Gas Treatment in Trickle-Bed Biofilters: Biomass, How Much Is Enough?

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Abstract: The objective of this article is to define and validate a mathematical model that describes the physical and biological processes occurring in a trickle-bed air biofilter for waste gas treatment. This model considers a two-phase system, quasi-steady-state processes, uniform bacterial population, and one limiting substrate. The variation of the specific surface area with bacterial growth is included in the model, and its effect on the biofilter performance is analyzed. This analysis leads to the conclusion that excessive accumulation of biomass in the reactor has a negative effect on contaminant removal efficiency. To solve this problem, excess biomass is removed via full media fluidization and backwashing of the biofilter. The backwashing technique is also incorporated in the model as a process variable. Experimental data from the biodegradation of toluene in a pilot system with four packed-bed reactors are used to validate the model. Once the model is calibrated with the estimation of the unknown parameters of the system, it is used to simulate the biofilter performance for different operating conditions. Model predictions are found to be in agreement with experimental data. © 1997 John Wiley & Sons, Inc. Biotechnol Bioeng 54: 583–594, 1997.

Keywords: trickle-bed biofilter; mathematical model; volatile organic compound (VOC); waste gas treatment; biofiltration

INTRODUCTION

Biofiltration, as a control technology for volatile organic compound (VOC) laden exhaust gases, continues to receive attention in the environmental field. A biofilter consists of a packed bed of organic or synthetic material on which microbial films are supported. Biodegradable pollutants present in waste gas pass through the biofilter, diffuse through the attached microbial film, and are consumed. Since pollutant degradation occurs at normal temperatures and pressures, biofiltration represents a potentially energy efficient technology when compared to traditional physical and chemical control processes (e.g., incineration and carbon adsorption). However, biofilters are living pollution control systems and are subject to dynamic changes. This characteristic has hindered the widespread application of biofiltration in the United States, where regulatory requirements typically stipulate continuous compliance with emissions limitations or destruction efficiency. To help develop biofiltration as a viable technology capable of meeting regulatory constraints, researchers have focused on understanding the fundamental physical, chemical, and microbiological processes that control reactor performance.

Recently, biofilter models have been introduced that account for detailed representations of biofilm biodegradation mechanisms. Shareefdeen et al. (1993) proposed a model for a single-component waste stream that accounted for oxygen limitations in the biofilm. Smith (Smith, P. J. 1993. A fundamental approach to modeling the treatment of VOC laden exhaust gases in biofilters, M.S. Thesis, University of Cincinnati) developed a two-phase trickle-bed biofilter model that incorporated decay and microbial shearing in determining the spatial distribution of biomass. Deshusses et al. (1995a, b) developed a dynamic model that calculates the variation of the contaminant concentration over time assuming constant biofilm thickness and includes multiple-substrate degradation.

In this article, the conventional theoretical model of a synthetic media trickle-bed biofilter is enhanced with new features. The model describes the degradation of one limiting substrate (VOC pollutant) in a homogeneous biomass by one type of microbial species. The dynamics of the system are characterized by a quasi-steady-state term that accounts for biofilm growth. As a response to experimental observations, a new approach in the calculation of the specific surface area as a function of biomass growth is presented. The experimental data from the pilot system show an initial increase and a subsequent drop in contaminant removal efficiency with time while biomass is accumulating in
the system. The initial improvement in performance is easily explained since an increase in the mass of microorganisms results in increased contaminant removal capacity. To explain the subsequent drop in removal efficiency, the key variable is the reduction in the specific surface area of the biofilm that accompanies biomass accumulation. In this model, the biomass is assumed to grow in the void fraction between the packing solids since the solids do not move to accommodate the new microorganisms. Therefore, available area for the contaminant to diffuse into the biofilm and removal efficiency decreases. To remove the excess biomass that causes the contaminant removal efficiency to drop, the reactor is backwashed regularly with full media fluidization. The backwashing technique parameters, duration and frequency, also have a significant influence on the effectiveness of the reactor and, therefore, are considered as variables in the model.

