS} (calculated as the difference between COD_{tot} and COD_{dissol}).

Removal efficiency

Changes in water level due to evapotranspiration or precipitation were not compensated. Therefore removal efficiencies were calculated based on surface loadings:

$$\text{Removal efficiency (\%)} = \frac{C_{a,0} - C_{a,11}}{C_{a,0}} . 100 = \frac{C_{v,0} . H_{eff,0} - C_{v,11} . H_{eff,11}}{C_{v,0} . H_{eff,0}}$$

with C_a: surface loading (g m^{-2}), C_v: pollutant concentration (mg l^{-1}), 0: day 0 (addition of the water), 11: day 11 (end of the experiment) and H_{eff}: the effective water level in the system.

The presence of vegetation on the floating mats increased the water level with 3.5 cm (on average). This was determined at three different dates by lifting the mats out of the water and allowing the remaining water to seep out. The water level before and after removal of the wetland was measured. H_{eff} was calculated as follows:

$$H_{eff}\ (m) = H_{meas} - 0.035\ m$$

with H_{meas}: the measured water level in the system (m). For the control H_{eff} was equal to H_{meas}.

Effect of temperature and season

The testing periods (17 in total) over the two campaigns were grouped according to season to evaluate the effect of season on the removal of the pollutants. To evaluate the effect of temperature, four temperature categories were discerned as outlined in Table 2. As an important part of the removal occurred within the first 4 days after addition of the water, a distinction was made between the average air temperature measured over the first 4 days (T4d) and the average air temperature measured over the complete duration of the testing period (T11d).

Statistical analyses

Statistical analyses were performed by means of the statistical package SPSS 15.0 (SPSS, Chicago, USA). For examination of the normality of the distribution of the pollutant removal efficiencies all removal efficiencies (independent of measuring campaign, vegetation, temperature group and season) were grouped together and tested by means of the Kolmogorov–Smirnov test of normality. It was concluded that all removal efficiencies were normally distributed. Also the pH and water temperature matched the normal distribution. Homogeneity of variances was tested with a Levene test and showed that all variances were equal. The significance of differences in removal performance for the different pollutants and pH between the CFWs and the control was evaluated using Univariate Analyses of Variance including the factors vegetation (present or absent), temperature (T4d and T11d and this for all 4 considered temperature groups $T<5°C$; $5<T<10°C$; $10<T<15°C$ and $T>15°C$) and season (winter, spring, summer and winter). These factors were treated as fixed factors. Significant differences in removal performance between the two CFWs and the effect of rain on influent concentrations were assessed by means of Wilcoxon rank tests. Differences in water temperature at 5 and 60 cm depth were evaluated by a paired t-test. A confidence level of 5% was adopted to evaluate the significance in all different tests.

RESULTS

The buoyancy of the floating mat varied over the year. During summer time, the mat was submerged and an above standing water layer of 1-2 cm was observed. During autumn and winter the floating mat rose from the water so that the top of the floating mat, covered with litter, was approximately 2 cm above the water surface. The vegetation formed a dense root package with highly interwoven roots rather than individual roots dangling underneath the floating mat. Average depth of the root package was 12 to 15 cm at the end of the experiments whereas initial depth was only 3 to 5 cm. Shoots of *Carex spp*. had a length varying between 90 and 120cm in September 2008 with the tallest plants in the middle of the mat.

Individual evaluation of the two CFWs showed no significant difference between both test installations. Therefore, the data of both CFWs were combined for comparison with the control. An overview of the average water and air temperature during each testing period is presented in Figure 1. There was no significant difference in water temperature between the CFWs and the control when temperature was below 15°C. At higher temperatures the water in the control was significantly higher ($p = 0.005$). During one testing period (7-1-2008 – 18-1-2008) ice formation occurred. Ice was quicker formed and lasted longer in the CFWs although no significant difference in water temperature for both depths was observed during this testing period when comparing with the control. Furthermore, no significant differences in temperature at 5 cm and 60 cm depth were observed in the CFWs, ($p=0{,}354$) whereas temperature differed significantly with depth for the control ($p=0.001$), with the lower temperatures observed at 60 cm.

The redoxpotential (Eh) at day 0 was 118 ± 112 mV. The Eh-value in the water column showed a decline during the first 2 (control -5 cm; CFW -60 cm) to 4 (control-60cm; CFW -5cm) days followed by an increase (Figure 2). The Eh in the floating mat at a depth of 5 cm declined slower followed by an increase after day 6. The average redox potential measured in the mat was higher than the redox potential in the water column of the CFWs at a depth of 5 cm and was comparable to the values of the control at the same depth (Table 3).

The pH decreased during the 11-day testing period for the CFWs whereas a stable pH was observed in the control (Figure 3). The pH at the beginning of each 11-day testing period was lower for the CFWs than for the control (p=0.03), except for the first testing period (February 2007). During the first period, the pH on day 1 was the same in all three installations. Average influent pH-values for the CFWs and control were respectively 7.25 ± 0.19 and 7.49 ± 0.18. At higher temperatures, the pH of the effluent tended to be higher, compared to the effluent-pH during colder periods. However, this was not caused by a temperature effect, but related to a higher initial pH of the influent during warmer periods. The pH at the end of each testing period was significantly lower ($p< 0.001$) in the CFWs (7.08 ± 0.21) than in the control (7.48 ± 0.26) (Table 4). The difference in pH between begin and end of each testing period was only influenced by the presence of the floating mats (p=0.036); no effect of temperature or season was observed. Contrary to pH, the reduction in conductivity was significantly affected by temperature (T4d and T11d $p < 0.001$) with the lowest decrease at temperatures between 10 and 15°C for both the CFWs and the control. The average decline of conductivity was low with respectively 4.2 ± 7.3 % and 1.6 ± 4.6 % for the CFWs and the control.

