Mobility and sequestration of trace elements
Cr (VI) and COD removal in polyculture constructed wetlands at pilot scale treating landfill leachate (O.18)


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INTRODUCTION

Sanitary landfills are still the most widely used method for solid waste disposal around the world, and they release a wide range of chemical compounds due to waste degradation along their entire life cycle. Landfill leachate (LL) is recognized as one of the most critical issues for landfill operators. LL may contain large amounts of organic matter, both biodegradable and refractory, as well as considerable ammonia-nitrogen, heavy metals, chlorinated organics and inorganic salts. The discharge of untreated LL into surface and ground waters is a common problem in many developing countries. Therefore, there is a clear need for cost-effective and reliable technologies for LL treatment. Thus, Constructed Wetlands (CW), have been recently reported to having a high potential in this respect. However, experiences up to now are mostly limited to developed countries with seasonal or temperate climates and using mainly cosmopolitan plants. Therefore, the aim of this research was to study the performance of COD and Cr (VI) removal from LL using pilot-scale Sub Surface Constructed Wetlands (SSCW) planted with polyculture varieties of the tropical native plants *Gynerium sagittatum* (Gs), *Colocasia esculenta* (Ce) and *Heliconia psittacorum* (He).

MATERIALS AND METHODS

The experiment was carried out during six month in the Presidente regional landfill (3º56’01.54” N y 76º26’26.05”O) at San Pedro village, in southwest Colombia. Four sub-surface CW rectangular tanks units (7.80 x 2.30 x 0.60 m in length, width and depth, respectively), were fitted and run in parallel. Each tank was filled out to a depth of 0.50 m with gravel (ϕ= 25 mm and porosity (η) = 40%). In order to distribute the plant species within the CW units, the bioreactors were divided into three equal sections, each one 2.6 x 2.3 m in length and width, respectively. At each section of one CW unit, 36 healthy cuttings (0,10-0,15 m height) of one single species were placed in a chosen order. Meanwhile, in the other CW unit, 36 cuttings of each species were randomly planted throughout the whole length. Both, experimental units and plants allocation in the setting were randomly done. The final distribution of plant species in the bioreactors were: CW1 (He-Ce-Gs); CW2 (randomly), CW3 (Ce-Gs-He), CW4 (Gs-He-Ce). The initial plant density at each CW was 6 plants m⁻². The CW were daily fed by gravity under continuous regime (24 hr d⁻¹) with a water inflow of 0,5 m³ d⁻¹ each, and the theoretical HRT was set at 7 d. All SSCW received the effluent from a high-rate anaerobic pond (BLAAT®). The influent and effluent from each SSCW were analyzed for COD, DTOC, and TKN weekly, Cr (VI) fortnightly and BOD₅ monthly according with APHA (2005). Temperature, pH, ORP, EC and DO were measured once a week.
RESULTS AND DISCUSSION

Table 1 shows the average figures of parameters monitored during the study. The pH in the inlet and outlet was alkaline, but no differences between units were observed. High pH values are a key factor for the dissolution of some mineral elements in the liquid-solid interphase, and also contribute to the precipitation of heavy metals (Castilhos et al., 2003). The temperature ranged between 26 and 27 °C, keeping mesophilic conditions for the development of biological processes in the liquid-solid matrix. DO values in the CW effluents were a least 3-fold higher when compared to the influent. This suggests the plants influence on the oxygen balance in the water column. Meanwhile, EC did not show significant differences between units. The ORP reached negative values after the first week (figures ranging between -22 and -6 mV), thus, indicating the onset of a weak anoxic condition in the rizosphere. This was probably caused by the production of organic exudates jointly with the alternation of dark-light process that switch between photosynthesis and respiration, which also affect the DO balance. The BOD5 /COD ratio was below 0.3 and almost 80% of the COD was under soluble form, thus, indicating mostly the predominance of refractory organic matter. In relation with CODfiltered, DTOC and BOD5, the removal efficiencies were no higher, with better performances in CW4 for all parameters (50%). However, there were no differences between CWs. For mentioned parameters, all CW achieve the Colombia standard for water reuse in agriculture.

Table 1. Average figures for physico-chemical parameters monitored on the LL. (* N=26; ** N=11; ***N= 20; + N=10.)

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Influent</th>
<th>CW1</th>
<th>CW2</th>
<th>CW3</th>
<th>CW4</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD total (mg L⁻¹)*</td>
<td>Mean</td>
<td>681,0</td>
<td>409,9</td>
<td>441,4</td>
<td>426,2</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>187,7</td>
<td>117,2</td>
<td>70,0</td>
<td>6,2</td>
</tr>
<tr>
<td>COD filtered (mg L⁻¹)***</td>
<td>Mean</td>
<td>456,3</td>
<td>321,8</td>
<td>346,1</td>
<td>229,6</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>117,2</td>
<td>142,4</td>
<td>112,0</td>
<td>86,2</td>
</tr>
<tr>
<td>TOC dissolved (mg L⁻¹)***</td>
<td>Mean</td>
<td>253,4</td>
<td>169,3</td>
<td>189,3</td>
<td>168,0</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>81,2</td>
<td>62,5</td>
<td>85,2</td>
<td>67,3</td>
</tr>
<tr>
<td>BOD₅ (mg L⁻¹)***</td>
<td>Mean</td>
<td>151,7</td>
<td>70,0</td>
<td>63,3</td>
<td>52,5</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>94,5</td>
<td>46,7</td>
<td>63,3</td>
<td>60,0</td>
</tr>
<tr>
<td>Cr (VI) (µg L⁻¹)***</td>
<td>Mean</td>
<td>135,2</td>
<td>92,0</td>
<td>63,5</td>
<td>46,7</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>207,9</td>
<td>102,5</td>
<td>63,9</td>
<td>95,8</td>
</tr>
<tr>
<td>pH**</td>
<td>Mean</td>
<td>8,2</td>
<td>9,2</td>
<td>7,9</td>
<td>7,9</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>0,2</td>
<td>0,2</td>
<td>0,2</td>
<td>0,2</td>
</tr>
<tr>
<td>Temperature (°C)***</td>
<td>Mean</td>
<td>27,7</td>
<td>1,7</td>
<td>26,3</td>
<td>26,9</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>1,7</td>
<td>1,9</td>
<td>1,9</td>
<td>1,9</td>
</tr>
<tr>
<td>Ec (µS cm⁻¹)**</td>
<td>Mean</td>
<td>4,9</td>
<td>0,9</td>
<td>3,6</td>
<td>3,5</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>4,9</td>
<td>4,9</td>
<td>3,0</td>
<td>3,0</td>
</tr>
<tr>
<td>DO (mg L⁻¹)***</td>
<td>Mean</td>
<td>0,6</td>
<td>1,7</td>
<td>1,5</td>
<td>1,8</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>0,4</td>
<td>0,9</td>
<td>0,9</td>
<td>0,9</td>
</tr>
<tr>
<td>ORP (mV)***</td>
<td>Mean</td>
<td>-22,6</td>
<td>-10,8</td>
<td>90,2</td>
<td>93,8</td>
</tr>
<tr>
<td></td>
<td>SD</td>
<td>106,0</td>
<td>110,8</td>
<td>90,2</td>
<td>93,8</td>
</tr>
</tbody>
</table>

Regarding Cr (VI), the concentrations found in the inflow were always higher than the Colombian standard for water reuse in agriculture (100 µg L⁻1). The removal efficiency of this metal was similar in all CW with values ranging between 32 to 76%, but all effluents from CWs exhibited concentrations lower than the Colombian standard and the CW4 had the best performance with 76% of removal efficiency. The decrease of Cr (VI) in the SSW units might also be a consequence of its chemical reduction to the trivalent form that may occur under anoxic conditions, and in the presence of an electron donor, such as bivalent iron and/or sulphides and/or few other reducing compounds as pointed out by Doumett et al. (2010). The results indicate that the polyculture CW with tropical plants can be used for phytoremediation of LL with good removal potentials for COD, DTCO, BOD5, and Cr (VI). CW4 showed the best behaviour indicating that distribution of the plants within the reactor can positively influence the performance of CW.

CONCLUSIONS

This work showed that CWs planted with polyculture of tropical plants are able to treat leachate with good removal efficiencies (40 to 60%) for COD, BOD, DTCO, and Cr (VI).

REFERENCES


Treatment of Acid Mine Drainage (AMD) Using Wetlands (O.22)

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INTRODUCTION
The lack of comparable data for different types of CWs, the limited knowledge about the mechanism of heavy metals removal in the root zone of CW (Maier et al., 2007), the versatility of microbial processes and the fact that they can be influenced by variety of wetland design options (Stottmeister et al, 2003) has sparked interest among the bioengineers to characterize the different metal removal processes and to look for methods to stimulate the dissimilatory sulphate reduction (DSR).

Within this study, first results are presented about the comparison of different types of CWs (lab-scale experiments).

METHODS

Experimental design and conditions
Four laboratory-scale model wetlands (Unplanted Gravel System, Sub-Surface Flow System, Surface Flow System, Hydroponic System) were established in the green house. The CWs were metal basin (length: 100 cm, width: 15 cm, height: 35 cm). The wetlands were planted with \textit{Juncus effusus} and fed with artificial AMD, modified according to Bissinger et al., (2001) with a mean pH of 2.65 and mean inflow of 6 l/d.

Sampling
The water loss by evapotranspiration of the system was monitored and with this data the water flow rate was estimated for every day as a basis to calculate the loads of the AMD ingredients (in g/m².d). Sampling was done on monthly basis by a syringe with a 30 cm needle along the flow path and at different depths in the basin.

Analysis
Sulphate and sulphide were measured by a photometric quick test. The concentrations of metals were estimated using ICP-OES. The pH and redox potential were measured using respective electrodes.

RESULTS AND DISCUSSION
High redox potential suggests limited/absence of electron donor (root exudates) and no DSR (absence of hydrogen sulphide) but still there is removal of sulphate, iron and magnesium load.
CONCLUSIONS

The higher percentage of load removal in planted wetlands than the unplanted wetlands implies some activities near the root zone which is based on the assumption of precipitation of ions near the plant root surface due to the difference in the rate of uptake of ions and water.

ACKNOWLEDGEMENTS

I sincerely thank UFZ for providing me the opportunity to pursue my PhD and HIGRADE for the financial support.

REFERENCES

Distribution of Total Mercury in Peat Soils in Western Siberia

(O.31)

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INTRODUCTION

The behaviour of mercury in the environment depends on Hg physical-chemical properties and a wide variety of chemical compounds which can be formed in natural condition. M. Meili (1991) suggested a “Hg/Biomass” conception which supposes existence of connection between contents of the Hg and contents of biogenic elements. According to this model different natural object has different biogenic elements, with which Hg forms stable compounds. Carbon is typical for soil, surface water and sediment, Nitrogen - for biota. At transition from one natural object to another the basic element changes, changes the form of Hg bounds and occurs fractionating. Indicator of bound change is the ratio Hg/B, where B is one of biogenic elements. The purpose of the work was to analyse the Hg distribution in peat soils at oligotrophic bogs of Western Siberia with the use of "Hg/Biomass” model.

METHODS

The objects of the study were peat soils at oligotrophic bog located at the northeast part of the Great Vasyugan Mire at the field station "Vasyuganie" (IMCES SB RAS). Three ecosystems typical for oligotrophic bogs were chosen: pine-shrubs-sphagnum biogeocenosis (tall and low ryam) and open sedge-sphagnum bog. The peat deposit in the study region is 1–3 m thick. The determination of Hg contents in peat was conducted by Hg gas-analyzer RGA-11 (Mercury gas analyzer, 1990). The analyzer detection limit is 1 ng/g. The physical properties of peat soils, fraction composition of the organic matter, and fraction composition of the nitrogen were determined by common methods in laboratory of the analytical studies Siberian Institute of Peat (Inisheva et al., 2003).