MATERIALS AND METHODS

The biofilter system used in this study consists of four stainless steel reactors, each packed with pelletized diatomaceous earth biological support media (6 mm R-635 Celite) to a depth of 112 cm and internal diameter of 14.6 cm. The pelletized medium was selected after initial screening revealed it to be superior to two other candidate media (Sorial et al., 1995b). The average size of the packing solids was 6 mm in diameter and 4 mm in length. They were represented in the model by equivalent spheres of 6 mm diameter sized to have the same volume as the packing solids. The sphericity factor, calculated as the ratio of the surface of the equivalent sphere to the surface of the pellet, was 0.857. The porosity of the packed bed was 0.34. The organic feed to the biofilter consisted of toluene volatilized in the influent air stream. Prior to the addition of toluene, the feed air was purified and contained only oxygen and nitrogen. The biofilters were fed 20 L of an aqueous solution of nutrients per day. Nitrate was the sole nitrogen source. The temperature was maintained at 32°C through the biofilter length, and the outlet pressure was very close to atmospheric pressure. The biofilters were operated in a co-current gas/liquid downward-flow mode. A more detailed description of the experimental apparatus can be found in previous publications (Smith et al., 1994; Sorial et al., 1995a, b).

The empty-bed residence time (EBRT) and the influent concentration for each biofilter are summarized in Table I. Two different loadings [4.1 and 6.2 kg chemical oxygen demand (COD)/m³ day], two different initial toluene concentrations (250 and 500 ppmv), and four different values of the EBRT (2, 1, 1.33, and 0.67 min) were used. The experimental results obtained with this system are described by Smith et al. (1995). The value of the kinetic parameters (Monod constant, yield coefficient, maximum specific growth rate, and decay coeffi-

<table>
<thead>
<tr>
<th>Biofilter</th>
<th>Toluene initial conc. (ppmv)</th>
<th>Empty-bed residence time (min)</th>
<th>Loading (kg COD/m³-day)</th>
</tr>
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<tbody>
<tr>
<td>A</td>
<td>500</td>
<td>1.33</td>
<td>6.2</td>
</tr>
<tr>
<td>B</td>
<td>250</td>
<td>0.67</td>
<td>6.2</td>
</tr>
<tr>
<td>C</td>
<td>250</td>
<td>1</td>
<td>4.1</td>
</tr>
<tr>
<td>D</td>
<td>500</td>
<td>2</td>
<td>4.1</td>
</tr>
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</table>

cient), the biofilm biomass density, the biofilm/water diffusivity ratio, and the initial biofilm thickness were not known a priori and could not be measured. Hence, they were estimated together with the validation of the model.

Performance Characteristics of Trickle-Bed Biofilters

Extensive description of the operating characteristics of the biofilters can be found in Sorial et al. (1995b). The initial phase of this research focused on the evaluation of three different types of biological attachment media for biofiltration: a peat mixture, a synthetic channelized medium, and a randomly packed pelletized medium. After the pelletized medium was shown to be the most effective packing material, its performance characteristics were further evaluated. These investigations suggested that an increase in biofilter operating temperature permitted a higher practical VOC loading and that toluene removal efficiency decreases with a buildup in head loss due to excessive accumulation of biomass in the reactor. This experiment was conducted with toluene loadings of 0.46, 0.92, 1.84, and 2.27 kg COD/m³ day, influent toluene concentrations of 50 ppmv, 100 ppmv, and 250 ppmv, and the two different EBRTs of 1 and 2 min.

The resulting contaminant removal efficiency and pressure drop in the biofilter are shown in Figure 1. The biofilter loading was initially 0.46 kg COD/m³ day (50 ppmv influent toluene concentration at 2 min EBRT), and the removal efficiency of the biofilter stabilized at more than 99.96%. On day 45, the EBRT was reduced to 1 min, doubling the toluene loading to 0.92 kg COD/m³ day. The resulting removal efficiency dropped to and stabilized at 92%. Simultaneously with the decrease in the EBRT, the pressure drop began to increase.