Average influent characteristics were highly variable (Table 1). This is due to the cyclic nature of human activities, industrial discharges and rain events when dealing with combined sewer networks, etc. (Leitão et al., 2005). In three testing periods (3/12/2007; 1/1/2008 and 14/4/2008) water was pumped from the waste water treatment plant to the test installations during or within 24h after a heavy rain event. As both rainwater and domestic wastewater are transported within the combined sewer system, the occurrence of heavy rain events affects the concentrations present in the influent. This resulted in a significant lower conductivity (p=0.043) and Ntot concentration (p=0.018) of the influent at day 0. Influent values for conductivity and Ntot were 773 ± 81 μS cm^{-1} and 16.9 ± 3.8 mg L^{-1} for periods with rainfall, and 1102 ± 78 µS cm^{-1} and 23.1 ± 5.0 mg L^{-1} for periods without heavy rainfall. The decrease of conductivity over the 11 day periods was apparently not affected (p=0.124) by the decrease in conductivity of the incoming water after rain events. All other pollutant concentrations, except NH_4-N and NO_3-N, were also lower during periods with heavy rainfall, but the diminution was not significant. Concentrations of NH_4-N and NO_3-N were higher after rain events.

Evolution of concentrations (Ntot, NH_4-N, TP and TOC) are presented in Figure 4 for the different seasons. As the influent concentrations varied over the different testing periods, concentration results have been normalised by dividing the concentrations measured at time t (Ct) by the initial concentration of each testing period (Cin) in accordance with Headley and Tanner (2008). The difference in removal performance between the control and the CFWs was in general already apparent after 2 days (Figure 4). Overall removal efficiencies and the removal efficiencies over the different seasons and temperature groups are presented in Table 5. To evaluate the effect of vegetation, temperature and season the p-values obtained by the Univariate Analyses of Variances can be found in Table 6.

For both the control and the CFWs a net removal of Ntot was observed, except for the control during the testing period of 3/12/2007 (-4.3%) (Table 5). Except for the testing periods carried out during spring, increasing concentrations of NH_4-N were observed (Figure 4). This resulted in almost no net removal over the 11-day period for NH_4-N. In the control, for 5 out of the 17 periods, negative removal efficiencies as low as -50.9 % were obtained for NH_4-N. The removal of NH_4-N and Ntot was significantly affected by the presence of the floating mats ($p = 0.042$ and $p = 0.03$ respectively). There was no significant seasonal effect on the removal of NH_4-N ($p=0.622$) or Ntot ($p = 0.792$) for both the control and the blank as removal performances showed a large variability over the different testing periods. In contrast with NH_4-N, there was a significant effect of temperature on the removal of Ntot (T4d p=0.01, T11d p=0.03). Highest removal of Ntot for both the CFWs and the control was observed between 5 and 15°C, but the variability in removal efficiency was quite high throughout the entire year. The removal of Norg tended to vary in both the control and the CFWs with the highest removal in spring (Table 5) but none of the considered parameters had a significant influence. Due to the low concentrations of NO_3-N present in the influent (0.20 ± 0.23 mg L^{-1}), it was difficult to obtain a good evaluation of the removal of this pollutant.

The average TP-concentration in the influent was 2.16 ± 1.04 mg L^{-1} (Table 1). An initial release of TP in the control during the first 2 to 4 days was observed during all seasons and temperature periods (Figure 4). Removal of P was influenced by temperature (T11d p=0.019), with the highest removal between 5 and 15°C, but not by the presence of a floating macrophyte mat (p = 0.65). The removal of P in the CFWs showed a high variability over the different testing periods (Table 5). Although no significant effect of season was found, higher removal performance was obtained during summer and autumn for the CFWs (Table 5).

The profile of COD-removal shows the same pattern for both the CFWs and the control but the removal in the CFWs proceeded faster during the first 2 days (data not shown). Removal efficiency after 2 days was 38.2 ± 5.3% and 21.0± 7.2 % for the CFWs and control respectively. After 11 days the removal efficiency increased to 52.9 ± 11.6 and 32.6 ± 15.5 % for the CFWs and control respectively (Table 5). The decrease of COD was significantly influenced by the floating mat (p=0.003) and the interaction between vegetation and season (p=0.045). Further exploration indicated no significant seasonal influence for the control and CFWs. Increasing COD removal in the control coincided with increasing temperature although this was not significant (Table 5). This effect was less clear for the CFWs as removal performance dropped at temperatures between 5 and 10°C.

To evaluate the contribution of sedimentation to the overall COD removal, a fractionation of the COD present in the influent was carried out during the first three testing periods (Table 7). This fractionation showed no significant difference between the initial concentration of COD_{dissol} in the influent and the concentration after 11 days. An initial average concentration of 52 ± 7 (CFWs) and 45 ± 7 mg L^{-1} COD_{SS} (control) showed a decrease within the first 2 days to a level equal to the concentrations determined at day 11. Opposite to the concentration of COD_{dissol} the TOC-concentration (which was also filtered over 0.45 µm) showed a constant decrease for both the control and the CFWs with a higher removal in the CFWs (Figure 4), athough the effect of the presence of the floating mat was not significant.

Sulphate was eliminated to a higher extent in the CFWs (18.3 ± 27.5 %) than in the control (10.0 ± 26.6 %) (Table 5) but the difference was not significant (Table 6). Sulphate reduction was higher during spring and summer for the CFWs compared to the control.