RESULTS AND DISCUSSION

Distribution of Hg in profile of peat soils of different types has similar character. The maximal concentrations of Hg are observed in the upper part of the profile. The gradual reduction of Hg contents occurs in deeper layers. Hg concentrations near the mineral bottom are at the background level typical for underlying rocks of Western Siberia. Minimal concentrations are typical for low ryam and open bog, maximal ones - for tall ryam. This distribution shows presence of the geochemical barrier on the investigation territory, where occurs the accumulation of various chemical elements, including Hg which carries out from soils of the catchment area. The correlation analysis was conducted for revealing of principal factors supporting the existed distribution of Hg concentration in peat soils (see Table 1). Analyzed parameters included the botanical composition, degree of the decomposition, ash content, density of the solid phase, contents of carbon and nitrogen, and redox potential.

Hg concentrations in different horizons of peat soil were calculated basing on the "Hg/Biomass" model. It was revealed that in aerobic layers of the peat profile with oxidizing conditions the main biogenic element linking Hg is carbon, but in anaerobic layers with reducing conditions the linking biogenic element is nitrogen.

\[
H_{\text{gross}} = N_{\text{gross}} (a - b C / N); \quad (1)
\]

\[
H_{\text{gross}} = HA (a - b C / N), \quad (2)
\]
where Hg_{gross} is a gross contents of Hg in ng/g; N_{gross}, HA are gross contents of the nitrogen and humic acids in g/kg; C and N are carbon and nitrogen concentrations; a and b are coefficients of regression.

Table 1. Correlation coefficients between gross mercury contents and physical properties C, N concentration in the studied peat soils.

<table>
<thead>
<tr>
<th>Botanical composition</th>
<th>Decomposition degree</th>
<th>Ash contents</th>
<th>Redox potential</th>
<th>Density</th>
<th>Solid phase density</th>
<th>C total</th>
<th>N total</th>
<th>C/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tall ryam</td>
<td>-0.74</td>
<td>-0.74</td>
<td>-</td>
<td>0.85</td>
<td>-0.72</td>
<td>0.74</td>
<td>0.72</td>
<td>-0.78</td>
</tr>
<tr>
<td>Low ryam</td>
<td>-0.81</td>
<td>-0.56</td>
<td>-0.52</td>
<td>0.79</td>
<td>-0.96</td>
<td>0.85</td>
<td>-0.65</td>
<td>-0.50</td>
</tr>
<tr>
<td>Open fen</td>
<td>-0.79</td>
<td>-0.84</td>
<td>-0.56</td>
<td>0.98</td>
<td>-</td>
<td>0.75</td>
<td>-0.59</td>
<td>-0.68</td>
</tr>
</tbody>
</table>

The comparison of experimental data with model data obtained by equations 1 and 2 has shown that calculated method gives good approximation of experimental result for most horizons. Fig. 1 shows a comparison of experimental data for gross contents of Hg in peat soil and data obtained in accordance with Equation 2. The significant bias exists when occurs the sharp change in the botanical composition, ash content and degree of peat decomposition in profile of peat soil.

![Fig. 1. The comparison of experimental data with data got on model (equations 2) of gross contents of mercury in peat soil.](image)

**CONCLUSIONS**

The Hg content in peat depends on the physical properties of peat soils: the botanical composition, ash content, degree of the decomposition and concentrations of the N, C, and humic acids. Depth distribution of Hg is characterized by presence of two horizons revealed on the redox condition. The suggested model of transformations of Hg compounds in the profile of peat soils founded on the "Hg/Biomass" conception allows calculating the gross contents of Hg using known contents of main biogenic element. The modeled and experimental data on Hg contents in peat soils are in a good agreement. It was revealed that the most probable Hg-linking centre in peat deposit are N-containing centers in the whole profile of peat soils excluding of upper aerobic layer where C-containing centers of humic acids also can be the Hg-linking centers.

**REFERENCES**


Mercury gas analyzer of RGA-11, (1990) Technical description and operating instructions, Tomsk, KTI "Optics".

Lead in the peat cores from ridge-hollow complex in the taiga zone of West Siberia (O.55)

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INTRODUCTION

Peat cores from bogs are used as archives to reconstruct the historical records of atmospheric metal deposition. There have been many studies of metal concentrations in peat bog profiles, with much of the recent work on peat bogs as archives of atmospheric metal deposition published by many authors. Lead in particular, has received considerable attention, and there are now many independent records of atmospheric Pb deposition using peat cores from bogs [1-3].

One aim of our investigations was to study content and distribution of Pb in the cores from ridge-hollow complex. Second aim was to determine whether the Pb concentration in peat cores can help to indicate anthropogenic activity.

METHODS

The ridge-hollow complex (RHC) located in the southern taiga zone of West Siberia in northeast spurs of the Vasjugan bog. The studied area included two peat cores. One of them was formed on the ridge site (core “Ridge”) and second was formed in the hollow site (core “Hollow”). Peat samples were collected underneath by using a Russian peat corer with 10 cm intervals. Measurement of Pb and Ti were carried out by atomic emission spectrographic analysis.

RESULTS AND DISCUSSION

Lead concentration in the both profiles are not significantly different but distribution pattern of Pb in the upper layers of the peat cores are not similar (fig. 1). The more intensive peak of Pb is found in the core “Ridge” between 10 and 40 cm, whereas, the Pb concentration in the core “Hollow” are not so strong.

Figure 1. Pb and Ti concentration and ash content in the peat cores: A – core “Ridge” and B – core “Hollow”; A,% - ash content.
Lead and Ti concentration and ash content in the both core are good agreement. Taking Ti as an indicator of concentration of lithogenic-deraved aerosols supplied by rock weathering, the concentration of Pb which was supplied to the bog via atmospheric deposition of these aerosols estimated as follows Pb\textsubscript{lithogenic}=|Ti|\textsubscript{sample}×(Pb/Ti)\textsubscript{atmospheric soil dust}. The explanation of calculation is given by [2]. The results of these calculations are shown for example of peat core “Ridge” and indicate the introduction of an anthropogenic Pb component according to 14C dating just after 110±80 BP (Fig.2). The anthropogenic component of atmospheric Pb has dominated the Pb inventory in this peat core ever since. Lead concentration are clearly lower in all layers of the studied cores compared to the Pb given by [1]. So in both sections were found a peak between 170-150 cm in the core “Ridge” and 210-190 cm in the core “Hollow” which is reflected in a sharp increase of Pb, Ti and ash content. The peak is agreement with high watering in both sections.

CONCLUSIONS

Lead concentration in the two cores from RHC are not significantly different. The two peaks of high Pb concentration were found in the both cores: one of them is found in the upper layers, the second – in the middle part of both cores. The modern level of atmospheric Pb deposition on the studied area is small and it concentration in the surface peat cores corresponds to the level in the background areas.

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REFERENCES


Patterns of mercury methylation in urban artificial wetlands (O.69)  
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INTRODUCTION  
Surface-flow artificial wetlands (SFAWs) can effectively manage erosion, flooding, and pollutant loadings, but these vegetated aquatic habitats can also be sources of methylmercury (MeHg), a bioaccumulative neurotoxin of global concern produced by anoxic sulphate reducing, iron reducing, and methanogenic prokaryotes. In Canada, two important subcategories of SFAWs are wet stormwater management ponds and created habitat wetlands, which are similar in size, design, construction, and hydrology. Mercury methylation has been confirmed in temperate created wetlands (Sinclair et al. 2012) and subtropical vegetated wastewater treatment wetlands (Stamenkovic et al. 2005; Rumbold and Fink 2006), but has not been investigated in surface-flow stormwater management ponds in temperate climates. In addition, although mercury methylation has been observed to differ over fine spatial scales (Mitchell et al. 2008) this has not been explored in managed wetlands. Characterization of the extent and fine scale distribution of net MeHg production in both habitat and wet stormwater artificial wetlands is important for understanding, predicting and mitigating this issue.

METHODS  
We examined mercury methylation in three stormwater treatment wetlands and five habitat wetlands in the Greater Toronto area. Sites ranged in age from one to ten years, in size from 0.43-1.9 hectares, and in maximum sampled depth from 0-61cm. Three to five points in each wetland were identified based on plant microhabitat: shore, ecotone between terrestrial and aquatic vegetation, the middle of emergent vegetation, the edge of emergent vegetation, and the deepest accessible open water. These locations were evaluated by describing plants species, abundance and biomass, measuring standard water quality parameters, and analysis of sediment cores. Sediment total mercury concentration, methylmercury concentration, and mercury methylation rate were measured at three horizons (0-2 cm, 2-4 cm, and 4-6 cm) in each core sample using cold vapour generation or gas chromatography coupled to ICPMS (inductively coupled plasma mass spectroscopy). Our analysis focused on percent mercury present as methylmercury (%-MeHg), a measure which integrates net MeHg production and accumulation over time.

RESULTS AND DISCUSSION  
%-MeHg in stormwater wetlands was significantly lower and less variable than in habitat wetlands (1.05 ng/g dw ± 0.59 vs. 4.14 ng/g dw ± 3.58) despite higher average tHg contamination (80.24 ± 40.36 ng/g dw vs. 40.61 ± 14.56 ng/g dw). A negative though nonsignificant relationship was observed between plant biomass in a 30 cm radius of the coring site and %-MeHg. Although other authors have observed a positive relationship between %-MeHg and plant biomass (Windham-Myers et al. 2006), attributing the effect to stimulation of microbial activity through release of labile organic carbon, artificial wetlands contain fewer topographic microsites and different soil microfloras (Faulwetter et al. 2009) than the natural wetlands where previous work was carried out. Radial oxygen loss from the dense vegetation in the stormwater wetlands (Table 1) may oxygenate the sediments and reduce the activity of mercury methylating bacteria, while the lower availability of microsites...
may reduce the anoxic microhabitats for mercury methylating bacteria in the organic carbon rich root zone and slow the oxic regeneration of their inorganic substrates. Other explanations for the disparity in %-MeHg include older age of stormwater wetlands, cooler temperatures at time of sampling, and lower mercury bioavailability.

**Table 1. Descriptive parameters and mercury burdens of eight wet surface-flow stormwater wetlands and habitat wetlands in the Greater Toronto Area**

<table>
<thead>
<tr>
<th>Wetland</th>
<th>Age</th>
<th>Size (ha)</th>
<th>Average Depth (m)</th>
<th>Sampling Depth (cm)</th>
<th>Temp (°C)</th>
<th>Av [MeHg] ± st.dev. (ng/g dw)</th>
<th>Av [tHg] ± st.dev. (ng/g dw)</th>
<th>Av %-MeHg ± st.dev.</th>
<th>Vegetation biomass (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Storm#1</td>
<td>11</td>
<td>0.5</td>
<td>0.4</td>
<td>1.5</td>
<td>12.25</td>
<td>0.83 ± 0.04</td>
<td>61.1 ± 11.9</td>
<td>1.4 ± 0.3</td>
<td>326.6</td>
</tr>
<tr>
<td>Storm#2</td>
<td>10</td>
<td>0.5</td>
<td>0.4</td>
<td>1.5</td>
<td>12.25</td>
<td>0.8 ± 0.3</td>
<td>64.5 ± 4.4</td>
<td>1.3 ± 0.6</td>
<td>186.0</td>
</tr>
<tr>
<td>Storm#3</td>
<td>10</td>
<td>3</td>
<td>0.5-3m</td>
<td>11.5</td>
<td>16.96</td>
<td>0.59 ± 0.46</td>
<td>124.2 ± 50.7</td>
<td>0.5 ± 0.4</td>
<td>68.4</td>
</tr>
<tr>
<td>Habitat#1</td>
<td>8</td>
<td>0.63</td>
<td>1.4</td>
<td>18</td>
<td>24.8</td>
<td>0.3 ± 0.2</td>
<td>19.3 ± 7.5</td>
<td>1.2 ± 0.5</td>
<td>40.3</td>
</tr>
<tr>
<td>Habitat#2</td>
<td>5</td>
<td>3.38</td>
<td>1</td>
<td>81</td>
<td>23.24</td>
<td>1.1 ± 0.01</td>
<td>41.6 ± 7.6</td>
<td>2.6 ± 0.5</td>
<td>24.3</td>
</tr>
<tr>
<td>Habitat#3</td>
<td>5</td>
<td>2.2</td>
<td>0.5</td>
<td>49</td>
<td>21.6</td>
<td>2.2 ± 1.7</td>
<td>46.5 ± 19.8</td>
<td>4.2 ± 2.9</td>
<td>6.3</td>
</tr>
<tr>
<td>Habitat#4</td>
<td>2</td>
<td>0.43</td>
<td>0.4</td>
<td>50</td>
<td>26.00</td>
<td>3.9 ± 2.9</td>
<td>48.6 ± 3.8</td>
<td>8.1 ± 6.4</td>
<td>14.0</td>
</tr>
<tr>
<td>Habitat#5</td>
<td>1</td>
<td>0.5</td>
<td>0.4</td>
<td>57.7</td>
<td>28.7</td>
<td>2.1 ± 0.4</td>
<td>47.1 ± 8.1</td>
<td>4.5 ± 0.5</td>
<td>6.2</td>
</tr>
</tbody>
</table>

**CONCLUSIONS**

Although net mercury methylation was observed in all stormwater treatment wetlands surveyed, values were lower in these sites than in wetlands managed for habitat creation, an effect which may be attributable to higher plant biomass and simpler sediment structure. Ongoing analysis will assess the variability and fine spatial location of mercury methylation in these sites. More work is needed on the mercury biogeochemistry of artificial vs. natural wetlands, particularly in the context of vegetation-MeHg dynamics.

