Initial contaminant removal performance factors in horizontal flow reed beds used for treating urban wastewater

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Abstract

This study evaluates the effect of hydraulic loading rate (HLR), aspect ratio, granular medium size and water depth on the removal of selected contaminants during the start up of horizontal subsurface flow reed beds (HFRBs). Experiments were carried out in a pilot-scale HFRB system comprising four pairs of lined beds of almost equal surface area (54–56 m² each bed), with the following aspect ratios: 1:1, 1.5:1, 2:1 and 2.5:1. The size of the granular medium of each pair varied from coarse granitic gravel ($D_{60} = 10\text{ mm}$, $C_u = 1.6$) to small granitic gravel ($D_{60} = 3.5\text{ mm}$, $C_u = 1.7$). The beds of the pair with longest aspect ratio were made shallower (0.27 m) than the rest (0.5 m). The system was sampled weekly from May 2001 to January 2002. The results indicate that HLR and water depth are determining factors in the performance of the HFRBs. Beds with a water depth of 0.27 m removed more COD (70–80%), BOD 5 (70–85%), ammonia (40–50%) and dissolved reactive phosphorus (DRP) (10–22%) than beds with a depth of 0.5 m (60–65% for COD, 50–60% for BOD 5, 25–30% for ammonia, and 2–10% for DRP). The higher efficiency observed shallower beds was related to their less reducing conditions (average redox potential ($E$) ranging from $-351$ to $-338$ mV) than beds with a depth of 0.5 m ($-390$ to $-358$ mV). The difference in $E$ status between two bed types seems to lead to differences in the biochemical processes. In fact, denitrification was estimated to be a significant reaction in shallower beds.

Keywords: Constructed wetlands; Hydraulic loading rate; Aspect ratio; Water depth; Subsurface flow

1. Introduction

Horizontal flow reed bed (HFRB) technology is mainly used for the treatment of urban and domestic wastewater from small rural communities. During the design of full scale HFRBs the most relevant parameters to be defined are: hydraulic loading rate (HLR), aspect ratio, granular medium size and water depth [1]. The effect of HLR on organic matter and nutrient removal in HFRBs has been assessed in several studies [2–4], and the results indicate that lowering the HLR improves the efficiency. Most of the HLRs are in the range of 10–100 mm/d [5, 6].

A popular notion in HFRB design has been that increasing the aspect ratio of the beds will produce a behaviour closer to plug flow and therefore better efficiency. However, because in the microscale mixing occurs, this notion is not completely correct [1]. Several studies with innocuous tracers have repeatedly demonstrated that an increasing aspect ratio delays the breakthrough time but the spread of the response curve is similar for lower and higher aspect ratios [7, 8]. This
characteristic implies that contaminant removal is not so dependent on the shape of the bed. Bounds et al. [9] found that there was no significant difference in TSS and COD removal in three parallel HFRBs with aspect ratios of 4:1, 10:1 and 30:1. In the EPA report [10] it is also described no difference between systems with aspect ratios of 1:4 and 4:1.

The combined effect of granular medium size and aspect ratio on organic matter and nutrient removal in HFRBs has not been studied extensively. In fact, there is no rational way of estimating the treatment efficiency of HFRBs with a particular medium size and aspect ratio. It is clear that the medium size can impact the contaminant removal efficiency, as this affects the specific surface available for the biofilm as well as affecting the hydrodynamic behaviour, which in turn affects the oxygen mass flux rates. However, studies with different-sized media in the range of 10–60 mm did not find significant different removal efficiencies [10].

To our knowledge, there are no reports which clearly delineate the effect of water depth on the treatment efficiency of HFRBs, because for practical experience the depth is normally set up around 0.6 m. The reasoning behind the 0.6 m depth is that it is the maximum depth at which the reeds will grow [11]. Water depth probably influences treatment efficiency because this parameter is involved in the mass transfer coefficient of oxygen from the atmosphere to the water [12]. Moreover, water depth determines the fraction of water volume which is in contact with the roots of the reeds. In one study a slightly better BOD₅ removal was found at greater water depth, when comparing 0.45 m with 0.30 m in HFRBs operated with the same HLR [10]. It is unclear whether this result was due only to the higher hydraulic retention time (HRT) of the deeper system. In an evaluation of 14 different HFRBs it was found that the system with the lowest water depth (0.3 m) showed the second highest ammonia removal efficiency [13]. In this study, complete root penetration to the bottom of the beds was observed in the two systems with significant ammonia removal capacity (one had a depth of 0.76 m and the other, as stated above, had a depth of 0.3 m). According to the authors, these results strongly support the hypothesis that plant roots in HFRBs are the primary source of oxygen needed for nitrification. Another study also suggests that total root penetration of the media was critical to pollutant removal and recommends that the system depth be set equal to the maximum root depth of the wetland plant species used in each particular project [14].