On day 82, the temperature was increased from 11°C to 15.6°C and the removal efficiency increased to over 99%. The pressure drop across the biofilter did not decrease, however. On day 97, the inlet concentration was increased to 100 ppmv, resulting in a loading rate of 1.84 kg COD/m³ day. The removal efficiency dropped to 86%, and the pressure drop remained stable at about 15 cm of water. The temperature was further increased to try to improve performance. Nevertheless, after a brief period of increased removal efficiency, it dropped...
to about 78%. During this period, the pressure drop increased rapidly. From this study, it was concluded that high pressure drop due to excessive accumulation of biomass in the system resulted in decreased removal of toluene, and, therefore, a method for the control of biomass accumulation within the system is essential in sustaining high levels of biofilter performance. The selected biomass control strategy employed backwashing of the packed bed with full media fluidization, i.e., in situ upflow washing with water. The second half of Figure 1 summarizes the results corresponding to this period. The biofilter was operated with an influent toluene concentration of 250 ppmv, an EBRT of 2 min, and a COD volumetric loading rate of 2.27 kg/m³ day. After several backwashing strategies that did not involve full media fluidization were evaluated, a procedure involving full media fluidization was selected on day 73. The water flow required to achieve 40% bed expansion was 3.2 m³/h, involving a total water volume of 200 L. Of this volume, 100 L was recirculated through the biofilter for 1 h, followed by rapid flushing of the fluidization water with 100 L of clean water.

This method of backwash with full media fluidization succeeded in controlling biomass accumulation in the reactor with no measurable pressure drop between backwashings and resulted in a consistent toluene removal efficiency of 99%. The stable performance of the biofilter with backwashing using full media fluidization is illustrated in Figure 1. On day 132, the recycle and flush water volumes were reduced to 70 L and 50 L, respectively, without any apparent effect on system performance. Regular backwashing with full media fluidization was practiced in all the remaining experiments.

After refinement of the biomass control strategy, additional experiments were conducted to evaluate the effects of the COD volumetric loading rate and the duration and frequency of backwashing on the performance of the trickle-bed biofilter (Smith et al., 1995; Sorial et al., 1995a). The four biofilters described in the previous section were operated using three different backwashing strategies that involved varying the frequency and the duration of the full media fluidization. Table II summarizes these values for the three backwashing strategies. Each strategy is identified with a number that will be used in the rest of the discussion.

Characteristic performance of biofilter A during the period between backwashings is presented in Figure 2. Biofilter A was operated with a COD volumetric loading rate of 6.2 kg/m³ day, an influent toluene concentration of 500 ppmv, an EBRT of 1.33 min, and backwashing strategy BW#1. Pollutant removal efficiency in the biofilter is presented as a function of time following backwashing. The initial time corresponds to the moment when the reactor was started after backwashing. The figure shows an initial increase and a subsequent drop in contaminant removal efficiency while biomass is accumulating in the system. Removal efficiency would typi-

<table>
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<th>Table II. Backwashing strategies.</th>
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<td>Backwashing strategy No.</td>
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<tr>
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<tr>
<td>BW#1</td>
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<td>BW#2</td>
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<td>BW#3</td>
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and wall and end effects are neglected. The model considers two phases: a gaseous phase and the biofilm. A liquid layer, present due to a small and intermittent flow of a solution of nutrients, was assumed to have minimal mass transfer resistance and is disregarded. The temperature in the biofilter and the physical properties of the gas and VOCs are assumed to be constant. The variables of interest are the VOC concentration profiles in the biofilm and gas phase and the thickness of the biofilm along the reactor, i.e., the biomass profile.

The model is solved assuming quasi-steady-state conditions because the characteristic time of VOC transport and utilization is much smaller than the one for bacterial growth. This means that the concentration of contaminant in the biofilm reaches steady state before the growth of the biofilm becomes significant. For each time step, the concentration profile along the biofilter is calculated assuming constant biofilm thickness. The calculated concentration profile is then used to compute the change in biofilm thickness during this time step. The value of the time step is chosen so that the convergence of the algorithm is assured. The solution obtained with this scheme is found to be identical to the exact solution of the system of partial differential equations describing the time-varying performance of the treatment system.

In the formulation of mass balance equations around the biofilter, the following assumptions are made: The biofilm is a stagnant phase with molecular diffusion as the only transport mechanism; axial diffusion is negligible; microbial growth is described by Monod kinetics; VOCs present are the only growth-limiting substances; and all kinetic parameters and the biofilm bacterial density are constant along the biofilter. The biofilter geometry is presented in Figure 3. The mass balance equation for the biofilm phase, expressed in spherical coordinates to account for the curvature of the medium particles, and the corresponding boundary conditions are