**ACKNOWLEDGEMENTS**

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**REFERENCES**


Recent Progress in Passive Biological Treatment of Selenium (O.85)

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INTRODUCTION

Passive biological treatment systems rely on naturally occurring biological, chemical, and physical processes to achieve treatment (CH2MHILL, 2010). Typically more land-intensive than active treatment systems, passive biological systems can be less expensive to operate and manage because of lower or negligible energy or chemical inputs (Ziemkiewicz et al. 2003). In passive treatment systems, oxidized forms of selenium (selenite and selenate) can be reduced to selenite, elemental selenium, and selenides through microbial reduction, followed by sequestration in soil and sediments (Gusek et al. 2009). Labile organic carbon released from an organic substrate serves as an electron donor. Common electron acceptors that must be removed prior to selenium include dissolved oxygen (DO) and nitrate. Termed biochemical reactors (BCRs), these passive systems have been employed previously for treatment of a variety of mine-impacted waters (e.g., ITRC, 2008) and in a variety of forms but their incorporation into passive treatment systems for selenium reduction is relatively new and in a variety of forms (e.g., Lundquist et al., 1994; Zhang and Frankenberger, 2003). Substrates composed of wood chips, saw dust, mushroom compost, horse manure, field hay, yard wastes, and limestone granules have been used in varying proportions.

PROCESS OVERVIEW

The geochemistry of BCRs is relied on in a staged approach for trace metal removal. Sulfate-reducing BCRs precipitate trace metals with biogenic sulfide (i.e., chemical reduction of selenite to elemental selenium), while selenate-reducing BCRs remove selenium as elemental precipitates (CH2MHILL, 2010). Because selenium compounds are more readily reduced than sulfur compounds and reduced sulfur compounds can act as a chemical reductant, any sulfides precipitated in the BCR (e.g., acid volatile sulfides) provide additional reducing capacity within the substrate. As additional selenium-bearing water passes through this substrate, biological reduction is expected, as well as chemical reduction.

Other removal processes occurring in passive treatment systems include volatilization and adsorption. Volatilization of selenium through bacterial, fungal, or algal-mediated methylation of selenium has been shown to be a significant loss of selenium in wetlands through the conversion to organic forms such as dimethyl selenide (Hanson et al., 1998; Lin et al., 2003). Physical adsorption of selenite to iron, aluminum, or manganese oxyhydroxides present within soil or sediments and to organic matter, readily occurs in passive treatment systems (Kadlec and Wallace, 2009).

BIOCHEMICAL REACTOR BYPRODUCT CONTROL

Because the BCR is comprised of organic media, secondary parameters (e.g., biochemical oxygen demand [BOD], color, sulfide, and reduced nitrogen) are generated that require treatment before discharge. Across different projects, post-BCR treatment has included aerated and non-aerated ponds, surface flow constructed wetlands, and subsurface flow gravel beds, singly or in combinations. Frequently described as aerobic polishing cells, these treatment units function by trapping particulate organic particles, increasing the DO content of the BCR effluent, as well as oxidizing chemical oxygen demand (COD) or BOD present.
PILOT STUDY OVERVIEW

Recent advances have come through the implementation by mining companies and the US Bureau of Reclamation (Reclamation) of treatability pilot studies, full-scale systems, and additional projects discovered through professional contacts and continued review of the literature (e.g., Walker and Golder Associates, 2010; CH2M HILL, 2012). Pilot studies have indicated consistently that total selenium can be reduced to <5 μg/L and further to method detection limits. These projects have demonstrated that passive treatment is a practical, cost effective, and technologically appropriate way to manage selenium, particularly in southern Appalachia because of low influent soluble selenium concentrations (10 to 20 μg/L), and relatively higher ambient and water temperatures than cold weather locations such as northern Canada. Similar cost-effective solutions are expected for selenium removal in other regions where siting and sizing constraints can be met.

Five pilot studies and two full-scale passive biological treatment projects undertaken since 2010 have demonstrated successful and sustained reduction of selenium in coal mine drainage, gravel mine seepage, reverse osmosis (RO) membrane concentrate, and contaminated groundwater. Findings indicate that selenium concentrations spanning a range of 10-1000 μg/L have been demonstrated to be treatable to 1-30 μg/L. This paper summarizes key findings of these studies and full-scale installations that show significant progress in selenium treatment using passive biochemical reactors integrated with constructed wetlands.

REFERENCES


**Phoomdi of Loktak Lake (Ramsar Site), Manipur, North-East India - Ecosystem Services** (O.105)

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**INTRODUCTION**
Loktak lake (93°46'-93°55' E; 24°22'-24°42' N), a Ramsar site, is the largest freshwater lake in North-East India. It is a floodplain wetland of Manipur river and covers an area of 287 sq.km. The lake has a catchment area of 8247 sq.km and is fed by 36 streams. Presence of floating vegetation, locally known as *phoomdi* is a unique feature of the lake. *Phoomdi* is a huge floating heterogenous mass of soil, vegetation and organic matter in different stages of decay, which floats over a vast expanse on the free water of the lake. They occur in various sizes and thicknesses, occupying almost half of the lake area. It comprises of nearly 128 plant species across 46 families. *Phoomdi* acts as natural habitat for a wide variety of fauna, viz. 81 species of birds, 25 species of reptiles, 6 species of amphibians and 22 species of mammals. *Phoomdi* produces huge amount of plant biomass which is dredged out from the lake and discarded. In the present study, the potential of *Spirodela polyrhiza* (L.) Schleiden biomass, a macrophyte that helps in initial *phoomdi* formation, was studied as an adsorbent for removal of Pb (II) and Cd (II) from wastewater environment.

**METHODS**
Batch adsorption experiments were carried out in 150 mL conical Erlenmeyer flasks containing 100 mL of Pb (II) and Cd (II) solutions. The influences of pH, adsorbent dosage, contact time, initial concentration of metal ions and temperature on biosorption of Pb (II) and Cd (II) ions from aqueous solution were studied. The equilibrium and kinetic data of biosorption studies were evaluated using isotherm models (Freundlich and Langmuir) and kinetic models (Pseudo first order and Pseudo second order). Throughout the experiments, pH varied between 2.0-6.0 for Pb (II) and 2.0-7.0 for Cd (II), contact time from 5-120 minute, adsorbent dosage from 0.05-0.25 g, temperature from 20-40°C, and initial metal concentration from 100-160 mg L⁻¹ for Pb (II) and 10-40 mg L⁻¹ for Cd (II), respectively.

**RESULTS AND DISCUSSIONS**
The adsorption of Pb (II) and Cd (II) onto *S. polyrhiza* biomass was found to be dependent on pH, adsorbent dosage, contact time, initial concentration of metal ions and temperature. The maximum adsorption capacity calculated from Langmuir isotherm was 137 mg g⁻¹ for Pb (II) and 36.0 mg g⁻¹ for Cd (II).
Table 1 Comparison of adsorption capacity of Pb (II) and Cd (II) onto *S. polyrhiza* with various adsorbents.

<table>
<thead>
<tr>
<th>Adsorbents</th>
<th>Metal ion</th>
<th>Adsorption capacity (mg g(^{-1}))</th>
<th>pH</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Eichhornia crassipes</em></td>
<td>Pb</td>
<td>26.32</td>
<td>-</td>
</tr>
<tr>
<td><em>Ceratophyllum demersum</em></td>
<td>Pb</td>
<td>44.80</td>
<td>-</td>
</tr>
<tr>
<td><em>Myriophyllum spicatum</em></td>
<td>Pb</td>
<td>46.40</td>
<td>-</td>
</tr>
<tr>
<td><em>Laminaria hyperborea</em></td>
<td>Pb</td>
<td>50.30</td>
<td>5.0</td>
</tr>
<tr>
<td><em>Ulva lactuca</em></td>
<td>Pb</td>
<td>54.30</td>
<td>5.5</td>
</tr>
<tr>
<td><em>Nizimuddinia zanardini</em></td>
<td>Pb</td>
<td>110.3</td>
<td>5.5</td>
</tr>
<tr>
<td><em>S. polyrhiza</em></td>
<td>Pb</td>
<td>137</td>
<td>4.0</td>
</tr>
<tr>
<td><em>Eichhornia crassipes</em></td>
<td>Cd</td>
<td>12.60</td>
<td>-</td>
</tr>
<tr>
<td><em>Cystoseia indica</em></td>
<td>Cd</td>
<td>19.56</td>
<td>5.5</td>
</tr>
<tr>
<td><em>Fontinalis antipyretica</em></td>
<td>Cd</td>
<td>28.00</td>
<td>5.0</td>
</tr>
<tr>
<td><em>Bifurcaria bifurcata</em></td>
<td>Cd</td>
<td>30.30</td>
<td>5.0</td>
</tr>
<tr>
<td><em>Laminaria hyperborea</em></td>
<td>Cd</td>
<td>31.30</td>
<td>5.0</td>
</tr>
<tr>
<td><em>S. polyrhiza</em></td>
<td>Cd</td>
<td>36.0</td>
<td>6.0</td>
</tr>
</tbody>
</table>

**CONCLUSIONS**

Considering the present findings, it can be concluded that *S. polyrhiza* is an effective adsorbent of Pb (II) and Cd (II). The comparison of \( q_{\text{max}} \) values between our work and other reported data from literature shows that *S. polyrhiza* is a better adsorbent for Pb (II) and Cd (II) ions compared to other adsorbents (Table 1). It gives an additional application of unused biomass available on the *phoomdi* of Loktak lake for cleaning the wastewater environment contaminated with various heavy metals.

**ACKNOWLEDGEMENTS**

Maibam Dhanaraj Meitei gratefully acknowledges University of Hyderabad for funding the scholarship through University Grant Commission, New-Delhi, India

**REFERENCES**

Hydraulic conductivity influence on metals removal (O.106)

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INTRODUCTION

A comparison between model and experimental pilot-scale Horizontal Subsurface Flow Constructed Wetland (HSFCW) located in Lecce (Apulia, South Italy) has been reported in the paper. The experiments were carried out in three constructed wetlands. Two of them were planted with two different species of macrophytes and the third was used as a control reactor. The objectives of the study are to compare hydraulic behaviour of the CWs with the trend of the model by varying the hydraulic conditions and then to evaluate the effect of the clogging and the efficiency of the different species of macrophytes in removing metals. At the beginning of the experience and after 24 months, the results show a good correlation in the hydraulic behaviour between model and physical data by modifying input parameters as a consequence of the clogging. The three HSFCWs have a similar capacity in removing Cr, Fe and Pb.