The objective of the present study is to evaluate the effect of HLR, aspect ratio, granular medium size and water depth on the removal of COD, BOD₅, ammonia and dissolved reactive phosphorus (DRP) during the start up of HFRBs for the treatment of urban wastewater. A discussion is made on the contributions of aerobic and anaerobic respiration in the removal of BOD₅ in the HFRBs studied.

2. Materials and methods

The pilot plant used in this study treats part of the urban wastewater generated by the Can Suquet housing development in the municipality of Les Franqueses del Vallès (Barcelona, north-east Spain). The plant consists of eight parallel HFRBs constructed by excavation, with a plastic lining to avoid infiltration, and a sloped bottom (Fig. 1). All the beds have approximately the same surface area (54–56 m²) and their aspect ratio (length to width) varies in pairs. Pair A has an aspect ratio of 1:1, pair B of 1.5:1, pair C of 2:1 and pair D of 2.5:1. Furthermore, the size of the granular medium within each pair also varies. Thus, type 1 beds contain a coarse granitic gravel ($D_{50} = 10$ mm, $C_u = 1.6$) while type 2 beds contain finer granitic gravel ($D_{50} = 3.5$ mm, $C_u = 1.7$). The porosity of the granular medium is 39% for the coarse gravel and 40% for the finer. Types A–C beds have a slope that ranges from 0% to 1%, while the slope in type D beds is approximately 2.5%. Differences in slope were due to construction factors. Type D beds were constructed shallower in order to assess the effect of water depth on treatment efficiency. The water level was adjusted to 0.05 m under the surface in all beds, resulting in an average water depth of approximately 0.5 m for types A–C, and of 0.27 m for type D. All beds have three perforated tubes (0.1 m in diameter) inserted in the middle part of the gravel and uniformly distributed throughout the length of the bed, which allow intermediate samples to be obtained. The pilot plant began to operate in March 2001, when the beds were planted with the common reed (Phragmites australis, 3 plants/m²). The reeds covered the entire surface of all beds by the end of August 2001.

The urban wastewater is screened previously and flows into an Imhoff tank. This tank is connected to another from which the wastewater is pumped to a distribution chamber. The primary effluent is then divided by means of a wire with eight holes, and flows to each bed. Between the pump and the distribution chamber there is a valve and a flowmeter. For this study, four different total flowrates were used, resulting in four HLRs: 20, 27, 36 and 45 mm/day. The electromagnetic flowmeter has a totalizer that allows the total volume of water pumped between sampling campaigns to be evaluated and checked against the theoretical volume, taking into account the selected flow. In all the campaigns the error was lower than 15%. With all these systems and mechanisms the HLR was well adjusted and known. It was verified that the influent contaminant results displayed a normal distribution. The average BOD₅ concentrations of influent for the four periods
during which the plant was operated with different HLRs were quite similar: 120 \pm 42, 120 \pm 7, 140 \pm 23 and 150 \pm 37\, mg/L. As a result, the differences in surface organic loading during the four periods were mainly related to the HLRs. The average organic surface loading rates were as follows: 2.4 \pm 1.1, 2.9 \pm 0.1, 4.6 \pm 0.7 and 7.1 \pm 1.6\, g\,BOD_5/m^2d for the periods during which the plant was operated with HLRs of 20, 27, 36 and 45\, mm/d, respectively.

Sampling campaigns were conducted approximately on a weekly basis from May 2001 to January 2002. A total of 25 campaigns were carried out, of which 10 correspond to an HLR of 20, 2–27, 7–36 and 6–45\, mm/d. Grab samples of the primary effluent (from the distribution chamber) and the effluents of the beds were taken using plastic containers. pH measurements were taken in situ using a Crison 506 portable pH-meter. TSS, COD, carbonaceous \( \text{BOD}_5 \), ammonia and DRP analyses were done immediately using conventional methods [15].

