Effect of diffusional mass transfer on the performance of horizontal subsurface flow constructed wetlands in tropical climate conditions


ABSTRACT

The effect of mass transfer on the removal rate constants of BOD$_5$, NH$_3$, NO$_3$ and TKN has been investigated in a Horizontal Subsurface Flow Constructed Wetland (HSSFCW) planted with Phragmites mauritianus. The plug flow model was assumed and the inlet and outlet concentrations were used to determine the observed removal rate constants. Mass transfer effects were studied by assessing the influence of interstitial velocity on pollutant removal rates in CW cells of different widths. The flow velocities varied between 3–46 m/d. Results indicate that the observed removal rate constants are highly influenced by the flow velocity. Correlation of dimensionless groups namely Reynolds Number (Re), Sherwood Number (Sh) and Schmidt Number (Sc) were applied and log–log plots of rate constants against velocity yielded straight lines with values $\beta = 0.87$ for BOD$_5$, 1.88 for NH$_3$, 1.20 for NO$_3$ and 0.94 for TKN. The correlation matched the expected for packed beds although the constant $\beta$ was higher than expected for low Reynolds numbers. These results indicate that the design values of rate constants used to size wetlands are influenced by flow velocity. This paper suggests the incorporation of mass transfer into CW design procedures in order to improve the performance of CW systems and reduce land requirements.

Key words | constructed wetlands, mass transfer, tropical climate

INTRODUCTION

Research studies on the use of constructed wetlands for wastewater treatment began in Europe in the 1950s and in the US in the late 1960s (Marks 1999). In Tanzania, the technology was introduced in the late 1990s by the Waste Stabilization Ponds and Constructed Wetland Research Group based at the University of Dar es Salaam by first investing in research and later on full scale application in different parts of Tanzania. To date, more than 10 units of variable sizes have been built to treat domestic wastewater.

CWs are designed and established to treat wastewater by mimicking the chemical, biological and physical processes that occur in natural wetlands (Kadlec & Knight 1996). They fall into two broad categories, the surface flow CWs and the subsurface flow CWs. In surface flow wetlands, wastewater flows over soil in which plants are grown. Water depth is usually less than 0.4 m. In subsurface flow wetlands wastewater flows through a medium, which may be soil, gravel or other porous material, which is typically less than 0.6 m deep (Kadlec & Knight 1996). However, due to low hydraulic conductivity of soil, which can lead to clogging, gravel is normally used in horizontal subsurface flow CW. Potential applications of CWs range from secondary treatment of wastewater from various sources, to polishing tertiary treated wastewater and diffuse pollution. Successful case studies showed that CWs are effective in the treatment of municipal wastewater (Vymazal 1999; Mustafa 2009; Scholz et al. 2010), industrial wastewaters (Maja et al. 2009; Njau & Renalda 2010) and wastewaters containing heavy metals (Cheng et al. 2002; Yeh et al. 2009).

Current state-of-the-art design models have been developed based on first-order equations trying to capture the inherent wetland complexity with only a few parameters. The latest variant of this, the so-called P-k-C$^*$ model, is described in detail in Kadlec & Wallace (2009). There is

large uncertainty in these model predictions that makes engineers apply high safety factors thus increasing the area of the CWs (Rousseau et al. 2009). The systems in Tanzania were designed using the first-order, irreversible pollutant reduction removal plug flow model for wastewater treatment as suggested by Reed et al. (1995). The equations of Reed et al. (1995) are based on the first-order plug flow (Fogler 1999) assumption for those pollutants that are removed primarily by biological processes, including biochemical oxygen demand (BOD), ammonia (NH₄) and nitrate (NO₃). While the temperature dependent rate constants for different wastewater parameters are usually used in the plug flow design approach, research studies reveal that the performance of the systems is influenced by hydrodynamics in the system especially in tropical areas where the difference between cold and hot months is insignificant. The baffled CWs, for example, have been observed to perform better than un-baffled CW systems in Tanzania. Yet, mass transfer has not received deserved attention as a limiting process in CWs in pollutant removal hence minimal consideration in the design. The objective of this paper therefore is to appraise the contribution of mass transfer on the overall performance of the subsurface flow CWs thereby examining the influence of interstitial velocity on the diffusional mass transfer in a Horizontal Sub-Surface Flow Constructed Wetland (HSSFCW) through different CW widths.