HSFCWs are widely used for removing pollutants from wastewaters the world over. Compared to conventional wastewater treatment technologies, treatment wetlands are mechanically simple and have relatively low operation and maintenance (O&M) requirements. Subsurface-flow wetlands consist of an emergent macrophyte community planted in a porous medium (usually gravel or sand), through which wastewater is passed for purification. Traditionally, wastewater treatment plants (WWTPs) based on HSFCWs separate the treatment stages. First, pre-treatment and the primary treatment are combined to eliminate solids, while subsequent stages consist of constructed wetlands and/or natural technologies. These WWTPs tend to eliminate solids with septic or Imhoff tanks, while horizontal SSF constructed wetlands are used for secondary and tertiary treatment.

Removal of metals by CWs has been studied increasingly in recent years (Ranieri et al., 2013). Otherwise hydraulic considerations have a significant role in prediction of the actual removal percentages for every contaminant. This study assesses and elaborates the hydraulic performance in the pilot-scale HSFCWs and observes trends over time. Design parameters such as aspect ratio, size of the porous media and hydraulic loading rate can improve the hydraulic behaviour of constructed wetland systems by imparting a hydraulic flow behaviour that approaches that of an ideal flow system (Ranieri et al., 2013). The experiments were conducted using tracer tests (KBr), which provided the residence time distribution (RTD). Particularity, after 24 months of operating, clogging conditions in experimental HSFCWs result in a lower hydraulic conductivity values. The objectives of this study are:

- to evaluate the hydraulic behaviour of constructed wetlands not planted and planted with different species (Phragmites australis and Typha latifolia), as a function of the hydraulic conductivity;
- to assess the correlation of the experimental RTD curves with the curve of the model, as a function on the variation of hydraulic conductivity and clogging;
- to evaluate metals removal as a function of the HRT.
Fig. 1. Constructed wetlands pilot plant at Sternatia di Lecce, Italy: (A) plan view and (B) longitudinal section.

Fig. 2. Metals versus HRT (A) in the Phragmites field; (B) in the Typha field; (C) in the Unplanted field.

REFERENCES
**Horizontal subsurface flow constructed wetlands for the removal of As from acidic contaminated water** (O.121)

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**INTRODUCTION**

Constructed wetlands have been increasingly applied for the treatment of metal-contaminated water, which is often acidic (e.g. acid mine drainage). Previous studies have shown than vertical flow wetlands with alternative media could be employed (Lizama Allende et al., 2012). Little is known about the performance of horizontal flow wetlands.

The aim of this study was to assess the effectiveness of horizontal flow constructed wetlands in the removal of arsenic (As) and iron (Fe) from acidic contaminated water, using different wetland media. The Azufre River, Northern Chile, was selected as the case study given its elevated concentrations of As and metals under acidic conditions (Ríos et al., 2011).

**METHODS**

The wetland system

Two types of wetland cells were built, using zeolite and limestone/cocopeat as the main media. Each wetland group had three replicates which were operated individually. Young *Phragmites australis* were harvested and planted five months before the experiments started.

Synthetic water resembling the As, Fe and B levels in the Azufre River was prepared using tap water and reagents (2.6±0.5 mg/L As, 30.8±6.2 mg/L B, 97.3±14.0 mg/L Fe, pH 2.0±0.2).

The system was located in a greenhouse and was pseudocontinuously dosed daily during 22 weeks. The hydraulic loading rate was 30 mm/d resulting in a dosing rate of 150 mL/hr.

**Sampling and analysis**

Water samples were filtered immediately using 0.45 μm filters and acidified to pH<2 using HNO₃. Water quality parameters were also measured. Media samples were collected after the experiments finished and were dried at 40°C until constant weight was achieved. Plants from the inlet and outlet of every cell were divided in shoots and roots, dried at 55°C and the metal concentrations were analysed by USEPA methods 3051A and 3060A. Total and dissolved metal fraction in all samples was analysed by ICP-OES and ICP-MS.

**RESULTS AND DISCUSSION**

Pollutants removal and water quality

Arsenic removal rates were 99.8% and 99.9% in limestone/cocopeat and zeolite wetlands, respectively; whereas iron removal rates were 87.3% and 96.1%. pH and Eh were the most affected parameters: limestone/cocopeat raised the pH to 6.95, while zeolite raised it to 4.1. Redox potential decreased to negative values in limestone/cocopeat wetlands (from 475 to -37 mV), whereas it was still positive in zeolite wetlands (315 mV). Thus, alkaline and reducing conditions provided in limestone/cocopeat wetlands were not sufficient to achieve better removal as in zeolite wetlands, although both performed better than in vertical flow.
Metal accumulation in media

Zeolite presented higher As and Fe levels than did limestone and cocopeat. Furthermore, these levels tended to decrease towards the outlet, and towards the bottom: arsenic fluctuated between 230 and 8 mg/kg, iron fluctuated between 11,000 and 4,500 mg/kg. This suggests that the media plays the main role in accumulating the target pollutants.

Metal accumulation in P. australis

Plants located in the inlet uptook higher As and Fe levels than did plants in the outlet, in both wetland types: Arsenic concentrations reached 130 mg/kg in zeolite and 40 mg/kg in limestone/cocopeat, whereas iron concentrations reached 10,000 and 22,000 mg/kg respectively. Roots accumulated higher As and Fe levels than did shoots, as in previous studies (Kadlec and Wallace, 2009). Despite these high concentrations, the accumulation of pollutants was < 3% of the total mass loading, thus confirming the minor role of plant uptake as a metal removal mechanism in constructed wetlands (e.g. Ye et al., 2003).

Fig. 1. Total As (left) and Fe (right) concentrations in the inflow and outflow from the system.

CONCLUSIONS

Alternative media may be used since high As and Fe removal efficiencies were observed for both wetland types. Sorption capacity of zeolite was slightly more effective than coprecipitation with iron oxides -triggered by the pH adjustment of limestone- for this case study. Metals were mainly retained in the media rather than in the plants.

REFERENCES

Partitioning Behaviour of Metal-based Engineered Nanoparticles in Wetland Soils and Sediments (O.151)

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INTRODUCTION
The increasing production and commercial use of engineered nanoparticles (NPs) will inevitably lead to their (unintentional) release into terrestrial and aquatic environments. Due to their small sizes these nanoparticles can exhibit a variety of physicochemical properties that differ significantly from their bulk counterparts, and can hence pose a potential threat to human health and ecosystems. Therefore, extended knowledge of their environmental behavior and fate is essential in order to assess possible exposure routes and improve risk assessment. Partitioning of NPs between liquid and solid phases as well as their dissolution behavior in complex environmental media can provide important information on their mobility, bioavailability and toxicity. Association of NPs with solid phase compartments will result in limited mobility and therefore render them less bioaccessible (Gimbert et al., 2007).

In this study we aim to contribute in the identification of factors that can influence the interaction of different nanoparticulate materials (CeO\textsubscript{2} and Ag NPs) with sediment and/or soil constituents, as may occur when nanoparticles are released into a wetland. Partitioning kinetics of NPs between solid and liquid phases and dissolution behavior are investigated, as well as how they are affected by soil and sediment properties.

METHODS
Sediment and soil samples were collected from different locations and air-dried at 25 °C. Additionally, the sediment samples were further dried in an oven at 65 °C. Afterwards, the different sediment and soil samples were crushed and/or grinded if necessary and passed through a 2 mm sieve (1 mm in case of sediments). Sediment and soil properties (e.g., pH, EC, CEC, %OM) were determined.

CeO\textsubscript{2} and Ag nanoparticle aqueous dispersions were purchased from PlasmaChem GmbH (Berlin, Germany). Cerium and silver standard solutions were either obtained from ChemLab NV (Zedelgem, Belgium) or from Merck KGaA (Darmstadt, Germany).

Sediment and soil suspensions were prepared with Milli-Q water (EMD Millipore Corp., MA, USA) in a 1/10 ratio (S/L) and spiked with a known amount of nanoparticles (NPs) or ions. Blank and control samples were also included in the set-up. The suspensions were covered with parafilm and left shaking on a shaking plate for 24 hours. Aliquots were taken in between at designated times. Part of these aliquots was directly digested with aqua regia prior to (pseudo) total elemental analysis via ICP-OES. The other part was subjected to gravitational settling for 10 minutes or centrifugation at different centrifugation speeds (500 and 2000 rpm) for 10 minutes. The resulting supernatant was analyzed with ICP-OES after aqua regia digestion.

In a second experiment, the impact of centrifugation speed on the amount of suspended matter in the supernatant was also studied.
RESULTS AND DISCUSSION
The sediment and soil samples differed significantly in properties. A selection of sediment and soil properties is presented in Table 1.

Table 1. Selected properties of sediment and soil samples (mean ± SD, n = 3).

<table>
<thead>
<tr>
<th>Sample</th>
<th>pH-H2O</th>
<th>EC</th>
<th>CEC</th>
<th>OM</th>
<th>CaCO3</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(±)</td>
<td>(µS cm⁻¹)</td>
<td>(meq 100 g DM⁻¹)</td>
<td>(%)</td>
<td>(%)</td>
</tr>
<tr>
<td>Sediment A</td>
<td>7,89 ± 0,06</td>
<td>1929 ± 26</td>
<td>10,72 ± 0,10</td>
<td>4,75 ± 0,10</td>
<td>14,33 ± 0,98</td>
</tr>
<tr>
<td>Sediment B</td>
<td>7,88 ± 0,02</td>
<td>864 ± 8</td>
<td>12,56 ± 0,19</td>
<td>5,08 ± 0,05</td>
<td>15,22 ± 0,50</td>
</tr>
<tr>
<td>Sediment C</td>
<td>7,56 ± 0,02</td>
<td>961 ± 14</td>
<td>9,50 ± 0,00</td>
<td>3,47 ± 0,04</td>
<td>10,10 ± 0,76</td>
</tr>
<tr>
<td>Sediment D</td>
<td>7,55 ± 0,03</td>
<td>930 ± 11</td>
<td>24,28 ± 0,19</td>
<td>9,35 ± 0,13</td>
<td>10,17 ± 0,50</td>
</tr>
<tr>
<td>Soil I</td>
<td>4,52 ± 0,01</td>
<td>113 ± 1</td>
<td>8,67 ± 0,71</td>
<td>3,68 ± 0,07</td>
<td>3,87 ± 0,09</td>
</tr>
<tr>
<td>Soil II</td>
<td>7,53 ± 0,07</td>
<td>229 ± 2</td>
<td>9,27 ± 0,25</td>
<td>3,11 ± 0,01</td>
<td>4,60 ± 0,03</td>
</tr>
<tr>
<td>Soil III</td>
<td>7,06 ± 0,01</td>
<td>170 ± 3</td>
<td>8,49 ± 0,26</td>
<td>4,61 ± 0,19</td>
<td>2,51 ± 0,25</td>
</tr>
</tbody>
</table>

The concentration of Ce and Ag in the supernatant previously spiked with either ENPs or ions significantly depends on equilibration time and centrifugation speed, as illustrated in Fig. 2. Remarkably, CeO₂ and Ag ENPs are often more mobile, i.e. more present in the supernatant, compared to their corresponding Ce (III) and Ag (I) ions. The Ce and Ag concentrations observed in supernatant differed significantly between different sediment and soil suspensions, suggesting that sediment and soil properties influence the partitioning behaviour of the ENPs. In particular, CeO₂ NPs seem to be easily sorbed to pure sand in absence of (dissolved) organic material, whereas they were found to be very mobile in presence of small amounts of (dissolved) organic material. This may have an important impact on the mobility, (bio)availability and toxicity of metallic nanoparticles released into natural and constructed wetlands.