Two additional sampling campaigns were carried out in July and August (when the HLR was 36\, mm/d) in order to obtain the dissolved oxygen (DO), and oxidation and reduction potential \( (E) \) profiles in all the perforated tubes of all beds. Profiles were carried out by submerging DO and \( E \) probes attached to an arm (moved slowly at less than 5 cm/min) from the top to the bottom of the beds. DO was measured using the YSI 58 oxymeter with a high-sensitivity membrane in order to reduce the background signal. \( E \) was measured using a Crison 506 with a platinum electrode and an Ag/AgCl reference system. Moreover, during these two campaigns influent and effluent samples were taken and analysed for soluble COD, ammonia, nitrite and electron acceptors (DO, nitrate and sulphate) using the methods described in APHA-AWWA-WPCF [15] to qualitatively evaluate the relative contribution of the aerobic and anaerobic respiration to the degradation of organic matter.

Statistical procedures were carried out using the SYSTAT statistical software package. One- and three-way ANOVA methods were used to check the influence of each factor considered (HLR; type of bed, which includes aspect ratio and water depth; and granular media) for every water quality parameter of the effluent and to evaluate interactions between factors. For all ANOVA tests it was checked that the variables were normally distributed. Otherwise, the variables were log-transformed. The Bonferroni method was used to test all pairwise comparisons of marginal averages.

3. Results

Table 1 shows the main physical and chemical properties of the primary effluent. The pH of the effluents was around 7.28, depending on the bed, and ranged between 7.00 and 7.62. Average effluent pH from different beds was not statistically different. TSS removal was excellent in all beds (around 98\%) because the effluent concentration was usually lower than 2\, mg/L. There were no apparent differences between beds in TSS removal.
3.1. Organic matter removal

Beds A–C had very similar average COD removals (Table 2). Beds of type D had the higher average COD removals, particularly bed D2. In fact, only bed D2 showed statistical differences from the averages of the all other beds ($p < 0.05$) in the effluent COD concentrations when a pairwise comparison was carried out. All the factors studied caused significant statistical differences in the average effluent COD concentrations (Table 3). The ANOVA test did not reveal significant interactions between the three factors for the COD. As the HLR increased, the effluent COD concentrations also increased generally, regardless of whether the results of type D beds were included in the data analysis (Fig. 2a). The average corresponding to an HLR of 27 mm/d was higher than that observed for an HLR of 20 mm/d probably due to the fact that the beds were operated at an HLR of 27 mm/d for a very short period. The pairwise comparison of the effluents’ COD concentrations showed significant differences ($p < 0.05$) between HLR of 27 and 35, and 45 mm/d with all the rest.

The type D beds had a significantly lower effluent COD concentration than the rest (Fig. 2b). The pairwise comparison of the COD concentrations only yielded significant differences between type D beds and all the rest. The beds containing a coarser gravel had a slightly higher average effluent COD concentration and statistically different than the rest (Fig. 2c). Nevertheless, when the data of type D beds were not used, the average COD concentrations were not statistically different (Table 3). The inclusion of bed D2 in the fine gravel group is therefore the factor that determines the differences when all data are used.

Types A–C beds showed a clearly lower overall BOD$_5$ removal than type D beds (Table 2). The pairwise comparison of the effluent BOD$_5$ concentration averages yielded significant statistical differences between bed D2 and the rest (with the exception of D1). Also, bed D1 showed significant differences when compared to A2, B1 and C2.

The HLR and the type of bed caused significant statistical differences in the average effluent BOD$_5$ concentration (Table 3, Figs. 2d and e). The average BOD$_5$ concentrations obtained for the two kinds of medium size were not significantly different (Fig. 2f). There were not significant interactions between the factors. The effect of the HLR on the average effluent BOD$_5$ concentrations was very similar to that observed for COD: when the HLR increased it resulted in a higher effluent BOD$_5$ concentration. The evaluation of the pairwise comparison showed that the highest HLR (45 mm/d) produced averages statistically different from all the rest. Moreover, HLRS of 20 and 36 mm/d also resulted in statistically different BOD$_5$ concentrations. When data from type D beds was not included in the ANOVA method, the results were equal to those obtained using all the data.