![Figure 3. Biofilm model representation.](image-url)
\[ D_f \left[ \frac{d^2 C_f}{dr^2} + \frac{2}{r} \frac{dC_f}{dr} \right] = \frac{\mu_m X_f}{Y} \frac{C_f}{K_s + C_f} \]

at \( r = R \)
\[ \frac{dC_f}{dr} = 0 \] (1)

at \( r = R + L_f \)
\[ C_g = HC_f \]

where \( C_f (M/L^3) \) is the VOC concentration in the biofilm; \( D_f (L^2/t) \) is the contaminant diffusivity in the biofilm and is assumed to be a fraction, \( r_d \), of the contaminant diffusivity in water, \( D_w \); \( \mu_m (r^-1) \) is the maximum bacterial growth rate; \( Y (M \text{ biomass}/M \text{ VOC}) \) is the yield coefficient; \( K_s (M/L^3) \) is the Monod saturation constant; and \( X_f (M/L^3) \) is the film bacterial density. The boundary conditions have been formulated assuming that there is no flux of contaminant into the surface of the packing solids and that the concentrations of VOC in the biofilm and gas phase at the interface are in equilibrium as defined by Henry’s law. The use of Henry’s law is justified due to the high water content of the biofilm.

Because the gas velocity and the VOC concentration are uniform radially across the bed, plug flow can be assumed for the gas phase. Assuming that there is no contaminant degradation in the gas phase and the concentration of contaminant in the gas phase at the inlet of the biofilter is uniform, the mass balance equation at the biofilm–gas interfacial and the initial condition can be expressed as

\[ \frac{dC_g}{dz} = -Ja_f \]

at \( z = 0 \)
\[ C_g = C_{g0} \] (2)

where \( C_g (\text{ppmv}) \) is the VOC concentration in the gas phase, \( J (M/L^2 t) \) is the flux of VOC into the biofilm, \( a_f (L^{-1}) \) is the biofilm–gas interfacial surface area per unit bed volume, \( u_0 (L/t) \) is the approach velocity of the gas, i.e., the gas flow rate divided by the total cross-sectional area of the biofilter, and \( C_{g0} \) is the initial VOC concentration in the gas phase. The flux of VOC out of the gas phase is equal to its flux into the biofilm surface:

\[ J = \frac{PM_v}{R_g T} = J_f = r_d D_w \frac{dC_f}{dr} \] (3)

where \( R \) (L) is the radius of the sphere equivalent of the packing medium, \( L_f \) (L) is the biofilm thickness, \( P \) (atm) is the system pressure, \( M_v \) (M/mol) is the molecular weight of the VOCs, \( R_g \) (atm L^3/mol K) is the universal gas constant, and \( T \) (K) is the system temperature.

Since the biofilm thickness along the biofilter is not constant, another equation is needed to characterize its variation. The variation in the thickness of the biofilm with time is due to bacterial growth and decay. If \( b (r^-1) \) is the specific combined shear/decay coefficient, this equation is

\[ \frac{dL_f}{dt} X_f = \left( r_d D_w \frac{dC_f}{dr} \right) Y - L_f X_f b \]

at \( t = 0 \)
\[ L_f(z, t) = L_{f0}(z) \] (4)

where the initial time is considered to coincide with the restart of the biofilter after backwashing and \( L_{f0}(z) \) is the initial biofilm thickness than can vary along the reactor depth.

A variable of interest is the concentration of biomass in the reactor, \( X_f (M/L^3) \), since this concentration can be measured and it is an indication of the biofilm thickness. Here, \( X_f \), which represents the mass of microorganisms per volume of reactor, varies along the reactor depth as contrasted with the density of biomass within the biofilm, which is considered constant. Assuming the bed volume remains constant with biomass growth, the biofilm concentration in the reactor can be calculated as

\[ X_f = X_f(\epsilon_0 - \epsilon_f) \] (5)

where \( \epsilon_0 \) and \( \epsilon_f \) are the clean packed-bed porosity and the porosity of the bed with biofilm, respectively.