![Total Ce concentration in supernatant](image)

Fig. 1. Cerium concentrations measured in the supernatant two hours after spiking the sediment suspensions with CeO₂ nanoparticles (NPs) and Ce ions. Supernatants are obtained by subjecting the samples to 10 minutes of gravitational settling, or 10 minutes of centrifugation (at 500 and 2000 rpm, respectively). “Control Ce ions” and “Control CeO₂ NPs” refer to solutions spiked with same amounts of ions and NPs in absence of sediment.

REFERENCES
Wetlands and water reuse
Two Years of Evaluation of Pilot-Scale Hybrid Constructed Wetlands for Natural Disinfection of Wastewater (O.29)

Florentina Zurita\textsuperscript{a}, Daniel Rojas-Bravo\textsuperscript{a} Rosa E. Lozano-Mares\textsuperscript{a}, José A. Jacinto-Trápaga\textsuperscript{a}


INTRODUCTION

Constructed wetlands (CWs) and waste stabilization ponds (WSP) are the most widely used ecological wastewater treatment systems in use in the world (Zurita et al, 2012). These technologies have proven to be effective treatment alternatives, using natural processes, for treating wastewater in small and medium communities worldwide. These systems are capable of reaching nearly 100% in parasitic eggs removal due to longer retention times in comparison to more expensive and energy-intensive conventional technologies (Shafari et al., 2012); although a one-stage system is usually not sufficient to effect pathogen reduction to safe target levels for irrigation purpose.

Therefore, in this study three two-stage hybrid systems were evaluated in order to compare their efficiency for total coliform (TCol) and \textit{Escherichia coli} removal as indicator organisms of the presence of pathogenic microorganisms.

METHODS

Description of the wetland systems

Three two-stage hybrid ecological wastewater treatment systems (HEWTS) were evaluated in duplicate. System I consisted of a horizontal flow (HF) constructed wetland (CW) followed by a stabilization pond. The constructed wetlands were continuously fed with a hydraulic retention time of 4 days. The effluent from the constructed wetlands flowed by gravity to the stabilization ponds. System II was also configured with a horizontal flow CW as a first stage which was then followed by a vertical flow (VF) CW as a second stage. The CW operated in the same way as in the system I but the effluent was collected in a tank and pumped intermittently every 2 hr on to the substrate of the vertical flow CW. System III was configured with a vertical flow CW followed by a horizontal flow CW. The vertical flow CW was intermittently fed by a pump programmed to discharge 2.8 L every 2 hr on to the surface, specifically over the plant. The effluent flowed by gravity to the next stage. The systems were evaluated during two years. A total of 200 L/d was treated in the three systems.

Water Quality Parameters

Total coliforms (TCol) and \textit{E. coli} were quantified by the Colilert method.

RESULTS AND DISCUSSION

The three two-stage HEWTSs were highly effective for the removal of both TCol and \textit{E. coli} due among other factors to the longer HRT in comparison to one-stage systems. During the first year, the concentration of \textit{E. coli} was reduced from $1.6 \times 10^5 \pm 7.0 \times 10^5$ to $4210 \pm 1457$, $1060 \pm 326$ and to $213.1 \pm 59.0$ MPN/100mL, in the systems I, II y III respectively; achieving global removal of 99.74%, 99.93% y 99.99%. In the second year, the efficiencies were similar. The concentration of $2.2 \times 10^6 \pm 8.4 \times 10^5$ was decreased to $4346 \pm 2440$, $1030 \pm 832$ and to $370 \pm 230$ NMP/100 mL in the systems I, II and III, respectively with global removal of 99.80% , 99.95% and 99.98%. System II and III which included
HFCWs and VFCWs were the most effective (p-value<0.05) (Fig. 1), reaching average reductions from 3.18 to 3.88 log unit, during the two years. It is clear that the presence of VF components were responsible for these better results. The intermittent feeding of VF CWs promotes unfavourable aerobic conditions for indicator organisms and pathogens (Vymazal, 2005).

Fig. 1. Reduction of \textit{E. coli} in the three two-stage hybrid ecological wastewater treatment systems during two years of evaluation.

CONCLUSIONS
The use of hybrid constructed wetlands allows reclaimed water disinfection by natural processes without the use of conventional disinfectants. In the most effective two systems evaluated in this work, a >3 log unit \textit{E. coli} reduction was achieved, fulfilling the WHO guidelines desirable in wastewater treatment systems. The results obtained in the HFCWs-WSPs were lower, but they may be increased by providing a longer HRT. In this way, the advantages of these technologies over the conventional ones become more evident when the wastewater is treated with the purpose of irrigation reuse.

ACKNOWLEDGEMENTS
The work was funded by a grant from the Consejo Estatal de Ciencia y Tecnologia del Estado de Jalisco (COECYTJAL).

REFERENCES
Ecological Stormwater Treatment at Freedom Park, Naples FL
(O.84)

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INTRODUCTION

Freedom Park is a 20 ha stormwater runoff quality improvement project in the Gordon River watershed in Naples FL. Constructed ponds and wetlands and restored wetland habitats are integrated into a passive park setting of trails, boardwalks, educational facilities, and natural landscaping. Operated by the Collier County Growth Management Division and the Parks and Recreation Department, the system uses wetlands to reduce nitrogen and phosphorus in water pumped from ditches draining the watershed. Water pumped from the Gordon River supplements the system during dry, low flow seasons and helps reduce river phosphorus and nitrogen loads. Restored natural forested wetlands enhance site biodiversity and enrich the visitor experience. A versatile educational facility supports multiple civic functions that have sustained a steady increase in visitor use since the park opened. Detailed background and preliminary data for this project can be found in Bishop et al (2012).

Since system start-up in October 2009, water quality samples have been collected on a monthly basis during the June-October rainy season to assess concentration reductions in nitrogen, phosphorus, and trace metals. Annual monitoring of ecological monitoring transects in the restored section of the Park has been conducted to document the effectiveness of removal of non-native vegetation, a common management technique in south Florida. This paper summarizes key findings of monitoring during the period from 2009 through 2012, and documents how Freedom Park provides an illustrative example of a treatment wetland project that provides multiple ecological and social benefits.

WATER QUALITY TREATMENT

Phosphorus and Nitrogen

The project was designed to reduce phosphorus and nitrogen concentrations in stormwater and pumped river baseflow by approximately 30% and 70%, respectively. Figure 1 shows time series of inflow-outflow data collected from 2008-2011 for total phosphorus (TP) and total nitrogen (TN). The pond and wetland system reduces median concentrations by 84%, and significantly attenuates peak inflow concentrations. The pond and wetland system reduces median TN concentrations by 37%. Outflow nitrogen concentrations are predominantly organic nitrogen, and represent the attainable background.

Metals

Inflow stormwater concentrations did not exceed state water quality standards for common metal contaminants in stormwater, but significant reductions were observed through the wetland. For data available through 2012, measured reductions in median inflow concentrations for arsenic, copper, iron and zinc were 31%, 43%, 69% and 56% over the past four years. Outflow concentrations averaged 5.0, 1.7, 51.3, and 4.8 µg/L for As, Zn, Fe and Pb, respectively. The final concentrations of all nutrients and metals are consistent with expectations of “background” concentrations for constructed marshes in this region, and well below observed ecological effects thresholds.
Ecological Monitoring

In response to removal of non-native vegetation and opening of the forest canopy, percent ground cover doubled to 76% and ground cover species increased to 26 by Year 3.

CONCLUSIONS

Freedom Park demonstrates how a regional stormwater treatment wetland facility can be designed as an attractive yet functional community asset. Freedom Park is producing outflow nitrogen and phosphorus concentrations consistent with similarly loaded wetlands.

![Graph showing total nitrogen and phosphorus concentrations for Freedom Park, 2008-2011.](image)

**Fig. 1. Total Nitrogen (U) and Phosphorus (L) Concentrations for Freedom Park, 2008-2011.**

ACKNOWLEDGEMENTS

We gratefully acknowledge the Collier County Growth Management Division, Parks and Recreation Department and Conservation Collier for supporting the planning and implementation of Freedom Park. The South Florida Water Management District provided critical project funding and support. Johnson Engineering collected site monitoring data.

REFERENCES

Pilot-Scale Treatment of Domestic Wastewater Sequentially by Anaerobic Reactors and Constructed Wetlands: Evaluation for Water Reuse (O.94)

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EXTENDED ABSTRACT

The study involves treatment of domestic wastewater sequentially by up-flow anaerobic sludge bed reactor (UASB) and an anaerobic baffled reactor (ABR) operated in parallel followed by serially operated horizontal (HFCW) and vertical (VFCW) sub-surface flow constructed wetlands.

The study aimed to solve wastewater treatment problems of many small communities in Mediterranean countries such as Turkey by constructed wetlands in combination with anaerobic reactors operating at ambient temperatures. Constructed wetland systems are strongly affected by local conditions such as climate, land and wastewater characteristics. Therefore it is required to develop systems for local conditions together with appropriate design and operation criteria. This study will be useful in that respect. Pilot-scale studies were performed for the treatment of domestic wastewater of about 30 people in order to show applicability of constructed wetland systems downstream of anaerobic pretreatment. The experimental period covered more than 2 years (26 months). The present study involves evaluation of the whole system in terms of effluent quality.

Anaerobic pretreatment of domestic wastewater served as a pretreatment step before the constructed wetland system by decreasing the organic load. Anaerobic pretreatment has two positive effects on constructed wetland systems. First effect is protecting constructed wetlands from clogging because of high suspended matter removals, which increases the service life of wetlands. Second effect is decreasing the land requirement for the constructed wetlands by decreasing the organic matter loading. An advantage of anaerobic pretreatment over aerobic treatment is the exclusion of the need for aeration and less sludge production eliminating operational costs which is particularly important in rural areas. UASB and ABR were operated in parallel. Reactor volumes were 0.5 m³ and 1 m³, organic loading rates were in the range of 0.33-0.77 kg COD/m³.day and 0.30-0.72 kg COD/m³.day, respectively in UASB and ABR reactors and both reactors were operated at a hydraulic loading rate of 12.1 hours. In the UASB reactor, removal of COD ranged between 20-85 %. The effluent COD ranged between 96-369 mg/L where influent concentrations were 230-850 mg/L. The influent concentrations of total suspended solids (TSS) ranging between 90-450 mg/L was decreased to 20-180 mg/L in the effluent of UASB reactor. In the ABR, removal of COD ranged between 15-80 % and the effluent COD ranged between 141-472 mg/L. TSS was decreased to 20-176 mg/L in the effluent of ABR.

In downstream of anaerobic reactors, HFCW mainly removed organic matter and supported denitrification whereas VFCW mainly obtained nitrification and phosphorus removal. Surface areas were 18 m² and 13.7 m², dimensions were 3mx6mx0.8m and
3.7mx3.7mx0.8m, hydraulic retention times were 1.4–2.2 and 0.5–1 days, hydraulic loading rates were 111 - 167 L/m2day and 146 - 219 L/m2day, and porosities were 28 % and 33 %, respectively for the horizontal and vertical flow systems. The removal efficiencies of the constructed wetland system are shown in Table 1.