3.2. Nutrient removal

Ammonia removal was generally low in all beds. Beds D1 and D2 showed the highest removal (Table 2). The

![Table 1](https://example.com/tables/table1.png)

Table 1
Main physical and chemical properties of the primary effluent pumped into the HFRB system

<table>
<thead>
<tr>
<th>Parameter</th>
<th>n</th>
<th>Average</th>
<th>Std. deviation</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>22</td>
<td>7.37</td>
<td>0.18</td>
<td>7.05–7.70</td>
</tr>
<tr>
<td>TSS, mg/L</td>
<td>16</td>
<td>110</td>
<td>40</td>
<td>50–180</td>
</tr>
<tr>
<td>COD, mg/L</td>
<td>22</td>
<td>260</td>
<td>60</td>
<td>160–400</td>
</tr>
<tr>
<td>BOD$_5$, mg/L</td>
<td>22</td>
<td>130</td>
<td>40</td>
<td>60–220</td>
</tr>
<tr>
<td>NH$_3$, mg N/L</td>
<td>24</td>
<td>61.5</td>
<td>7.0</td>
<td>48.0–75.4</td>
</tr>
<tr>
<td>DRP, mg P/L</td>
<td>24</td>
<td>10.5</td>
<td>2.3</td>
<td>7.0–15.3</td>
</tr>
</tbody>
</table>

Note: $n$ is the number of readings used for the calculations.

![Table 2](https://example.com/tables/table2.png)

Table 2
Overall averages and standard deviations of the water quality parameters in the effluents for all the HFRBs

<table>
<thead>
<tr>
<th>Parameter</th>
<th>A1</th>
<th>A2</th>
<th>B1</th>
<th>B2</th>
<th>C1</th>
<th>C2</th>
<th>D1</th>
<th>D2</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD, mg/L</td>
<td>98±25 (62)</td>
<td>97±23 (63)</td>
<td>100±24 (62)</td>
<td>91±26 (65)</td>
<td>94±26 (64)</td>
<td>92±23 (65)</td>
<td>79±21 (70)</td>
<td>55±17 (79)</td>
</tr>
<tr>
<td>BOD$_5$, mg/L</td>
<td>56±21 (57)</td>
<td>60±19 (53)</td>
<td>59±21 (54)</td>
<td>52±19 (60)</td>
<td>53±21 (59)</td>
<td>57±20 (56)</td>
<td>37±21 (71)</td>
<td>21±25 (84)</td>
</tr>
<tr>
<td>NH$_3$, mg N/L</td>
<td>47.0±9.9 (24)</td>
<td>45.6±8.1 (26)</td>
<td>46.6±8.6 (24)</td>
<td>44.1±7.4 (28)</td>
<td>44.8±8.3 (27)</td>
<td>42.9±6.7 (30)</td>
<td>38.4±7.7 (38)</td>
<td>30.1±8.4 (51)</td>
</tr>
<tr>
<td>DRP, mg P/L</td>
<td>10.6±2.6 a (3)</td>
<td>10.2±2.2 (2)</td>
<td>10.4±2.4 (2)</td>
<td>9.7±2.1 (8)</td>
<td>10.1±1.7 (4)</td>
<td>9.6±1.6 (9)</td>
<td>9.5±2.6 (10)</td>
<td>8.2±2.3 (22)</td>
</tr>
</tbody>
</table>

Note: Average percentage removals are shown in parenthesis.

aNo removal.
A pairwise comparison of the effluent ammonia concentration averages showed significant statistical differences between bed D2 and all the rest. Also, bed D1 had statistically significantly lower ammonia concentrations than A1 and B1. In contrast to COD and BOD$_5$, the HLR did not produce statistically significant differences in the average effluent ammonia concentration (Table 3, Fig. 2g). Type D beds produced significant differences in
the average effluent ammonia concentrations (Table 3, Fig. 2h). The beds containing coarser gravel had a significantly slightly higher average effluent ammonia concentration (Table 3, Fig. 2i). Nevertheless, when the data of type D beds was not included, the average ammonia concentrations were not statistically different. The inclusion of bed D2 in the fine gravel group is therefore the factor that determined the differences observed. There were not significant interactions between the three factors tested.