The presence of the vegetation resulted in higher Fe and Mn concentrations after 2 to 4 days, although the effect was less pronounced for Mn (data not shown). The increase in Fe and Mn was followed by a decrease until the initial concentration (Mn) or lower (Fe). Iron concentrations were during all seasons and when temperature was between 5 and 15°C, higher in the CFWs than in the control. This was also the case for Mn during spring and at temperatures higher than 15°C. It was not possible to detect a significant effect of vegetation on the removal for the considered heavy metals. The influent concentrations of heavy metals were in general low (Table 1). Overall average removal efficiencies were positive although great differences could be observed for the individual periods as was reflected in the high minimum and maximum removal efficiencies (Table 5). Removal varied between -67% (Zn) and 98 % (Pb). The CFWs performed better for the removal of Cu, Mn, Ni and Zn whereas the controls showed higher removal efficiencies for the removal of Fe and Pb although this effect was for none of the considered metals significant. Both Mn and Pb were significantly influenced by season (respectively $p=0.025$ and $p=0.03$) with the highest removal during winter (Mn) and autumn (Pb).

DISCUSSION

Hogg and Wein (1988a, 1988b) studied the source of buoyancy in natural floating *Typha* wetlands and found that natural buoyancy showed a seasonal trend, mostly associated with the increase of fresh biomass. Furthermore, the release of gas during anaerobic decomposition of dead organic matter and the temperature-dependent gas solubility can affect the floating of the mats. Although artificial buoyancy was provided in this study, biomass development influenced the floating of the mats. During spring and summer, standing stock increased and caused the mats to sink a few centimetres below the water surface. During autumn and winter, senescence of the vegetation resulted in a rise of the mats. The effect of the vegetation is expected to be more important in young CFWs whereas with ageing, the contribution of the anaerobic decomposition will increase as more dead biomass accumulates on top of the floating wetland (Hogg and Wein, 1988b). Smith and Kalin (2000) reported that artificial floating *Typha* wetlands became auto-buoyant after the third growing season.

Average air temperatures were used to evaluate the effect of temperature rather than the water temperatures. This was because water temperature was measured at the moment of sampling and did not reflect the course of temperature during the day. Kadlec (1998) found that the diurnal water temperature variation was approximately 5°C for surface flow constructed wetlands. Average daily air temperature does reflect this diurnal changes. Water temperature was not higher in the CFWs during the colder periods, but did cause a cool down effect at temperatures above 15°C. This is in contrast with substrate-based systems where the presence of the vegetation acts as a buffer against colder temperatures (Brix, 1997). Tanner et al. (1995) found that the water in vegetated subsurface flow constructed wetlands was up to 1°C cooler in summer compared to an unvegetated wetland. The longer lasting ice period observed in the CFWs was probably due to a combination of reduced impact of the wind on the water surface and shading caused by the vegetation, preventing the sun from melting the ice. In full scale applications rocking of the floating wetland by the wind may hamper ice-formation and result in a longer ice-free period.

The difference in redox potential observed at 5 and 60 cm depth was due to reduced oxygen diffusion (Dusek et al., 2008). Smith and Kalin (2000) described how the presence of a floating macrophyte mat limits the diffusion of oxygen from the air to the water whereas for the control, oxygen diffusion was less influenced. Oxygen diffusion in the CFWs was hampered as the complete water surface was covered by vegetation, shielding the water surface from the wind. The presence of a *Lemna* cover in the control did not influence the redoxpotential.

A release of oxygen by the roots might explain the higher redox potential in the floating macrophyte mat (Table 3; Figure 2). The measurements indicated that the oxygen released by the roots was higher than the diffusion rate observed in the control. The oxygenation effect of the vegetation seemed to be limited in depth as the Eh measured at a depth of 60 cm was lower in the CFWs. As such there is a larger tendency towards reduction at greater depths mediated by the higher abundance of biofilm in the CFWs favouring oxygen consuming reactions. The effect of the floating macrophyte mat on the Eh was not only limited in depth but also in space as the Eh measured at 5 cm depth in the CFWs was lower compared to the Eh in the mat and comparable with the values measured in the control at a depth of 5 cm. Neither the CFWs nor the control revealed any temperature-driven or seasonal trend. As such, the growth stage of the vegetation did not influence directly the Eh-value. However, the existence of temperature-driven and seasonal changes in Eh have been proven in subsurface flow constructed wetlands by Dusek et al. (2008). They found a decreasing Eh-value with increasing temperature due to an increased microbial activity and a decreased solubility of oxygen. Lowest Eh-values were observed during June and July opposite to April and October, during which they recorded the highest values. The lower Eh-values observed in the CFWs coincide with a higher removal of SO_4^{2-}. In more reducing conditions, sulphates are converted to sulphides which precipitate (Stein and Hook, 2005).

The continuously lower pH in the CFWs was the result of plant activity as plants excrete by their roots protons and organic acids (Coleman et al., 2001). This was already seen from the second testing period onwards. Although only half of the wastewater present at the end of the 11-day testing period was removed, the difference in pH between the CFWs and the control remained stable and did not increase during the following testing period. It seems that there was an equilibrium between the excretion and consumption of pH-lowering substances. On average, the difference in pH between the CFWs and the control at the beginning of each testing period was 0.25.