**PROCESSES IN CONSTRUCTED WETLANDS**

Reaction in CWs is heterogeneous in nature since the dissolved species are removed by a bio reaction occurring on a solid substrate. The overall pollutant removal involves transport as well as bio reaction processes. Transport process is the first step in the removal mechanisms (Kadlec & Knight 1996). In subsurface flow wetlands the solid surfaces such as gravels and plant roots, contains the biofilms responsible for microbial processing. The dissolved materials, such as BOD, must move from bulk of water to the vicinity of the solid surface, then diffuse through a stagnant water layer to the surface, and then penetrate the biofilm while undergoing chemical transformation (Figure 1). Convection and diffusion processes are involved in the transport of reactive species. However, diffusion through the stagnant layer on the surface of the biofilm is a major contributor to the overall resistance to the mass transfer.

The reaction and diffusion within the biofilm are not expected to depend on the velocity in the bulk of the CW water because the film is not subjected to flushing. However, the transfer of the constituent from the bulk water to the surface of the biofilm is expected to be influenced by velocity, with higher velocities giving more turbulences and better transfer. In the limit of very rapid transfer from water to biofilm surface, the overall rate is dependent solely on process within the biofilm. Wetlands do not typically operate under such a high velocity, hence speed of water (which is defined by the hydraulic conductivity K), is a variable of interest (Katima 2005).

**MATERIALS AND METHOD**

Some data were generated from existing CWs that are operational in Tanzania. Three existing CWs were studied; one based at the University of Dar es Salaam (UDSM) receiving its wastewater from an anaerobic pond, another based at Ruaha Secondary School in Iringa Tanzania receiving wastewater from a septic tank and another serving Klerruu Teachers College in Iringa Tanzania receiving wastewater from an anaerobic pond. The wastewater samples were collected and analyzed for biochemical oxygen demand (BOD₅). The samples for BOD₅ analysis were collected from inlet of the wetland, within the wetland, and outlet of the wetland. At each sample section, one sample was taken. The inlet samples were taken initially and outlet samples were taken based on the actual retention times determined earlier by the tracer studies (Cornel 2004; Peters 2004) and calculated for each particular section. The required wastewater flow rates into the CWs were regulated manually by valves and measured manually. Table 1 below presents the characteristics of the existing wetlands studied.

Other data were obtained from an experimental setup established at the University of Dar es Salaam to examine the influence of velocity, and therefore mass transfer, on the removal of a wide range of wastewater parameters. Six
HSSFCW cells, rectangular in shape, different in width (ranging from 0.25–2.0 m), with the same length (6.9 m) and same total depth (1 m) were built downstream of the maturation pond with bottom slope of 1.12% lined with concrete. They all together form a total surface area of 41.4 m². The filling medium in each cell consisted of clean and graded gravel with size ranging from 12–20 mm uniformly packed to cover a depth of 0.6 m from the bottom of the bed.

The cells were planted with *Phragmites mauritianus* at an initial planting density of 3 plants/m². Sampling was started three months after planting. Table 2 shows the characteristics of the wetland cells used in this study while Figure 2 and Plate 1 present the layout of CW cells and a photo of the setup, respectively.

All six cells were designed to receive wastewater from the same source (maturation pond). Initially, the maximum flow rates (that caused surface ponding) were established for each CW cell to avoid operating the systems with flooding conditions. The maximum flow rates ranged from 2–16 m³/day corresponding to hydraulic loading rate (HLR) of 1,150–1,255 mm/day respectively. The cells were then operated at different flow rates below the maximum in order to establish different wastewater velocities within the cells knowing the media porosity (35%) and the cross section area of each CW cell. The flow rates were regulated by valves and measures manually by means of measuring cylinder and stop watch. An observation pipe was established at the centre of each cell and was used to measure the actual depth of water (effective depth) in each CW cell during sampling.