The average removal of DRP was low in all beds (Table 2). DRP removal in bed D2 was higher than in the other beds. The pairwise comparison of the averages of the effluent DRP concentrations showed significant statistical differences between bed D2 and beds A1, A2 and B1. There were not significant interactions between the three factors tested. The HLR caused significant statistical differences in the average effluent DRP (Table 3); however, there was not a clear trend (Fig. 2j). The evaluation of the pairwise comparison of the averages showed that an HLR of 20 mm/d produced an effluent DRP average statistically different than those of HLRs of 27 and 36 mm/d.

The type of bed produced statistically significant effluent DRP concentrations (Table 3, Fig. 2k). The pairwise comparison of the DRP concentrations in the effluent showed significant differences between beds of types D, A and B. Beds containing a coarser gravel had a significant slightly higher average effluent DRP concentration (Table 3, Fig. 2l). Nevertheless, when the data of D type beds was not included in the ANOVA method, the averages were not statistically different.

### 3.3. DO and E profiles

DO concentrations were extremely low in all beds (Table 4). Unfortunately, within that range of concentration the relative error of the DO measurement is high (from 100% to 200%), so the values showed can only be used qualitatively. The highest DO concentrations were observed in beds of type D. Average $E$ values were very low indicating very reducing conditions in all beds (Table 4). The $E$ values were higher in type D beds.

Fig. 3 shows the $E$ profiles obtained from the perforated tubes of beds A2 and D2 during the July campaign. The $E$ decreases with depth in both beds because of the stronger reducing conditions with depth. Also, it can be seen that $E$ variation with depth is lower in bed D2. Vertical redox profiles clearly vary in response to distance from the point of wastewater inlet in the case of bed A2. Therefore, the water becomes more oxidised along the length of this bed. Vertical $E$ profiles of beds A1, B and C were similar to that of A2, and the profiles of D1 were similar to that of D2.

### 3.4. Electron acceptors

Electron acceptors entering the HFRBs were measured in the two additional campaigns (July and August). The DO concentration of the influent was lower than 0.10 mg/L. Nitrate was not detected in either of the two campaigns (in the influent and the effluents). Thus, the influent contribution of these two electron acceptors to the oxidation of organic matter was insignificant. Effluent sulphate concentration was systematically lower than in the influent (Table 5) and therefore contributed to the oxidation of organic matter.

From the sulphate, ammonia and COD concentration data shown in Table 5 it is possible to make a numerical estimation of the contribution of sulphate reduction and denitrification to the oxidation of organic matter. Although nitrates were not detected in the influent, it

<table>
<thead>
<tr>
<th>Parameter</th>
<th>A1</th>
<th>A2</th>
<th>B1</th>
<th>B2</th>
<th>C1</th>
<th>C2</th>
<th>D1</th>
<th>D2</th>
</tr>
</thead>
<tbody>
<tr>
<td>DO, mg/L</td>
<td>0.11/0.09</td>
<td>0.08/0.08</td>
<td>0.07/0.08</td>
<td>0.08/0.10</td>
<td>0.12/0.07</td>
<td>0.09/0.07</td>
<td>0.09/0.14</td>
<td>0.12/0.17</td>
</tr>
<tr>
<td>$E$, mV</td>
<td>−370</td>
<td>−358</td>
<td>−361</td>
<td>−387</td>
<td>−390</td>
<td>−371</td>
<td>−351</td>
<td>−338</td>
</tr>
</tbody>
</table>
is assumed that most of the ammonia was removed by nitrification–denitrification reactions (in particular in type D beds). The amount of soluble organic matter removed is the difference between influent and effluent COD, and is expressed in glucose equivalents (to calculate this one must multiply the result by a factor of 0.94). The amount of organic matter removed by sulphate reduction is estimated as the difference between influent and effluent sulphate concentration, and is transformed to glucose equivalents by means of the following stoichiometric equation [16]:

\[
\text{C}_6\text{H}_{12}\text{O}_6 + 3\text{SO}_4^{2-} + 6\text{H}^+ \rightarrow 6\text{CO}_2 + 3\text{H}_2\text{S} + 6\text{H}_2\text{O}.
\]