Oxygen present in the water at the beginning of the testing period was rapidly consumed by various reactions, e.g. nitrification and aerobic decomposition of organic material. Organic nitrogen was converted to NH_4-N and caused during all seasons except spring an increase of NH_4-N in the control as the production of NH_4-N was faster than its removal by nitrification, sorption… . This increase due to ammonification was not observed for the CFWs. Removal of NH_4-N in the CFWs was enhanced by the presence of the vegetation and the coconut coir compared to the control. Plants can take up both NH_4^+ and NO_3^- (Kadlec and Knight, 1996) but as the influent concentrations of NO_3^- were low, uptake of NH_4^+ was preferred. Furthermore, submerged plant parts and decomposing litter provide sorption and attachment area for biofilm formation (Vymazal, 2007). Coconut fibres have proven to be useful in biofilters used for removal of NH_3 from gaseous streams by sorption (Baquerizo et al., 2009). The coconut coir, present to enhance rooting and development of the vegetation, is expected to degrade with time and will be replaced by a matrix comprised of roots, decomposing plant material and particulate matter originating from decomposed leaves and roots and adhering suspended particles. Yet in the young floating macrophyte mat, the coconut fibres enhance the removal of NH_4-N from the wastewater.

In Europe, typical removal efficiencies of NH_4-N in long-term engineered wetland systems range between 35% and 50% (Verhoeven and Meuleman, 1999; Vymazal, 2002). More specific for CFWs, NH_4-N and Ntot removal has been reported to vary between -45 and 75% for NH_4-N, and 36 and 40% for Ntot (Boutwell, 2002; Kyambadde et al., 2004; DeBusk and Hunt, 2005; Gonzalez et al., 2005). Low removal efficiencies in the current test installations for NH_4-N were attributed to reducing conditions, limiting nitrification. The obtained results during this study were remarkably lower (25.5 ± 2.9%), although the CFWs performed better than the control (2.9 ± 22.0%) (Table 5). Reducing conditions seemed to be the main factor limiting N-removal. The removal of Ntot was influenced by temperature with the highest removal between 5 and 15°C. At higher temperatures N removal was limited as higher temperatures are associated with more reducing conditions (Dusek et al., 2008). Also the removal of NH_4-N and TOC showed a decline when temperature rose above 15°C. Kuschk et al. (2003) found that nitrification was most restricted during summer. Consistent with a reduced oxygen solubility, the input of oxygen into the rhizosphere by the plants can be restricted during summer periods (Kuschk et al., 2003). In addition, an increase of microbial activity will further favour the establishment of reducing conditions. Akratos and Tsihrintzis (2007) and Kuschk et al. (2003) indicated that at temperatures below 15°C, neither the bacteria responsible for N-removal, nor the vegetation functioned properly. In our situation, reducing conditions nullified this possible positive temperature effect at temperatures above 15°C. At temperatures below 5°C it has been reported that biological processes drastically slow down or cease (Mitsch and Gosselink, 1993). Nitrification rates in wetlands were recorded to drop rapidly below 6°C (Werker et al., 2002). In contrast with studies indicating an effect of temperature on N-removal, other studies pointed out only very small differences in N-removal between warmer and colder periods (Maehlum and Stalnacke, 1999; Mander et al., 2000).

Riley et al. (2005) concluded that, as plant uptake plays only a minor role in NH_4-N removal from normal wastewater, the effect of the vegetation on the seasonal variation of ammonium removal will be minor. This is in correspondence with the results of this study, indicating no significant seasonal effect. Also for Ntot no seasonal effect was detected although other reports have demonstrated significant differences between seasons (Tanner et al., 1995; Kadlec, 1999; Del Bubba et al., 2000; Spieles and Mitsch, 2000). The removal of N by the CFWs was more influenced by temperature. Furthermore, the presence of the floating mats caused a less variable removal of Ntot and NH_4-N from the water column compared to the control. This can be due to a more stable temperature and the effect of biofilms associated with the floating macrophyte mat. Especially the latter can be of major importance in systems with highly variable water levels and systems subject to irregular loadings or batch-fed systems (e.g; stormwater treatment, combined sewer overflow treatment).

Phosphorous removal by floating wetlands was reported to vary between 6 (UK) and 83.2% (Uganda) (Kyambadde et al., 2004; Gray, 2005). Ortho-phosphate removal in a single treatment system might be highly variable. For a drinking-water basin equipped with floating *Phragmites* mats removal varied between 0 and 96% (Garbett, 2005). Also in the current study TP removal rates were highly variable. For both the control and the CFWs, removal was more influenced by temperature than by season. This could be due to the limited contribution of the vegetation to the removal of P as plants contain only a small amount of the phosphorous that enters higher P-loaded wetlands. The soil/litter compartment is accepted to be the major long-term P storage pool in traditional substrate based constructed wetlands. Furthermore, sedimentation of P associated with particles, sorption, and microbial uptake are possible removal pathways. The retention and remobilisation of phosphorus in wetlands is controlled by the interaction of redox potential, pH, Fe, Ca and Al (Vymazal, 2002). As no initial substrate was present in all testing installations, P-removal was poor and varied, especially for the CFWs. Over time a sediment layer developed in all systems, formed by the suspended particles present in the influent and, for the CFWs, decomposing plant material but this building up did not result in increased P-removal. Although the presence of the floating mats did not significantly affect the removal of P, removal pathways may differ. In addition, the release of oxygen by plant roots may increase the adsorption capacity of wetlands for P (Walhugala et al., 1987). Phosphorus release in the control can be associated with remobilisation due to reductive dissolution of Fe(III). Although lower redox potentials were present in the CFWs, thus enhancing P-release, more alternative removal pathways (binding with sediments, sorption to coconut coir, plant uptake,..) of P are present in the CFWs compared to the control.