Table 1. Removal efficiencies (%) of the constructed wetland system

<table>
<thead>
<tr>
<th>Period</th>
<th>COD</th>
<th>BOD&lt;sub&gt;5&lt;/sub&gt;</th>
<th>TN</th>
<th>TSS</th>
</tr>
</thead>
<tbody>
<tr>
<td>I: Summer</td>
<td>89±9</td>
<td>91±4</td>
<td>55±21</td>
<td>82±19</td>
</tr>
<tr>
<td>II: Winter</td>
<td>79±13</td>
<td>70±12</td>
<td>19±11</td>
<td>71±10</td>
</tr>
<tr>
<td>III: Rapid-draw (R=0)</td>
<td>90±3</td>
<td>89±24</td>
<td>29±9</td>
<td>97±1</td>
</tr>
<tr>
<td>IV: Rapid-draw (R= 100%)</td>
<td>91±4</td>
<td>87±8</td>
<td>66±25</td>
<td>98±1</td>
</tr>
<tr>
<td>V: Rapid-draw (R= 200%)</td>
<td>92±3</td>
<td>86±6</td>
<td>66±11</td>
<td>97±5</td>
</tr>
</tbody>
</table>

Recirculation of the wastewater was performed from the effluent of VFCW to the influent of HFCW in order to operate HFCW as a denitrification unit. Thereby nitrate produced in VFCW was provided to HFCW for further denitrification. Consequently, effluent NO<sub>3</sub>-N concentrations ranging between 50–60 mg/L during the nonrecirculating period (Period III) decreased to below 15 mg/L when the recirculation ratio was 100% in Period IV. Recirculation of the effluent was found to increase particularly total nitrogen removal in the wetland system. Organic matter removal was comparably higher in VFCW (68.0±15.5 % COD removal) than HFCW (61.9±17.0 % COD removal). TNK and NH<sub>4</sub>-N removal efficiencies were significantly higher in VFCW (65.0±28.6 % TKN and 67.9±29.6 % NH<sub>4</sub>-N removal) compared with HFCW (17.1±12.5 % TKN and 13.1±9.0 % NH<sub>4</sub>-N removal). These results indicated that operation of HFCW and VFCW in series will be more efficient than operating these wetlands single or in parallel in terms of total nitrogen removal, since one wetland serves as a nitrification unit and the other as a denitrification unit. Phosphorus removal was also higher in VFCW. VFCW achieved up to 60–90 % phosphorus removal whereas HFCW could remove only less than 20%.

Table 2 shows that effluent concentrations meet irrigation criteria for pH, TSS and BOD<sub>5</sub> parameters as well as the discharge standards. However, for fecal coliform and residual chlorine, disinfection is required in the case of reuse of wastewater for irrigation of agricultural plants and urban recreational areas.

Table 2. Comparison of the effluent characteristics of the pilot-scale system with the irrigation water criteria

<table>
<thead>
<tr>
<th>Period I</th>
<th>Period II</th>
<th>Period III</th>
<th>Period IV</th>
<th>Period V</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>7-7.5</td>
<td>7-7.5</td>
<td>7-7.5</td>
<td>7-7.5</td>
</tr>
<tr>
<td>TSS (mg/L)</td>
<td>8 ± 6</td>
<td>15 ± 6</td>
<td>2 ± 3</td>
<td>1 ± 0</td>
</tr>
<tr>
<td>BOD&lt;sub&gt;5&lt;/sub&gt; (mg/L)</td>
<td>8 ± 4</td>
<td>25 ± 11</td>
<td>9 ± 8</td>
<td>7 ± 3</td>
</tr>
<tr>
<td>Residual Chlorine (mg/L)</td>
<td>--</td>
<td>--</td>
<td>--</td>
<td>&gt;1</td>
</tr>
</tbody>
</table>

The study involves evaluation of the whole system in terms of effluent quality. It was shown that it was possible to achieve on average up to more than 95% organic matter removal and above 90% nitrogen removal. Consequently the effluent met the regulations for discharge limits in terms of organic matter and nitrogen with COD and TN concentrations achieved down to below 20 mg/L. It was also shown that effluent of the system can be reused for irrigation if it is disinfected properly via chlorination. The study showed that a hybrid constructed wetland system with anaerobic pretreatment is a very effective method to treat domestic wastewaters of small communities.
Implementing advanced CW technology in India SWINGS a cooperation project aimed at implementing integral domestic water treatment and reuse. (O.161)

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ABSTRACT

A proposal in the framework of cooperation between the European Union (EU) and India has resulted in funding for a research project involving several European and Indian partners. The project involves the establishment of constructed wetland technology and disinfection units in several sites in India. The design, construction, operation and research of the systems are joined efforts among the partners and financed by both the EU and the Indian Government. The project calls for the construction of two wastewater treatment facilities, one at the Indian Institute of Technology in New Delhi (IITD) and a second one at Aligarh Muslim University (AMU) with a combination of technologies to treat wastewater as well as to provide water with high quality for the reuse in productive activities such as irrigation, aquaculture and others. Additionally, two more sites dealing with disinfection and reuse of treated water are already established and in operation at International Center for Ecological Engineering (KLYUNIV) and Indira Gandhi National Tribal University (IGNITU).

One of the systems was designed to treat water generated from a students’ dorm in the campus of IITD. The system will serve 600 students, will be constructed adjacent to the building and integrated as part of the gardens. The effluent will be reused for irrigation purposes for the gardens. Historical raw wastewater data at the site show BOD₅ concentrations of around 150 mg/L, TSS of 220 mg/l and TN of around 45 mg/L. Average water usage per PE in the dorms is 135 l/d.

The projected solution is a combination of anaerobic primary treatment followed by a combination of constructed wetlands. Following the primary-secondary treatment and for research purposes, four parallel disinfection units, using different technologies are to be built. The primary treatment will be a two stage anaerobic digestion is an integrated UASB connected to a sludge digester system. The sludge will be recirculated form one reactor to another to improve the methanogenic step and increasing the overall performance of the primary treatment.

The CWs planned are of two types, the first stage will be a French style bed, where wastewater can be loaded sequentially on three parallel beds. Following, the second stage are two SFCW of different depth. The CW design includes the possibility of bypassing and/or recycling water in order to produced effluents with different characteristics and supplying treated wastewater to the disinfection units. Following the CWs disinfection units will be established; including, natural UV disinfection, an ultraviolet unit (UV), an anodic
disinfection system (AD), and an ultrafiltration unit (UF). All the systems are solar driven and demand no external energy.

Figure 1. Layout of the system for treating wastewater generated in the students’ dormitory at IITD

The plant at AMU will treat WW produce by around 700 PE at the Campus. The inlet concentrations are similar to the ones presented for the IITD. The system comprises the construction of a combination of anaerobic primary treatment, CW system and a disinfection unit that will allow the direct reuse of the effluent for crop irrigation. Figure 2 shows the characteristics of the system. The anaerobic treatment will be a two stage system a hydrolytic-acidogenic reactor followed by a methanogenic reactor with a capacity of ca. 50 m$^3$. The secondary treatment consists of two parallel CWs treatment lines, with a combination of vertical flow beds, followed by horizontal subsurface flow beds planted with local species. The design allows possibilities of bypass, recirculation and different operational options to test several exploitation strategies. The recirculating of treated effluents will enhance the performance and removing of substances such as total nitrogen and H$_2$S reducing possible odors and producing a better effluent. Following the CWs the system is fitted with a disinfection unit to produce effluent for irrigating fields and providing sufficient water to supply the needs of the plant itself.

Figure 2. Layout of the system for treating wastewater generated from AMU campus.

The implementation of the systems under the umbrella of SWINGS project, will give the opportunity to test advance CW technology operating under actual needs, tropical conditions, using local building material, indigenous plants and local experience. The flexibility and possibilities provided by the design of the treatment plants will permit the testing of different configurations will advocate for the use of CWs in India and increase the knowledge regarding design and operational needs. An additional benefit is the fact that all the water treated by the systems will be reused and will be treated according to the targeted needs. The reuse might include irrigation, toilet flush and with the technology under the project with the possibility of supplying water with higher quality standards.
Pollutant modelling in wetlands
Modelling stochastic behaviour of horizontal flow constructed wetlands subjected to variable loads (O.17)

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Constructed wetlands (CWs) are a wastewater treatment technology that exploits the physical, chemical, and biological processes occurring in soils to improve water quality. The low operation and maintenance costs allow for a widespread use of CWs, especially for treating wastewater from small communities (up to 2000 people). Despite the wide application of CWs, many aspects of CW behaviour in real environments still need to be investigated. For instance, the stochastic behaviour of CW outflows is generally neglected during the typical CW design procedure, and CW size should highly overestimated to face real outflow variability [Kadlec and Wallace, 2009]. Hence, further efforts are still need to improve the ability to properly design CWs. Here a process-based model approach is used to better understand the response of a horizontal flow (HF) CW subjected to stochastic loads.

The capability of HYDRUS wetland module [Langergraber and Šimůnek, 2012] in modelling the behaviour of a HF-CW subjected to variables loads is tested simulating the laboratory HF-CWs investigated by Galvão and Matos [2012]. In their experiments, HF-CWs were fed with unsteady COD influents characterised by sudden loads [Galvão and Matos, 2012]. Since Galvão and Matos, [2012]'s HF-CWs have been fed with synthetic wastewater, we have assumed COD influent fractionation different from typical domestic wastewater values. On the other hand, only few HYDRUS default values have been changed, comprising the nitrogen content of organic component to guarantee the nitrogen mass balance of inflow loads. On the whole, the model describes very well the system since the simulated global COD removal efficiency, 68%, is strongly near the value registered in the experiments, 67%. Moreover, the model fits the temporal trend of COD effluent concentration adequately well, as visible in Fig.1, with an average absolute error equal to 20%, confirming the goodness of HYDRUS as a valide tool to investigate the HF-CWs response to stochastic loads.

Subsequently, we have investigated the stochastic behaviour of the HF-CW, applying the standardized test procedure according to EN 12556-3 (2005).

Finally, we have started a more detailed stochastic analysis in order to be able to define COD influent load via an analytical stochastic noise. In this way, we will model more realistically the response of HF-CWs to stochastic inflows, hoping to be able to shine a light on the stochasticity of COD effluents. This analysis will be developed in the following steps. Firstly, we force the HF-CW model with a sudden step from a lower load to a higher value. This is aimed to understand the typical response time of the system. Different step heights are considered, combined with different HF-CW parameters. Secondly, we force the CW with a periodic signal with typical periods, T, chosen in relation to the response time of HF-CW evaluated from the previous point; different amplitudes are tested. Thirdly, we force the HF-
CW with a noisy input signal synthetically generated with stochastic methods [Ridolfi et al., 2011], which allow to control the statistical properties of the signal. Even in this case, the typical scales of the noise component are chosen in order to match the timescales of CW.

The stochastic analysis proposed here can be as a new tool to better understand these highly dynamic systems. Particularly, the main goal is to improve the HF-CW design procedure, limiting the safety overestimation of HF-CW size needed to face the HF-CW stochastic behaviour [Kadlec and Wallace, 2009].

![Fig.1: Comparison between COD effluent concentrations simulated (continuous line) and measured (circle with standard deviation bar) for higher and lower average inflows (upper panel and lower panel, respectively).]

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Numerical simulation of the treatment performance of a horizontal flow constructed wetland for polishing an SBR effluent (O.41)

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INTRODUCTION

The wastewater treatment plant of Balf, Hungary is a Sequencing Batch Reactor (SBR) with 4000 PE (700 m³/d) capacity including phosphorus precipitation (ÉDKTVF, 2008). It is located in the basin of Lake Fertő (Lake Neusiedl) which is an important natural sight including areas protected by strict national and international law.

The SBR plant in a long-term average meets the effluent quality threshold values. However, the values are exceeded during short periods due to variable inflow rates and concentrations to the treatment facility worsened by colder temperatures. The aim of this study was to numerically simulate the treatment performance of a horizontal flow (HF) constructed wetland (CW) for polishing the effluent of the SBR plant to meet the required effluent standards.

MATERIALS AND METHODS

The investigated system includes a buffer pond and a HF CW. The buffer pond was included into the theoretical system to achieve a better economy of scale of the filter bed as the batch is emptied on a daily basis in about 1.5 hours which would mean a high hydraulic loading rate. Only the process of mixing was considered in the pond and it had a constant outflow rate.