This gives a sulphate-to-glucose mass ratio of 1.60. The organic matter removed by denitrification, expressed in glucose equivalents, can be calculated as the difference between the influent and effluent ammonia content, assuming that 10% of the ammonia is removed by biological assimilation [13] and that the rest is oxidised to nitrate. Finally, nitrate is stoichiometrically related to glucose in the following equation [16]:

\[
\text{NH}_4^+ + 2\text{O}_2 \rightarrow \text{NO}_3^- + \text{H}_2\text{O} + 2\text{H}^+.
\]

\[
\text{C}_6\text{H}_{12}\text{O}_6 + 4\text{NO}_3^- \rightarrow 6\text{CO}_2 + 6\text{H}_2\text{O} + 2\text{N}_2 + 4e^-.
\]

These reactions give a nitrate to ammonia (in fact, ammonium, and note that in Table 5 is expressed in nitrogen) mass ratio of 4.43 and a nitrate to glucose ratio of 1.38. The results for the estimated organic matter removed by sulphate reduction and denitrification and the total soluble organic matter removal in each bed are shown in Table 5 in brackets. Sulphate reduction accounted in general for higher organic matter removal in deeper beds than in lower beds (types A–C). Only beds A1, A2 and B1 in August had a lower amount of organic matter removed by sulphate reduction than type D beds; however, these beds had a very low removal efficiency in comparison to the rest. In fact, if the importance of sulphate reduction is evaluated taken into account the total soluble organic matter removed, then it is clear that sulphate reduction was less important in type D beds. In both campaigns, denitrification was a significant mechanism for organic matter removal in type D beds, but not in the other beds. In fact, in beds A–C, the ammonia concentration was higher in the effluents than in the influent. The amount of organic matter removed by biochemical processes other than sulphate reduction and denitrification was more significant in August than in July. Note that in that campaign the sum of the organic matter removed by the reactions studied was in general clearly lower than the total amount of soluble organic matter that was removed. This is an enigmatic aspect of the investigation, as the beds were operated during both campaigns with the same HLR (36 mm/d) and very similar water temperatures (within the range of 20.8–23.6°C, considering all beds). This may be related to the fact that the influent soluble COD was higher in August.

### 4. Discussion

From the four factors evaluated, HLR, aspect ratio, granular medium size and depth, only the first and the last yield statistical differences in the performance of the HFRBs. This is in agreement with previous works listed in the introduction section. Nevertheless, in one study a better BOD$_5$ removal was found at greater depth [10], while our results show the opposite trend. There is no apparent reason for this difference. HLR gives significant differences for three of the four contaminants evaluated (COD, BOD$_5$ and DRP), and depth gives differences for all contaminants. Thus, HLR (in fact the surface loading rate) and depth are both very important factors in determining the performance of reed bed systems during the start up of the facilities and in the range of operational conditions and contaminants tested in this study.

The importance of HLR and water depth in the performance of the beds can be assessed by calculating

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Influent</th>
<th>A1</th>
<th>A2</th>
<th>B1</th>
<th>B2</th>
<th>C1</th>
<th>C2</th>
<th>D1</th>
<th>D2</th>
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</thead>
<tbody>
<tr>
<td>SO$_4^{2-}$, mg/L (July)</td>
<td>72</td>
<td>16 (35)</td>
<td>26 (29)</td>
<td>24 (30)</td>
<td>9 (39)</td>
<td>10 (39)</td>
<td>13 (37)</td>
<td>40 (20)</td>
<td>55 (11)</td>
</tr>
<tr>
<td>NH$_3$, mg N/L (July)</td>
<td>27.4</td>
<td>36.5 (—)</td>
<td>36.0 (—)</td>
<td>31.8 (—)</td>
<td>36 (—)</td>
<td>29.3 (—)</td>
<td>35.8 (—)</td>
<td>18.5 (26)</td>
<td>16.0 (33)</td>
</tr>
<tr>
<td>COD, mg/L (July)</td>
<td>88</td>
<td>48 (38)</td>
<td>57 (29)</td>
<td>51 (35)</td>
<td>43 (42)</td>
<td>38 (47)</td>
<td>46 (39)</td>
<td>36 (49)</td>
<td>39 (46)</td>
</tr>
<tr>
<td>SO$_4^{2-}$, mg/L (August)</td>
<td>73</td>
<td>51 (14)</td>
<td>51 (14)</td>
<td>51 (14)</td>
<td>17 (35)</td>
<td>20 (33)</td>
<td>22 (32)</td>
<td>37 (23)</td>
<td>47 (16)</td>
</tr>
<tr>
<td>NH$_3$, mg N/L (August)</td>
<td>36.0</td>
<td>41.8 (—)</td>
<td>38.8 (—)</td>
<td>40.2 (—)</td>
<td>36.2 (—)</td>
<td>38.6 (—)</td>
<td>37.8 (—)</td>
<td>28.2 (23)</td>
<td>26.0 (29)</td>
</tr>
<tr>
<td>COD, mg/L (August)</td>
<td>130</td>
<td>100 (28)</td>
<td>100 (28)</td>
<td>90 (38)</td>
<td>49 (76)</td>
<td>46 (79)</td>
<td>43 (82)</td>
<td>43 (82)</td>
<td>34 (90)</td>
</tr>
</tbody>
</table>