From the COD fractionation it could be concluded that sedimentation played an important role in the removal of COD from the water column (Table 7). As sedimentation is a physical process it is only slightly positively affected by increasing temperatures (Kadlec and Reddy, 2001). Next to sedimentation also part of the suspended solids may be captured within the floating mat, including the coconut substrate and the pendulous roots (Smith and Kalin, 2000; Headley and Tanner, 2006). This evidences the importance of the presence of the floating macrophyte mat for the removal of COD from the water column and was confirmed by the significant influence of the floating mats on COD removal during this study.

In contrast with the COD-removal, which occurred mainly within the first 2 days, TOC removal continued throughout the entire testing method. Although COD can be used to characterize the amount of organic substances present in the water, there is no universal relationship between TOC and COD. TOC-measurements are independent of the oxidation state of the organic matter and other organically bound compounds are not determined. Furthermore, during COD-measurements reduced inorganic species are also being oxidised (Eaton et al., 1995). As more reducing conditions prevail in the test installation with time, it is possible that part of the COD_{dissol} originates from the oxidation of reduced species during the COD-procedure, disguising a further removal of organic substances as can be seen from the TOC-decrease. This indicates that, next to sedimentation, other removal pathways can take place. Organic components can be degraded both aerobic as well as anaerobic by bacteria attached to the roots, rhizomes and substrates (Baptista, 2003). A biofilm layer present on the roots hanging in the water column could be visually observed in the CFWs. The results obtained by Akratos and Tsihrintzis (2007) indicated that for COD the temperature-dependence was less significant because removal was mainly the result of microbial activity of both aerobic and anaerobic bacteria which function even at temperatures as low as 5°C. This corresponded with the results gathered during the present study with removal efficiencies up to 60% at temperatures below 5°C. The removal of COD in the control seemed to be more related to temperature, with increasing removal efficiencies at higher temperatures in comparison with what was seen in the CFWs.

The presence of sulphate contributes to the removal of COD and TOC. Huang et al. (2005) indicated that influent sulphate was the major component contributing to the removal of organic matter. Reduction of sulphate was in general higher in the CFWs than in the control although there was for several periods no significant difference between the control and the CFWs (Table 5). Sulphate reduction was higher during spring and summer for the CFWs, but this was not reflected in an improved removal of TOC and COD.

The removal of heavy metals by constructed floating wetlands has only been limited documented in literature. Headley and Tanner (2008) investigated the application of CFWs for the removal of Cu and Zn from stormwater and found for Cu 65-75% and 0% removal for CFWs and control respectively. Zn was removed to a smaller extent: 10-35% and less than 10% for the CFWs and control respectively. This is in line with the current study where Cu and Zn removal was lower in the control. Revitt et al. (1997) studied the removal of metals from airport runoff and found much lower removal performances for Zn (-5 - -20%) than for Cu (20-30%). Our results suggested that, at the low influent concentrations during the present study, there was no significant contribution of the floating mat.

Conclusions

Removal of pollutants was better in CFWs compared to a control without floating macrophyte mat. The presence of the vegetation was the main factor contributing to the overall removal performance although this effect was not significant for each parameter. Concentrations were significantly lower in the CFWs for NH_4-N, Ntot and COD. Also the pH was influenced, resulting in lower pH-values in the CFWs. Both the effect of season and temperature were evaluated. Removal of Ntot, NH_4-N and P was the highest in the moderate temperature range (5 - 15°C), but diminished at higher or lower temperatures. In general removal seemed to be hampered at higher ($T>15$°C) rather than lower temperatures ($T<5$°C). Seasonal changes were detected but were not important, indicating that the growth cycle of plants did only influence the performance of the CFWs to a minor extent. Measurements of redoxpotential evidenced the less reductive conditions that persisted within the floating macrophyte mat, allowing a faster removal compared to unplanted systems. However, more reducing conditions persisted in the CFWs at greater depths. Furthermore, removal of each pollutant occurred faster within the CFWs (except for Fe and Mn), proving that CFWs have an added value over pond systems and are effective when intermittent loadings occur.

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Table 1. Average characteristics of the wastewater as measured at day 0 over the 17 different testing periods (n = 54)

Parameter	Average ± SD	Min - Max
pH (-)	7.35 ± 0.22	6.88 - 7.81
Conductivity (µS cm^{-1})	1035 ± 169	626 - 1282
NH_4-N (mg L^{-1})	16.1 ± 4.9	8.23 - 25.9
NO_3-N (mg L^{-1})	0.37 ± 0.53	< 0.1 - 2.85
Norg (mg L^{-1})	4.31 ± 4.65	0 - 12.0
Ntot (mg L^{-1})	21.8 ± 5.0	11.6 - 31.9
P (mg L^{-1})	2.16 ± 1.04	1.42 - 3.22
SO_4^{2-} (mg L^{-1})	64.2 ± 15.5	20.1 - 90.1
TOC (mg L^{-1})*	27.7 ± 11.1	9.58 - 52.4
COD (mg L^{-1})**	81.3 ± 24.7	43.0 - 124
Cu (µg L^{-1})	10.0 ± 4.6	1.39 - 19.5
Fe (µg L^{-1})	454 ± 263	72.2 - 1192
Mn (µg L^{-1})	164 ± 48	88.3 - 312
Ni (µg L^{-1})	10.0 ± 4.6	2.30 - 18.4
Pb (µg L^{-1})	6.10 ± 3.78	1.19 - 19.5
Zn (µg L^{-1})	57.5 ± 35.0	15.7 - 147