The HF filter bed was initially dimensioned based on the hydraulic loading rate as the inflow concentration values were low. The targeted hydraulic retention time in the first layout was 5 days with gravel media, with dimensions of 15m x 60m x 0.5m (length, width and depth respectively). A two-month simulation was run to establish stable microbial populations followed by simulation runs for the months November to March (including) in which the highest SBR effluent concentrations occur and the legal requirements were failed to be met in many cases. After the initial run various dimensions of the filter beds and various media was considered for better nitrogen and phosphorus removal which seemed to be critical parameters at low temperature and by first modelling results.

The CW2D biokinetic model (Langergraber and Šimůnek, 2005) of the HYDRUS Wetland Module (Langergraber and Šimůnek, 2012) was selected to simulate the HF bed for further investigations. The CW2D biokinetic model and a HF CW have been chosen because:

1. phosphorus removal was necessary to be modelled,
2. the large dimensions of the system would cause low mass load of oxygen demand per unit surface,
3. the nitrogen in the inflow is mainly nitrate thus denitrification is desirable which needs anoxic environment, and
4. robustness and absolute simplicity for low operation costs can outcompete alternative options for solving the issue and could popularize alternative solutions for wastewater treatment experts.
The model was set up using effluent concentration and volume data provided by the waterworks. As concentration data was available only on a weekly basis a data series was generated by organizing measurement data after months which provided a series of data of 306 elements, averagely 26 per month.

RESULTS AND DISCUSSION
The initial simulation runs showed more balanced concentrations after the HF bed, especially for COD and BOD levels which stayed all the time below the legislative threshold, a concentrated peak load resulted in concentrations still slightly higher than effluent standards at about 10 mg/l for ammonia nitrogen (threshold: 5 mg/l) and 1.6 mg/l for total phosphorus (threshold: 1 mg/l). A filter bed extended in length results in constant COD and BOD outflow but denitrification is not happening as the last meters of the bed have already aerobic conditions.

The model showed that different media with better adsorption parameters and a deeper bed for anoxic conditions are to be considered for further improvements.

OUTLOOK
In the presentation the outcomes of various simulation setups will be shown and discussed. The feasibility of using HF CW as the last element of the treatment chain to meet the legislative thresholds set on the effluent of the wastewater treatment plant at Balf will be further investigated.

ACKNOWLEDGEMENT
Many thanks for the base idea of the topic and all the support to the colleagues at the Institute of Geomatics and Civil Engineering, University of West Hungary, Sopron. Additionally, the main author thanks the colleagues at Sopron Waterworks for the data and guidance on-site, the participants of the 3rd Amadée Meeting for ideas and adsorption parameter data, and Péter Csáfordi and Zoltán Emődy-Wáman for providing accommodation and transport.

REFERENCES
Clogging of Vertical-flow Constructed Wetlands Treating Urban Wastewater (O.48)

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INTRODUCTION

Vertical-flow constructed wetlands are used worldwide as an alternative means of environmental pollution control (Scholz, 2010). A major drawback limiting their efficiency is clogging of the media as a result of suspended particles originating from the influent stream. In most of the previous research on clogging, very few scientists focused on clogging and performance efficiency as a function of hydraulic and organic loading rates, media size, and contact and rest times treating domestic wastewater (Dong et al., 2011).

The study aims were to assess the impact of operational and design variables on the clogging and overall water quality treatment performance of vertical-flow wetlands, to determine the effect of hydraulic and organic loading rates, particle size, contact and aeration time, and to evaluate a simulation model assessing the impact of sedimentation of suspended solids on clogging and treatment efficiency.

MATERIALS AND METHODS

Different laboratory-scale vertical-flow constructed wetlands filled with gravel and planted with common reed were used to assess clogging mechanisms and treatment efficiency as a function of hydraulic and organic loading rates, media size, and contact and rest time. The analysis of water quality parameters was carried out according to APHA (2005). The wetlands were constructed and operated between June 2011 and June 2012. Data from June 2011 (setting-up period) were not used. Furthermore, a numerical model to assess clogging has been developed. The model takes account of parameters including sedimentation, surface adsorption and convection. The finite element software COMSOL was used.

RESULTS AND DISCUSSIONS

Findings indicate that all systems performed relatively similar independently of their specific design and operation (Table 1). The filter with the highest chemical oxygen demand (COD) loading performed the worst in terms of outflow COD concentration (120 mg/l), but best in terms of COD load reduction (62%). With regards to clogging, the simulation model (Figure 1) indicates that none of the systems have shown any signs of clogging after about one year of operation based on the water quality analysis of all filters.

CONCLUSIONS

All constructed wetland systems have shown high removal efficiencies for the key water quality parameters COD, ammonia-nitrogen, ortho-phosphate-phosphorus and suspended solids; the removal efficiencies exceeded 61, 40, 57 and 76%, respectively. No sign of filter clogging was noticed. However, it is expected that the differences in design and operation will eventually lead to differences in the evolution of clogging patterns, which will subsequently help to understand the clogging mechanisms.
Table 1. Results for an example filter (two replicates) showing means, removal efficiencies and statistical differences (p<0.05) applying the non-parametric Mann-Whitney U-test.

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Unit</th>
<th>Example filter Mean</th>
<th>Concentration removal (%)</th>
<th>Load removal (%)</th>
<th>p value and H</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chemical oxygen demand</td>
<td>mg/l</td>
<td>120.1</td>
<td>61.5</td>
<td>61.5</td>
<td>&lt;0.000; H=1</td>
</tr>
<tr>
<td>Ammonia-nitrogen</td>
<td>mg/l</td>
<td>18</td>
<td>40.7</td>
<td>40.6</td>
<td>&lt;0.000; H=1</td>
</tr>
<tr>
<td>Nitrite-nitrogen</td>
<td>mg/l</td>
<td>0.29</td>
<td>86.9</td>
<td>86.8</td>
<td>&lt;0.618; H=0</td>
</tr>
<tr>
<td>Nitrate-nitrogen</td>
<td>mg/l</td>
<td>2.9</td>
<td>-89.9</td>
<td>-93.3</td>
<td>&lt;0.074; H=0</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>mg/l</td>
<td>5.7</td>
<td>20.6</td>
<td>20.8</td>
<td>&lt;0.163; H=0</td>
</tr>
<tr>
<td>Ortho-phosphate-phosphorus</td>
<td>mg/l</td>
<td>4.8</td>
<td>57.2</td>
<td>57.1</td>
<td>&lt;0.000; H=1</td>
</tr>
<tr>
<td>Suspended solids</td>
<td>mg/l</td>
<td>24.9</td>
<td>76.5</td>
<td>76.5</td>
<td>&lt;0.008; H=1</td>
</tr>
</tbody>
</table>

Example filter, aggregate diameter of 10 mm, contact time of 72 h, resting time of 48 h and chemical oxygen demand inflow of 311.7 mg/l. H, probability response; if H=1, filters are statistically different (p value<0.05), and if H=0, the difference is not significant.

Fig. 1. Comparison between measured and modelled distribution of suspended solids within eight example wetland filters.

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Modelling phosphorus removal in steel slag filters (O.58)

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INTRODUCTION

Steel slag filters are known to remove efficiently phosphorus from wastewater (Chazarenc et al., 2007, 2008). However, lack of similarity between reported studies concerning filter longevity and phosphorus removal performance prevents direct large-scale utilization of slag filters. Such irregularities create a need for a comprehensive model of phosphorus removal mechanisms in slag filters. Typically, phosphorus removal potential of slag filters are expressed in terms of mass of P removed per mass of slag (Vohla et al., 2011). The objective of this project was to propose a phosphorus removal mechanism model adaptable to numerical simulations and longevity prediction. An original approach based on crystal formation was used (Claveau-Mallet et al., 2012).

METHODS

The experimental program included two parts, column and jar tests. Column tests were run to investigate the influence of various operation parameters on crystal properties. Jar tests allowed to characterise the dissolution behavior of slag.

Column tests

Lab-scale tests were conducted using slag filters and synthetic phosphorus solutions. Plexiglas columns (17 to 40 cm in length and 15 cm in diameter) were bottom fed with solutions containing 19 to 106 mg P/L. Electric arc furnace slag from Contrecoeur, Canada, was used. Tests were run during 30 to 630 days. Void hydraulic retention time (HRT\textsubscript{V}) ranged from 1.5 to 29 hours. Effluent was periodically sampled and analysed for pH, o-PO\textsubscript{4} and calcium. At the end of experiments, the precipitates present in columns were sampled and analysed by X-ray diffraction, transmission electronic microscopy and scanning electronic microscopy. The size and the organization of crystals were analysed.

Jar tests

A given amount of EAF steel slag was placed in a 1L Erlenmeyer filled with 700 mL phosphorus solution or demineralised water. The Erlenmeyer was topped with a rubber cap and a pH probe. The remaining air present in the Erlenmeyer was filled with nitrogen. pH was monitored for 24 hours while the Erlenmeyer was mixed on a gyratory shaker at 100 rpm and 20°C. pH-time curves were transformed to CaO dissolution-time curves.

RESULTS AND DISCUSSION

The model is divided in three steps:

\textbf{Slag dissolution}. The reaction is simplified to the rate of dissolution of CaO. The effect is to raise pH and calcium concentration in the filter.

\hspace{1cm} a) The empirical reaction rate is determined on the basis of jar tests results.

\textbf{Hydroxyapatite (HAP) formation and growth}. Formation of HAP is related to saturation regarding P, Ca and OH. Thus, HAP formation is related to the contact time with slag.

\hspace{1cm} b) HAP growth was directly dependent on P concentration in the interstitial water. A correlation between HAP growth rate and P concentration in the influent was obtained.
c) Crystal structure was dependent on P concentration in the influent and water velocity. Observed crystal structures ranged from spherical aggregates to random needle network. An example of crystal structure is shown in Figure 1.

![Crystal spherical aggregates in slag filters.](image)

**Modification of hydraulic properties of the filter.** The presence of HAP crystals decreases the filter permeability, creates short-circuiting and reduces diffusion of hydroxides from slag surface to water.

d) Crystal structure had an effect on phosphorus removal performance. When crystals were dispersed and unorganized, space was not efficiently used and filters' longevity was reduced.

The authors propose to simulate the three steps in a numerical software that combines water transport, kinetic reactions and chemical equilibrium (Courcelles, 2011).

**CONCLUSIONS**

A conceptual model for phosphorus removal mechanisms in steel slag filters was proposed. The formation of HAP crystals and their effect on hydraulic properties of the filter is the key concept of the model. The conceptual model may be implemented in a simulation software in three steps: 1- slag dissolution, 2- HAP formation and growth and 3- modification of hydraulic properties of the filter.

**REFERENCES**


Simplified modelling of constructed wetlands for combined sewer overflow treatment - results from German systems and discussion of adaptation in France (O.145)

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INTRODUCTION

During rain events, discharges in combined sewer systems often exceed the downstream capacities (pipes, treatment plants). Runoffs can be partly stored in tanks before subsequent treatment, but combined sewer overflows (CSOs) cannot be avoided for extreme events. One approved technical solution to reduce the ecological impact on receiving waters is the use of vertical flow constructed wetlands, in Germany known as retention soil filters (RSFs). First RSF examples have been in operation since about 1990. Several hundred filters were built during the last 20 years, but only a few were monitored in detail (Dittmer & Schmitt, 2011).

Similar systems in France are currently under experimental research. New developments can be found in (a) avoiding storage tanks by increasing the size of the wetland and the filter media, (b) dividing the wetland into two filter beds, and (c) providing a permanent water layer in order to avoid water stress during extended dry periods (Meyer et al., 2012).

This study is focussed on a simplified modelling tool to improve design and operation, developed and tested in Germany. The proven first version called RSF_Sim is currently under validation in France in order to be implemented into commercial sewer models.