The values shown in brackets in the sulphate and ammonia rows are the amount of COD (expressed in glucose equivalents) that has been removed by sulphate reduction and denitrification. The values in brackets in the COD row is the amount of soluble organic matter removed in each bed expressed in glucose equivalents. For details on calculations see the text.
the contaminant removal kinetic constants. Contaminant removal in HFRB beds can be described assuming that they perform as ideal plug flow reactors with first-order kinetic constants [13,17]:

$$\frac{C_1}{C_0} = \exp(-kV_1 t),$$

where $C_0$ and $C_1$ are the influent and effluent contaminant concentration, $kV_1$ is the first-order volumetric kinetic constant and $t$ is the hydraulic retention time. $t$ is $V/Q$, where $V$ is the nominal volume ($V = \varepsilon h A$, $\varepsilon$ is the porosity of the medium, $h$ is the water depth and $A$ is the surface area) and $Q$ is the flow. Lumping $kV_1 h c = k_1$, taking into account that $A/Q$ is the inverse of HLR, and making the necessary arrangements, the model transforms from volumetric to areal [1]:

$$\ln \frac{C_0}{C_1} = k_1/HLR$$

We have estimated the values of the parameters of the linear regressions of all four contaminants in each bed. For the calculations, the values of the inverse of HLR were the actual values recorded between sampling campaigns. In order to reduce scattering of the data, two measures were taken: (1) only data from campaigns without rain episodes since the previous campaign were considered and (2) only data from campaigns with an effluent water temperature higher than 18°C were considered. This last measure was taken for smoothing the effect of temperature.

The linear adjustments obtained for ammonia and DRP were not accurate because $R^2$ values were in general under 0.230 (estimated parameters are not shown). The lack of good adjustment of the model for nutrients is assumed to be due to the low removal observed. The linear adjustments were in general quite good for BOD$_5$ (Fig. 4). The results of the COD models are not shown because they had a very similar trend to BOD$_5$. The linear adjustments were worse for COD than for BOD$_5$ and ranged from 0.311 to 0.725. As can be seen in Fig. 4, the slopes of the linear regressions of beds A1 to C2 were similar, ranging from 0.011 to 0.020 m/d. Type D beds had steeper slopes than the rest because they were more effective for BOD$_5$ removal. Note that the slope coefficient of variation (CV) was in general under 0.230 (estimated parameters are not shown). The linear adjustments were in general quite good for BOD$_5$. The linear adjustments were worse for COD than for BOD$_5$ and ranged from 0.311 to 0.725. As can be seen in Fig. 4, the slopes of the linear regressions of beds A1 to C2 were similar, ranging from 0.011 to 0.020 m/d. Type D beds had steeper slopes than the rest because they were more effective for BOD$_5$ removal. Note that the slope coefficient of variation (CV) was in general under 0.230 (estimated parameters are not shown). The linear adjustments were in general quite good for BOD$_5$. The linear adjustments were worse for COD than for BOD$_5$ and ranged from 0.311 to 0.725. As can be seen in Fig. 4, the slopes of the linear regressions of beds A1 to C2 were similar, ranging from 0.011 to 0.020 m/d. Type D beds had steeper slopes than the rest because they were more effective for BOD$_5$ removal. Note that the slope coefficient of variation (CV) was in general under 0.230 (estimated parameters are not shown). The linear adjustments were in general quite good for BOD$_5$.