Table 2  | Characteristics of an experimental setup at the University of Dar es Salaam

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Cell 01</th>
<th>Cell 02</th>
<th>Cell 03</th>
<th>Cell 04</th>
<th>Cell 05</th>
<th>Cell 06</th>
</tr>
</thead>
<tbody>
<tr>
<td>Length (m)</td>
<td>6.9</td>
<td>6.9</td>
<td>6.9</td>
<td>6.9</td>
<td>6.9</td>
<td>6.9</td>
</tr>
<tr>
<td>Width (m)</td>
<td>2.0</td>
<td>1.5</td>
<td>1.0</td>
<td>0.75</td>
<td>0.5</td>
<td>0.25</td>
</tr>
<tr>
<td>Substrate depth (m)</td>
<td>0.6</td>
<td>0.6</td>
<td>0.6</td>
<td>0.6</td>
<td>0.6</td>
<td>0.6</td>
</tr>
<tr>
<td>Slope (%)</td>
<td>1.12</td>
<td>1.12</td>
<td>1.12</td>
<td>1.12</td>
<td>1.12</td>
<td>1.12</td>
</tr>
</tbody>
</table>

The systems were operated from November 2009 to June 2010. The operational flow rates were varied from 0.47–12.6 m³/day corresponding to HLR of 90.82–1,095.65 mm/day respectively. Wastewater samples were collected from the inlet and outlets of each cell. The samples were analyzed for BOD₅, total suspended solids (TSS), chemical oxygen
demand (COD), nitrate nitrogen (NO$_3$–N), ammonia (NH$_3$–N) and total Kjeldahl nitrogen (TKN). The analyzed physical parameters include temperature, salinity, pH, electrical conductivity (EC) and total dissolved solids (TDS). Analyses were done according to the Standard Method for the Examination of Wastewater (APHA/AWWA/WEF 1998).

TSS was analyzed by a standard filtration and evaporation method. The TSS concentration were determined and reported in mg/l. The TSS removal rates were determined and plotted against velocities. BOD$_5$ concentrations on the other side were determined using the OxiTop Control system. 250 ml for inlet samples and 365 ml for outlet samples were filtered and put in the OxiTop bottles and incubated at 20 °C. Note that analyses were done on filtered samples in order to remove algal solids (and hence algal BOD). Filtration was done through standard glass fibre of the type normally used for suspended solids determination. After 5 days of incubation the BOD$_5$ concentration of the sample were obtained in mg/l BOD. The BOD removal rate constants were calculated based on the plug flow model and were plotted against velocity. NH$_3$–N was determined through a standard titrimetric method on samples carried through preliminary distillation. The NH$_3$–N removal rate constants were determined based on plug flow mode and plotted against velocity. NO$_3$–N on the other hand was analyzed by using a standard cadmium reduction. The NO$_3$–N removal rate constants were calculated based on the plug flow model. The NO$_3$–N removal rate constants were plotted against velocity.

Evaluation of results

Design equation for HSSFCW assumes plug flow characteristics. For BOD$_5$, NH$_3$, NO$_3$ and TKN

$$A_s = Q \left( \ln C_o - \ln C_e \right) \frac{1}{K_T h e}$$

(1)

where $A_s$ is the surface area of the wetland, m$^2$, $C_e$ the effluent BOD concentration, mg/l, $C_o$ the influent BOD concentration, mg/l, $K_T$ the temperature-dependent volumetric first-order reaction rate constant, d$^{-1}$, $\tau$ the hydraulic residence time, d.

Since all parameters are known, observed values of $K_T$ were obtained by substituting the values of concentrations obtained from analysis in Equation (1).

The observed values of $K_T$ can be partitioned into the mass transfer coefficient $k_m$ and the biochemical reaction rate constant $k_r$ as follows (Njau 1998; Fogler 1999)

$$\frac{1}{K_T} = \frac{1}{k_m} + \frac{1}{k_r}$$

(2)

If $k_m = k_r$, the system will be mass transfer controlled and the observed $K_T = k_m$.