* TOC: n = 28
** COD: n = 27

Table 2. Partitioning of the testing periods over the different temperature groups and seasons. A distinction was made between average air temperature during the first 4 days (T4d) and 11 days (T11d)

Temperature (°C)	T4d	T11d
<5	5/2/2007; 12/11/2007; 11/2/2008	12/11/2007; 11/2/2008
5_10	20/2/2007; 5/3/2007; 3/12/2007; 7/1/2008; 10/3/2008; 14/4/2008	5/2/2007; 20/2/2007; 5/3/2007; 3/12/2007; 7/1/2008; 10/3/2008
10_15	10/9/2007; 8/10/2007	10/9/2007; 8/10/2007; 14/4/2008; 8/9/2008
>15	13/8/2007; 26/5/2008; 16/6/2008; 30/6/2008; 18/8/2008; 8/9/2008	13/8/2007; 26/5/2008; 16/6/2008; 30/6/2008; 18/8/2008

Season	Testing period
Spring	14/4/2008; 26/5/2008; 16/6/2008
Summer	13/8/2007; 10/9/2007; 30/6/2008; 18/8/2008; 8/9/2008
Autumn	8/10/2007; 12/11/2007; 3/12/2007
Winter	7/1/2008; 11/2/2008; 10/3/2008; 5/2/2007; 20/2/2007;5/3/2007

Table 3. Average redoxpotential (mV) over the 11 days testing period at a depth of 5 and 60 cm for both the CFWs and control and within the floating macrophyte mat (CFWmat)

	Average ± SD (mV)	Min - Max (mV)
Control-5	68 ± 225	-278 - 595
Control-60	-93 ± 226	-303 - 513
CFW-5	-24 ± 145	-221 - 362
CFW-60	-122 ± 111	-236 - 227
CFWmat	72 ± 478	-162 - 501

Table 4. Average characteristics of the water after 11 days for the CFWs (n = 25) and the control (n=17) over the 17 testing periods

	CFWs		Control	
Variable	Average ± SD	Min-max	Average ± SD	Min - max
pH (-)	7.08 ± 0.21	6.76 - 7.54	7.48 ± 0.26	7.13 - 8.03
Cond (µS cm^{-1})	1017 ± 144	727 - 1284	1015 ± 135	762 - 1222
NH_4-N (mg L^{-1})	10.8 ± 7.0	3.4 - 28.6	16.5 ± 5.1	9.1 - 27.0
NO_3-N (mg L^{-1})	0.20 ± 0.23	< 0.1 - 0.83	0.08 ± 0.05	< 0.1 - 0.19
Norg (mg L^{-1})	1.6 ± 3.4	0.00 - 8.28	2.87 ± 2.73	0.00 - 5.91
Ntot (mg L^{-1})	13.1 ± 8.5	3.1 - 32.2	19.5 ± 6.6	6.0 - 31.2
TP (mg L^{-1})	1.77 ± 0.97	0.32 - 4.72	1.90 ± 0.69	0.44 - 3.86
SO_4^{2-} (mg L^{-1})	49.8 ± 16.5	9.1 - 78.4	53.7 ± 18.0	17.5 - 73.4
TOC (mg L^{-1})*	16.4 ± 8.8	5.9 - 33.9	23.0 ± 9.1	10.6 - 41.6
COD (mg L^{-1})**	46.6 ± 26.7	17.0 - 89.0	51.4 ± 27.5	33.0 - 115.0
Cu (mg L^{-1})	5.5 ± 4.9	1.6 - 22.7	8.4 ± 4.8	2.7 - 21.4
Fe (mg L^{-1})	325 ± 263	95 - 961	259 ± 134	83 - 578
Mn (mg L^{-1})	153 ± 51	85 - 241	176 ± 66	85 - 383
Ni (mg L^{-1})	6.1 ± 3.1	0.8 - 12.6	5.75 ± 2.88	2.45 - 12.9
Pb (mg L^{-1})	3.4 ± 1.8	0.1 - 7.4	4.58 ± 3.27	0.08 - 11.8
Zn (mg L^{-1})	29.7 ± 20.9	7.2 - 76.4	47.6 ± 31.8	8.8 - 130

* TOC: CFWs n = 15; control n = 10

** COD: CFWs n = 13; control n = 9

Table 5 Overall removal efficiency, seasonal and temperature-depended removal efficiency for the different parameters