![Fig. 4. BOD$_5$ areal kinetic constants of the 8 pilot HFRBs. CV is the coefficient of variation of the slope that is calculated by dividing the standard error by the slope value and multiplying the result by 100.](image)

The comparison of the estimated BOD$_5$ areal kinetic constants with other published data coming from secondary HFRB beds [11,18] shows that type D beds have similar constant values. The estimated kinetic constants of types A–C beds are clearly lower (some by a factor that ranges from 3 to 10) than the published data. The relatively low efficiency observed in beds A–C may be related to the fact that the study was performed during the start-up of the facilities when the root system of the reeds was not completely developed in all beds. Gersberg et al. [14] and Cooper et al. [11] suggested that root penetration was critical to pollutant removal in HFRBs. In our study it is clear that the amount of water volume in contact with the roots was lower in types A–C beds because of their greater water depth. For the same reason, the mass transfer coefficient of oxygen from the atmosphere to the water by diffusion will also be lower in types A–C beds. In fact, the highest DO and E values were found in type D beds. Thus, it is possible that the efficiency of types A–C beds will improve comparatively more than in type D beds as the reeds’ root system becomes completely developed.

With the results of our study, it is possible to calculate that to obtain a BOD$_5$ removal of 90% (or more) of an influent that has a maximum BOD$_5$ concentration of
220 mg/L (Table 1), it is necessary not to exceed an organic surface loading of 5.9 g/m² day for beds with a depth of 0.27 m (type D) and of 2.0 g/m² day for beds with a depth of 0.5 m (types A–C). The maximum organic surface loading estimated for type D beds is consistent with the recommendations given by US EPA [10] of 6 g/m² day. However, the results of the present study were obtained based on the first year of establishment of the reeds and therefore the maximum organic surface loading rates may be different in subsequent years.

The better performance obtained in type D beds may seem somewhat contradictory because they operate with a lower hydraulic retention time (approximately from 2.5 to 5.5 days in the HLR range tested) than the rest (approximately from 4.5 to 10.0 days). However, the evaluation of the redox conditions and the behaviour of the electron acceptors suggest that these beds perform with more energetically favourable biochemical reactions that in turn increase the efficiency of the system [16]. Organic matter is removed in HFRBs through a combination of five possible mechanisms: sulphate reduction, denitrification, diffusion of air at the air and water interface (aerobic respiration), oxygen transport through macrophytes (aerobic respiration) and methanogenesis [19,12]. Denitrification was a significant mechanism for soluble COD removal in type D beds, but not in the other beds. The amount of soluble COD removed by sulphate reduction was in general higher in types A–C beds than in type D beds. Thus, the higher DO and E values found in type D beds are related to the fact that the mechanisms that remove soluble COD are different or have a different relative importance. The most important biochemical mechanism involved in the degradation of organic matter that was not removed by sulphate reduction and denitrification was probably methanogenesis, because low E values are favourable to this reaction [20]. Nevertheless, there is no experimental evidence to support this statement.

5. Conclusions

HLR (in fact the surface loading rate) and water depth are both very important factors in the performance of HFRBs, while aspect ratio and granular medium size have no clear effect on the removal of contaminants in the range of operational conditions tested.

The results indicate that the removal of contaminants decreases as the water depth increases from 0.27 to 0.5 m. Higher removal in beds with a lower water depth is related to the less reducing conditions observed in these types of beds. The reason for the less reducing conditions may be related to the shorter diffusion path between the surface and depth and/or the higher capacity for the reeds to supply oxygen to the root zone.

The behaviour of electron acceptors and the estimation of the relative importance of the biochemical pathways suggest that, as the water depth decreases, more varied reactions occur. In fact, denitrification was estimated to be a significant reaction for the removal of organic matter in 0.27 m beds, while it did not occur in 0.5 m beds.

According to the data from this study, to obtain a BOD₃ removal of 90% and to comply with European Union standards [21] it is necessary to avoid exceeding an organic surface loading of either 5.9 g/m² day (when using HFRBs with a depth of 0.27 m) or 2.0 g/m² day (when using HFRBs with a depth of 0.5 m). However, these results are from the first year of establishment of the reeds (2001), and the maximum surface organic loading rates may be different in subsequent years. This study is still underway at the present time (October 2003) in order to provide more accurate data as pertains to the normal operation of HFRBs.

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