It is common to describe mass transport to particles inside the packed bed via dimensionless group correlations. These groups include the Sherwood number (Sh), the Schmidt number (Sc) and the Reynolds number (Re) (Krishnan & Ibanez 1997; Wesseling & Krishna 2000). The Sherwood number describes the mass transfer (Equation (2)).

$$Sh = \frac{k_m d}{D_w}$$

(3)

where $k_m$ is the mass transfer coefficient (m/d or m/s), $d$ is the particle diameter (gravel diameter for SSF CW) (m) and $D_w$ is the diffusion coefficient (m$^2$/d, m$^2$/s). The Schmidt number gives the ratio between the momentum transport and mass transport by diffusion, i.e., it compares the rates of transport by convection and diffusion (Equation (3)).

$$Sc = \frac{\mu}{\rho D_w}$$

(4)

where $\mu$ is the viscosity of liquid (water in this case) (kg/ms) and $\rho$ is the density of liquid (water) (kg/m$^3$). The Reynolds number gives the ratio between the inertia and friction forces and characterises the type of liquid (Equation (4)). The Reynolds number describes the character of flow.

$$Re = \frac{\rho u d}{\mu}$$

(5)

where $u$ is the superficial velocity of water. The Sherwood number is a function of the Schmidt number and the Reynolds number (Krishnan & Ibanez 1997). The correlation is expressed in Equation (5)

$$k_m d \frac{D_w}{\rho} = \beta \left( \frac{\mu}{\rho D_w} \right)^{\alpha} \left( \frac{\rho u d}{\mu} \right)^{\beta}$$

(6)

The value of the exponent $\alpha$ is therefore of less practical interest and is usually taken to be 1/3 (Krishnan & Ibanez 1997). For packed bed reactors Krishnan & Ibanez (1997) gave the correlation as expressed in Equation (6) for
100 ≤ Re ≤ 10,000
\[ Sh = 0.32S_1^{1/3}Re^{2/3} \]  \hspace{1cm} (7)

Wesselingh & Krishna (2000) gave the correlation as shown in Equation (7) for packed bed reactors

\[ Sh = \frac{0.34}{\varepsilon}Re^{2/3}Sc^{1/3} \]  \hspace{1cm} (8)

From Equation (5), \( \rho, \mu, C, d \) and \( D_w \) are constants at a fixed temperature thus, Equation (8) can be obtained (Njau 1998).

\[ k_m = C'u^\beta \]  \hspace{1cm} (9)

where \( C' \) and \( \beta \) are constants. The most important constant is \( \beta \). A log–log plot of \( k_m \) vs \( u \) from Equation (8) yields the exponent \( \beta \). As reported above this value is taken to be 2/3 for packed beds. Genders & Weinberg (1992), gave typical values of \( \beta \) for laminar flow as \( 0.3 < \beta < 0.5 \) and for higher flows (turbulent) as \( 0.45 < \beta < 0.8 \). Laminar or turbulent flow can be checked by determination of Reynolds number. For small values of \( Re < 1 \), the flow is laminar; for large values \( Re > 1 \) it is turbulent (Wesselingh & Krishna 2000). For tubes, with \( Re < 2000 \), the flow is laminar and \( > 2000 \) the flow is turbulent. Nepf (1999), reported that for \( Re < 200 \) the flow is laminar, otherwise the flow is turbulent.

**RESULTS AND DISCUSSION**

**Influence of velocity on the removal of biochemical oxygen demand (BOD\(_5\))**

Figure 3 shows the variation of \( K_{BOD} \) with velocity. It is clear from this graph that \( K_{BOD} \) increases with increasing \( u \). Log–log plot of \( K_{BOD} \) vs \( u \) yielded a straight line with slope of 0.87. Results for four field constructed wetlands showed a slope of 0.86. These results are in close agreement.