		Overall removal efficiency (%)		Seasonal removal efficiency (%)				Temperature depended removal efficiency (%)			
		Average ± 1 SD	Min-max	Spring	Summer	Autumn	Winter	T <5	5<T<10	10<T<15	T>15
NH_4-N	CFWs	34.9 ± 24.5	-17.8 - 81.4	35.5	34.5	34.3	35.1	31.2	34.5	51.0	17.3
	control	2.9 ± 22.0	-50.9 - 33.4	24.7	-10.4	7.3	0.9	3.0	2.0	9.4	-1.2
Norg	CFWs	44.5 ± 45.4	-46.0 - 100.0	84.7	27.1	56.4	37.2	45.6	60.1	28.2	60.9
	control	18.0 ± 59.4	-116.2 - 100.0	61.8	39.3	-12.5	-2.2	34.6	-30.4	60.0	27.9
Ntot	CFWs	42.3 ± 27.6	0.5 - 80.3	38.4	40.2	36.9	47.0	23.7	46.8	56.8	24.3
	control	15.2 ± 23.8	-4.3 - 74.0	22.6	3.0	2.0	26.3	4.6	26.8	27.1	1.6
TP	CFWs	22.1 ± 23.6	-24.0 - 78.5	-13.0	39.0	30.7	18.4	11.8	23.3	30.1	16.8
	control	5.8 ± 10.1	-11.1 - 29.4	13.7	5.6	-2.7	3.6	4.1	4.1	2.6	14.0
SO_4^{2-}	CFWs	18.3 ± 27.6	-24.5 - 86.4	21.8	37.1	-0.4	10.8	18.0	10.7	10.6	52.1
	control	10.0 ± 26.6	-43.5 - 65.6	-17.0	27.7	8.7	7.5	13.2	6.3	7.0	21.1
TOC	CFWs	36.3 ± 24.1	-0.7 - 71.2	36.0	19.9	35	49.1	48.9	22.4	49.5	9.7
	control	12.4 ± 32.2	-56.5 - 71.1	18.4	10.2	19	32.0	7.3	13.9	17.7	10.2
COD	CFWs	52.9 ± 11.6	25.4 - 69.7	n.d.	n.d.	62.5	42.7	60.3	50.2	69.7	n.d.
	control	32.6 ± 15.5	10.0 - 53.8	n.d.	n.d.	23.3	29.3	21.6	33.1	49.0	n.d.
Cu	CFWs	52.3 ± 26.9	-17.5 - 84.9	71.6	62.3	40.4	36.9	55.4	32.1	65.7	66.3
	control	29.2 ± 25.6	-40.8 - 55.9	52.2	29.2	21.9	16.9	29.5	15.4	40.4	36.3
Fe	CFWs	24.5 ± 36.1	-47.7 - 78.8	16.7	27.7	11.8	n.d.	65.2	18.0	-9.6	50.1
	control	38.4 ± 20.9	-3.3 - 65.5	48.0	41.7	20.4	36.6	27.4	33.3	32.0	50.9
Mn	CFWs	6.1 ± 28.5	-62.7 - 57.9	-7.7	-3.2	-13.9	16.9	27.4	16.6	-30.9	-11.6
	control	-2.0 ± 19.6	-49.3 - 24.2	-2.2	-9.7	-12.6	9.9	15.1	11.8	-25.9	-4.9
Ni	CFWs	16.4 ± 34.5	-21.8 - 90.3	-11.9	30.6	5.1	13.4	15.7	11.3	2.4	23.6
	control	10.9 ± 33.3	-35.6 - 68.5	18.7	16.4	-4.9	8.1	25.3	2.7	12.2	13.8
Pb	CFWs	33.0 ± 38.0	-33.9 - 97.7	61.5	25.3	80.4	20.1	16.5	29.6	27.1	53.8

	control	38.4 ± 35.7	-28.0 - 98.1	36.2	13.8	88.1	26.6	69.6	31.3	45.9	36.2
Zn	CFWs	42.1 ± 44.6	-66.7 - 88.9	86.4	60.0	57.1	27.5	29.1	31.5	57.2	73.6
	control	18.5 ± 38.6	-44.2 - 83.1	70.1	-11.6	21.1	10.5	33.3	12.9	7.6	32.0

n.d. not determined

Table 6 p-values obtained from the Univariate Analyses of Variance for the different factors and interactions influencing the removal performance of the different pollutants during the two measuring campaigns.

	NH_4-N	NO_3-N	Norg	Ntot	TP	SO_4^{2-}	TOC	COD
Vegetation	0.042*	0.466	0.888	0.032*	0.65	0.213	0.341	0.003*
Season	0.622	0.142	0.998	0.792	0.359	0.716	0.192	0.156
T4d	0.529	0.122	0.908	0.01*	0.103	0.154		0.238
T11d	0.796	0.078	0.706	0.03*	0.019*	0.468	0.334	0.625
Vegetation x season	0.541	0.666	0.365	0.709	0.283	0.301	0.53	0.045*
Vegetation x T4d	0.521	0.425	0.523	0.736	0.833	0.896		0.199
Vegetation x T11d	0.956	0.072	0.357	0.942	0.76	0.982		0.129
	Cu	Fe	Mn	Ni	Pb	Zn		
Vegetation	0.291	0.991	0.156	0.799	0.775	0.484		
Season	0.99	0.835	0.025*	0.943	0.03*	0.617		
T4d	0.484	0.203	0.691	0.85	0.272	0.024*		
T11d	0.766	0.607	0.967	0.989	0.965	0.065		
Vegetation x season	0.613	0.929	0.312	0.763	0.926	0.85		
Vegetation x T4d	0.977	0.375	0.062	0.952	0.436	0.147		
Vegetation x T11d	0.636	0.595	0.847	0.611	0.708	0.623		

* $p < 0.05$

Table 7 COD fractionation (mg L^{-1}) for 3 testing periods at the start and end of the 11 day testing periods

Testing period		Start (mg L^{-1})			End (mg L^{-1})		
		COD_{tot}	COD_{dissol}	COD_{SS}	COD_{tot}	COD_{dissol}	COD_{SS}
1	CFWs	106 ± 6	45 ± 4	61 ± 1	62 ± 10	46 ± 5	15 ± 6
	control	118 ± 0	40 ± 1	56 ± 10	79 ± 1	36 ± 4	21 ± 6
2	CFWs	101 ± 12	51 ± 4	86 ± 3	50 ± 13	67 ± 3	19 ± 1
	control	92 ± 2	50 ± 10	81 ± 12	42 ± 8	68 ± 9	14 ± 19
3	CFWs	114 ± 13	59 ± 3	71 ± 6	44 ± 15	53 ± 5	18 ± 8
	control	113 ± 3	46 ± 1	115 ± 11	67 ± 1	44 ± 3	71 ± 14
Average	CFWs	107 ± 11	52 ± 7	73 ± 12	56 ± 13	55 ± 10	17 ± 6
	control	108 ± 13	45 ± 7	84 ± 28	62 ± 17	49 ± 16	35 ± 30

line figure
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Figure 1. Average air and water temperature for the control and the CFWs during the different testing periods.