MATERIALS AND METHOD

The database for the modelling and simulation study consists of result from monitoring two 'state-of-the-art' full-scale RSFs. Both datasets could be combined, because a detailed comparison proved high levels of similarities (Meyer, 2011). Extreme feeding events were used to simulate operational limits. Long-term observations indicated influencing factors.

Model descriptions had to be developed for hydraulic operation and for treatment of the main pollutants COD and NH₄-N. The duties for the hydraulic model can be found in displaying the retention function, as well as in providing correct volumes and flows as basis for calculations of concentrations and contact times. According to the real vertical RSF structure, the filter basin was dived into a retention layer on top, a main process layer (sand), and a drainage layer. All calculations are based on simple mass balances within each layer, described as stirred tanks with typical time steps of 5 minutes. As an exception, outflow of the process layer can start only if the theoretical retention time has been passed.

The COD reduction model works with two fractions: particulate COD (COD_X) is reduced down to a constant background concentration by filtration. Dissolved COD (COD_S) was assumed to be reduced by a reduction rate. In order to find influencing factors for changing performances, the database was analysed for inflow concentrations, temperatures and the duration of previous dry periods, but only last showed a correlation (Meyer, 2011).

NH₄-N removal is described in two steps: Inflow masses are retained by adsorption during feedings, kinetic nitrification starts simultaneously to re-aeration after drainage. In order to
keep the adsorption calculation simple, a two-stage linear correlation was created as isotherm. It basically allows estimating regenerative retentions up to a proven maximum load. Fitting input parameter values could be achieved first by calibrating retention limits during single feeding events. Afterwards input averages were used to simulate all events.

RESULTS AND DISCUSSION

The results of the hydraulic model application showed good matches between the simulated and measured ponding water levels in the retention layer. Thus, hydraulic overloads can be well predicted. Limits were observed due to special flow characteristics: If the inflow rate from the CSO is lower than the maximum filter outflow rate, the model still assumes a perfect distribution inside the layers. In reality the given correlation leads to shortcuts. To overcome this, the filter bed calculation should be divided into surface zones.

Long-term COD_S simulation results for one of the RSFs with the basic version (44 % as constant removal performance) showed deviations of -10 to +30 % compared to the observed reduction of inflow loads. Especially positive %-values should be considered as potentially critical due to overestimation of treatment performances. When the previous dry period was respected as an event-specific influencing factor, this could be reduced to +3 % in maximum.

Long-term simulations of NH4-N retention with average input values showed deviations from +/-16 % in comparison of the observed/simulated mass retentions per event. In summary of all feedings, the total calculated outflow mass was similar to the measured one. Thus, the application of the given approach was satisfying for displaying long-term operation, but an increased database needs to be simulated for validation (e.g. in Tondera et al., 2013).

CONCLUSIONS AND OUTLOOK

The new developed model RSF_Sim combines simple basic approaches with carefully selected detailed descriptions. Functions for main parameters like COD fractions and NH4-N could be well displayed for critical single events as well as in their long-term behaviour. The number of input parameters was kept very small, and initial numbers are available from literature. Limits of application can be found due to irregular operational conditions.

The current research project on the French CSO-CWs (“ADEPTE”) will help to achieve new input values for the existing model RSF_Sim, and to adapt it to new needs. The role of the permanent water layer has to be investigated in real operation first. Also feeding with raw CSO may have an influence on the hydraulic operation: The increased sludge accumulation contains potential clogging risks, which can be managed by alternated loading of the filter beds. As a general step of improvement a horizontal discretisation of the filter is necessary.

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How modeling improves the design of French vertical flow CW (O.153)

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INTRODUCTION

Constructed wetlands (CW) are complex engineered systems designed for optimized treatment conditions found in a natural environment. Up to now, the most widely used approaches for CW design and optimizations are based on empirical rules inferred from the experience of the designers (trial and error). The limitations of this approach are due to the fact that they are cost and time consuming to get knowledge under various contexts.

Mechanistic modeling can be useful (1) to optimize the system design (position of the feeding points, drainage networks to avoid short-cuts and to ensure an adequate drainage); (2) to get a better understanding of biological and chemical transformation and degradation processes which takes place in CWs; (3) to predict the system aging and what happens when the system limits (hydraulic or organic) are exceeded. But currently, many gaps in the modeling processes make the models difficult to calibrate for a wide range of contexts (temperature, loads, wastewater characteristics, etc.). Consequently, such mechanistic models are not easy to use at the moment by designers. Simplified models appear interesting in this case, but they need to be validated with a large number of data under different operating conditions. Therefore, working with these both modeling approaches in parallel is a complementary approach that can provide design rules or design framework. In that way, mechanistic models could be used for a better validation of simplified models.

![Fig. 1. Steps to obtain a simplified model usable by designers of VFCW.](image)

Up to now, the mechanistic HYDRUS/CW2D model (Langergraber and Šimůnek, 2005) suffers from a lack of calibration of parameters to be applied to the modeling of the hydraulic and biological behavior of a French-type Vertical Flow Constructed Wetland (VFCW). The objectives of the work, in a first step, were (1) the validation of the concepts used in the model (to evaluate the model ability for predicting the carbon and nitrogen concentrations measured in the effluent of the VFCW); and, (2) the investigation of the processes occurring during a feeding period. From this study we identified the currently missing data to improve the mechanistic model and build the simplified model.

METHODS

The HYDRUS/CW2D model

For the hydraulic modeling we first compared the standard van Genuchten-Mualem function (VGM model) and a dual-porosity model (Morvannou et al., 2013) as the porosity
and the network of reed’s stalks may serve as preferential flow paths through which water can bypass most of the porous matrix. In that way, non-equilibrium conditions in pressure heads are created between preferential flow paths and the matrix pore region.

For the biological modeling, the HYDRUS/CW2D model appears to be reasonably complex but realistic to simulate VFCW behavior.

The French-type VFCW studied and data

The Evieu wastewater treatment plant (Ain, France) was described by Molle et al., 2008.

We used a combination of in-situ data (TDR probes, gas measurements), in/out data (tracer experiment, outflow measurements, raw/treated wastewater characteristics) as well as lab-scale data for comparison with simulation results.

RESULTS AND DISCUSSION

After the hydraulic calibration, the VGM model succeeds to reproduce water content variations at various depths of the VFCW. The CW2D model simulated the two major processes (ammonium adsorption and nitrification) occurring during the feeding period. However, it is not possible to reproduce the tracer breakthrough curve with this equilibrium approach. Moreover, despite the consistency between simulated and measured values on a batch average basis, concentration peaks observed at the beginning of each batch cannot be reproduced by CW2D and oxygen contents within the VFCW are poorly simulated (macropore desaturation in the VFCW is actually faster and access to oxygen is easier than modeled by HYDRUS). These disparities are probably due to non-equilibrium preferential flows, which rapidly discharge, NH\textsubscript{4}-N and NO\textsubscript{3}-N in the outflow (short circuit like).

The comparison between measured and simulated tracer breakthrough curves indicates that the non-equilibrium (i.e. dual-porosity model) approach seem to be the most appropriate for simulating preferential flow paths (Morvannou et al., 2013).

Thus, in order to simulate the VFCW behavior in the long term, it is necessary to take into account the degradation pathways of the organic matter (different hydraulic behavior according to the organic matter type) and its influence on the hydraulic properties of the porous material. In addition, the maturation of the filter and the production of the sludge layer have to be considered. However, HYDRUS/CW2D does not take into account for the solid transport, neither the surface runoff that occurs with batch feeding. Furthermore, the importance of ammonium adsorption onto organic matter has to be better studied to fit nitrification performances observed in the systems.

CONCLUSIONS

Up to now the first HYDRUS/CW2D model calibration allow us to validate the concepts used in the model and to better understand which processes occur within a VFCW. We identified the missing data to improve the mechanistic model and to build the simplified model.

REFERENCES


Simulation of constructed wetland microcosms using the HYDRUS wetland module (O.42)

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INTRODUCTION

Experiments with batch-operated constructed wetland microcosms have been carried out with different plant species at known temperatures and synthetic wastewater (Allen et al. 2002). Mburu et al. (2012) simulated data from these experiments using an implementation of the Constructed Wetland Model 1 (CWM1, Langergraber et al., 2009) in the AQUASIM software (Reichert, 1998). The objective of this work was to verify the implementation of CWM1 in the HYDRUS wetland module (Langergraber and Šimůnek, 2012) using the same experimental data as Mburu et al. (2012).

MATERIALS AND METHODS

HYDRUS numerically solves the Richards equation for saturated/unsaturated water flow and the convection-dispersion equation for heat and solute transport. The flow equation incorporates a sink term to account for water uptake by plant roots. The solute transport equations consider convective-dispersive transport in the liquid phase, diffusion in the gaseous phase, as well as non-linear non-equilibrium reactions between the solid and liquid phases (Šimůnek et al., 2011). Version 2 of the HYDRUS wetland module includes two biokinetic model formulations (CW2D and CWM1) simulating reactive transport in CWs (Langergraber and Šimůnek, 2012).

The column experiments used for the simulations were carried out at Montana State University (Allen et al. 2002). All columns had the dimensions of \( D = 20 \) cm and a height of 50 cm, including one unplanted microcosm and three planted with Carex, Schoenoplectus and Typha. Operation temperatures which were simulated were 12, 16, 20, and 24°C, all batch lasted 20 days using synthetic wastewater. This uniformity of columns and the way of operation facilitates to link numerical model results with physical model results and thus the verification of certain parameters and overall functionality.

Columns were represented as an axi-symmetrical 2D finite element mesh with 21x31 (horizontal x vertical) elements. Known parameters were set or interpolated and the same measurement time and observation point depth was used as for the physical model. Many simulations were run and only those estimated parameters were changed during the trials which seemed to cause inconsistent results with measured data. These estimated parameters include initial bacteria concentrations, adsorption coefficients, root re-aeration and initial amount of adsorpted ammonia.

Mburu et al. (2012) used the parameter estimation routines available in AQUASIM to estimate parameter values during calibration and thus presented different parameter values for each experimental set-up. We used a more process based approach, i.e. first calibrating the model to the data of the unplanted control and then for the other columns only changing parameters with reasoning.

For evaluation of data an MSExcel® sheet was set up for linking output files and generating diagrams representing important concentrations in both the liquid and solid phases. This facilitated greatly to evolve parameters by comparing results of different trials and columns.
RESULTS AND DISCUSSION

Data from the unplanted control were simulated well for COD, NH$_4$-N and SO$_4$-S for different plant species and temperatures, respectively.

For planted columns (with Carex and Schoenoplectus) simulated COD values fitted well to measured data when using the same linear adsorption parameters as for the unplanted column. In contrary, a standard adsorption parameter set representing a single Freundlich isotherm was not appropriate to fit ammonia concentrations of both planted and unplanted columns. Sulphate sulphur concentrations had an overall bad fit which were corrected by adjusting few parameters of the CWM1 biokinetic model (those responsible to differentiate between aerobic and anaerobic conditions, e.g. inhibition coefficients of certain bacteria).

Adding tap water at the bottom to balance evaporation losses as described by Allen et al. (2002) resulted in a strongly dilated layer of solute expanding upwards in time against a shrinking, more and more concentrated layer at the upper part. High longitudinal and transversal dispersion coefficients (5 and 4 m, resp.) had to be used to ensure proper mixing.

Another problem faced was that aerobic conditions in the columns could not be modelled in HYDRUS if the root oxygen re-aeration is exceeding the oxygen demand of the system. This case results in the saturation of the system in reality, however in the software there is no limitation on oxygen pumped in by the roots in the system.

OUTLOOK

Simulation runs for all planted columns at all temperatures are about to be performed for further refinement of the necessary biokinetic parameters to achieve best possible fit of modelled sulphate concentrations. Freundlich isotherm parameters will be set differently for planted and unplanted columns considering a two-step sorption process (i.e. in biofilm and filter material) as a bulk process if no common solution will be found. A saturation function will be built in in root oxygen pumping and simulations will be re-run.

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