The results reveal that BOD\(_5\) removal rate is increased with increased interstitial velocity. While the BOD\(_5\) volumetric removal rate constant (\( K_{BOD} \)) as a function of temperature cannot exceed 2.8/day in Tanzania range of temperatures for example, the BOD\(_5\) removal rate constants obtained in this study reached a value of 4.0/day within the flow velocities studied. Within the range of velocities studied the BOD\(_5\) removal rate constant has been found to vary with velocity according to the following equation.

\[ K_{BOD} = 0.12u^{0.87} \]  \hspace{1cm} (10)

**Influence of velocity on the removal of different forms of nitrogen**

Ammonia. Figure 4 shows the dependency of \( K_{NH3} \) on interstitial velocity. Log–log plot of \( K_{NH3} \) against \( u \) yielded a straight lines showing two distinct regions. At velocities lower than 2.4 m/s the slope of the line was 1.0. while at \( u < 2.4 \) the slope was 0.81. The correlation at higher velocities was poor. Again like with \( K_{BOD} \) there is a strong dependency of \( K_{NH3} \) on velocity. During design of constructed wetland, \( K_{NH3} \) is a very critical parameter because the temperature dependent removal rate constants for NH\(_3\) are always very low. In Tanzania for example, where an annual maximum temperature is 36 °C in some areas, the maximum temperature dependent ammonia removal rate constant is only 0.26/day. Because of its low value ammonia will be the parameter that controls the design area; huge land will be required for ammonia removal. In these experiments, values of \( K_{NH3} \) as high as 1.26/day have been observed within the velocity range tested in these study.
From these results, the ammonia removal rate constant has been found to vary with velocity according to the following equation

\[
K_{\text{NH}_3} = \begin{cases} 
0.004u & \text{for } u \leq 2.4 \text{ m/s} \\
0.051u^{0.81} & \text{for } u > 2.4 \text{ m/s}
\end{cases}
\]  \( (11) \)

Nitrate. Log–log plot of \( K_{\text{NO}_3} \) against \( u \) yielded a straight line with slope of 1.20 (Figure 5). Values of \( K_{\text{NO}_3} \) of up to 20.74/day have been obtained within the velocity range used in this study. This is much higher than the values of temperature dependent rate constant of 9.36/day calculated at highest ambient temperature in Tanzania. From these results, the nitrate removal rate constant has been found to vary with velocity according to the following equation

\[
K_{\text{NO}_3} = 0.170u^{1.20}
\]  \( (12) \)

Total Kjeldahl Nitrogen. Log–log plot of \( K_{\text{TKN}} \) against \( u \) yielded a straight line with slope of 0.94 (Figure 6). Values of \( K_{\text{TKN}} \) of up to 2.42/day have been obtained within the velocity range used in this study. From these results, the nitrate removal rate constant has been found to vary with velocity according to the following equation

\[
K_{\text{TKN}} = 0.048u^{0.94}
\]  \( (13) \)

Influence of velocity on the removal of total suspended solids

While Reed et al. (1995) design approach suggests a separate equation for TSS removal based on the assumption that TSS is not removed primarily by biological processes, Kadlec & Knight (1996) design method presume a first-order decay, plug flow model for all pollutants including TSS. As per Reed, TSS removal is a function of influent concentrations, effluent concentrations and HLR. In this experiment, when \( K_{\text{TSS}} \) obtained by assuming first-order decay was plotted against \( u \), a good fit was obtained at low velocities less than 2.5 m/day while the data were very scattered at higher velocities (Figure 7). An \( R^2 \) of 0.97 was obtained for the low velocity data.

CONCLUSION

The results of this study showed significant relationship between velocity and removal rate constants for BOD\(_5\), NO\(_3\), NH\(_3\) and TKN for domestic wastewater. Higher velocities of wastewater through the CW substrate have been observed to bring about higher pollutant removal rates. Thus, it can be concluded that the performance of HSSFCW depends on the mass transfer processes and the rate constants for different wastewater parameters depend on velocity according to the equation of the form \( K = Cu^\beta \). Table 3 shows the summary of the constants for the different parameters considered in this study.
It can therefore be concluded that the optimization of HSSFCW design should take into account the contribution of mass transfer effects on the rate constants. Designs should aim at increasing mass transfer in the system, therefore use of baffles in constructed wetlands should seriously be considered.

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