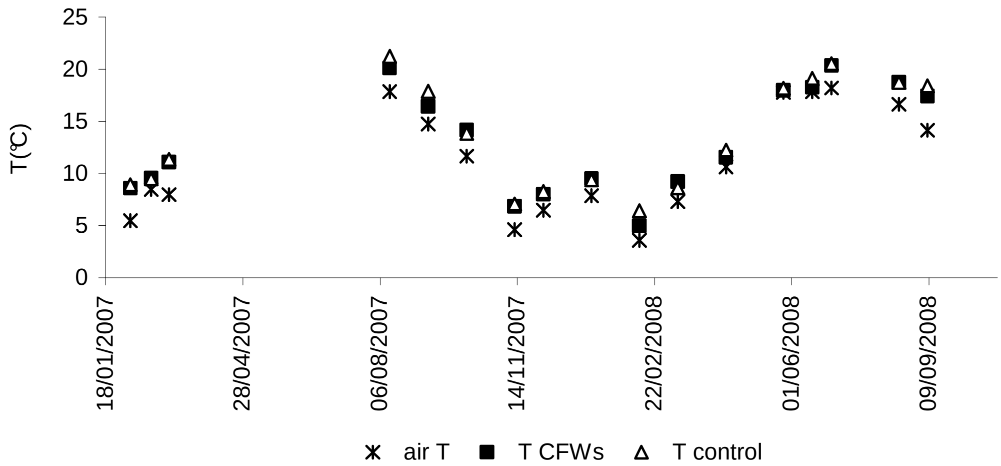

Figure 2. Evolution of the redoxpotential as a function of time over the 17 testing periods

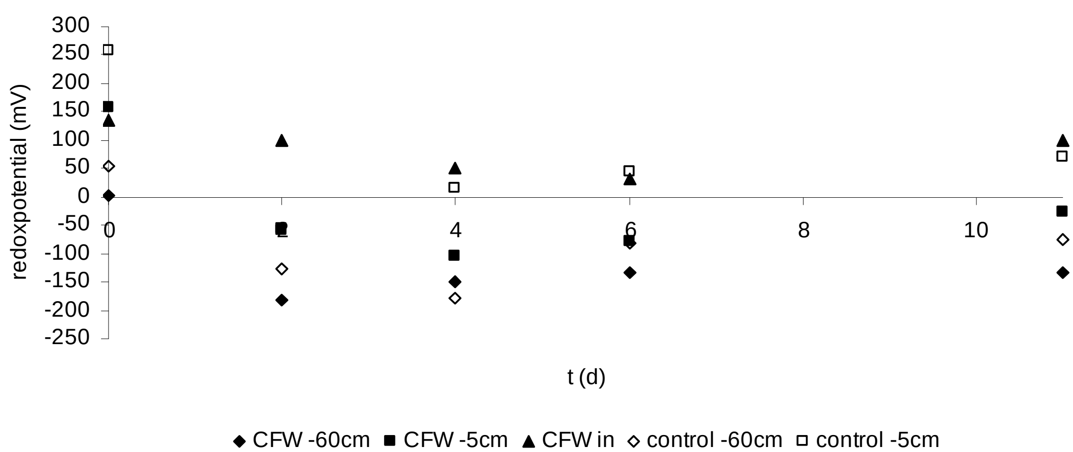

Figure 3. Average pH-values for the CFWs and the control over the 17 testing periods

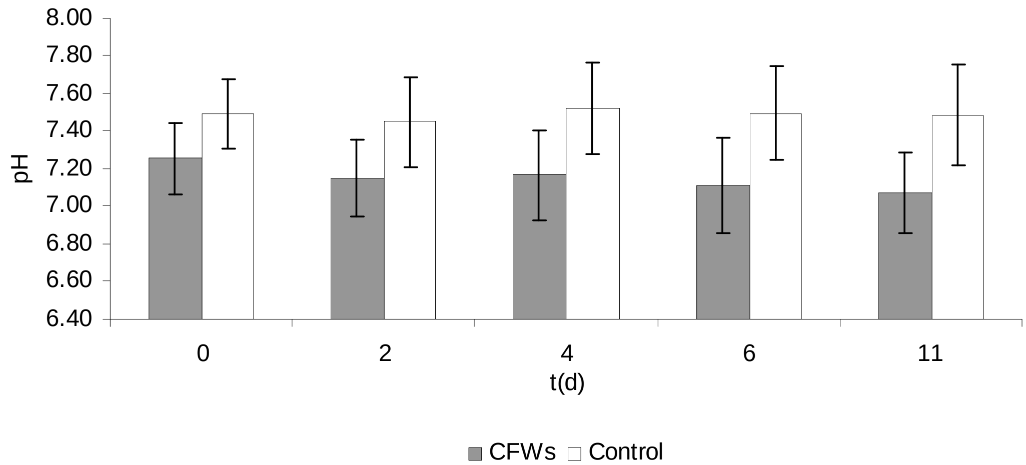

Figure 4. Evolution of concentrations for different selected pollutants for the CFWs (black symbols) and the control (white symbols) over the different seasons. The figures depict the proportion of the concentration that remains in the water at time *t* (Ct) since the start of the batch (Cin).