**Water/Biofilm Diffusivity Ratio**

The contaminant molecular diffusivity in the biofilm has been assumed to be a fraction of its diffusivity in water. This fraction is called the water/biofilm diffusivity ratio and is defined as

\[ r_d = \frac{D_f}{D_w} \] (6)

**Shear/Decay Coefficient Calculation**

Following the formulation of Rittmann (1982), the combined shear/decay coefficient represents the effects of biomass loss and combines biomass decay and physical shearing. The specific decay coefficient, \( b_d \), is assumed to be constant, and the specific shear rate, \( b_s \), is assumed to be a function of the biofilm thickness:

\[ b = b_s + b_d \] (7)

The different existing expressions to define the rate of biofilm detachment suggest that this process is not very well understood (Peyton and Characklis, 1993). Here, the shear rate is assumed to be proportional to the shear stress, \( \tau (L/t) \), which is proportional to the interpore gas velocity, \( u (L/t) \), \( u = u_0/\epsilon_f \):

\[ \tau = \beta \left( \frac{u}{\epsilon_f} \right) = \beta \left( \frac{u_0}{\epsilon_f^0} \right) \] (8)

The proportionality constant, \( \beta \), is chosen to be the default shear rate coefficient, \( \beta_0 \), corresponding to the default shear stress, \( \tau_0 \), when the bed is clean and no biofilm is present. Then
\[ \tau^0 = \beta \left( \frac{u_0}{\varepsilon_0} \right) \rightarrow b^0 \alpha \beta \left( \frac{u_0}{\varepsilon_0} \right) \]  

Eliminating the constants, the final expression for \( b \) is

\[ b = b^0 \left( \frac{u_0}{\varepsilon_0} \right)^2 + b_d \]  

**Specific Surface Area and Porosity Calculation**

The biofilm–gas phase interfacial surface area per unit volume of clean biofilter, \( a_0 (L^{-1}) \), is given by

\[ a_0 = \frac{3(1 - \varepsilon_0)}{\phi R} \]  

where \( \phi \) is the packing solids sphericity and \( \varepsilon_0 \) is the porosity of the clean reactor bed without biofilm. To calculate the specific surface area with biofilm growth, \( a_f (L^{-1}) \), the experimental results shown in Figure 2 and discussed in the introduction have to be considered. Experimental observations showed that the biofilter removal capacity decreases usually after 1 day of continuous operation when biomass has begun to accumulate in the biofilter. Monod kinetics and mass transfer models alone cannot explain the behavior observed in Figure 2. Actually, these models predict an increase in contaminant removal if the quantity of biomass increases. Therefore, the solution to this problem depends on reactor characteristics, particularly on the changes in biofilm–gas phase surface area available for contaminant transport into the biofilm.

Traditionally, it has been assumed that the packing solids will move to fit the growing biomass, i.e., solid particles that were in contact when the biofilter was clean will now separate to fit the developing biofilm that will completely cover them. However, in practice, the solids that were in contact when the biofilter was clean will remain in contact and, therefore, the biomass will grow only in the void space left between the solids. This assumption implies that the bed volume remains constant during system operation.

A schematic representation of these two cases is shown in Figure 4. The equations for the specific surface area and bed porosity in the reactor with biofilm will be formulated in the context of the second case. The values of these two parameters will depend on the number of spheres initially in contact with a given sphere, \( n \). This situation is illustrated in Figure 5. The terms \( A_L \) and \( V_L \) are, respectively, the biofilm surface area and the biofilm volume lost with each point of contact. Then the specific surface area is

\[ a_f = \frac{4\pi(R + L_f)^2 - n A_L}{4 \frac{3}{\pi} R^3} (1 - \varepsilon_0) \]  

\[ = \frac{3(1 - \varepsilon_0)}{2R} \left( 1 + \frac{L_f}{R} \right) \left( 2 - n \frac{L_f}{R} + 2 \right) \]  

The bed porosity with biofilm, \( \varepsilon_f \), can be calculated in the same way:

\[ \varepsilon_f = 1 - \frac{4 \frac{3}{\pi} (R + L_f)^3 - n V_L}{4 \frac{3}{\pi} R^3} (1 - \varepsilon_0) \]  

\[ = 1 - (1 - \varepsilon_0) \left[ \left( 1 + \frac{L_f}{R} \right)^3 - n \frac{L_f}{4} \left( \frac{2}{R} L_f + 3 \right) \right] \]  

With this formulation of the problem, the actual specific surface area and bed porosity are functions of the number of packing solids in contact with a given one and the ratio of the biofilm thickness to the radius of the equivalent sphere. The variation of the specific surface area with the dimensionless biofilm thickness, \( L_f/R \), for different values of \( n \) is plotted in Figure 6. It can be observed that for values of \( n < 4 \), the specific surface area increases with increasing biofilm thickness. For val-
Numerical Solution

The dimensionless form of Equation (2) is solved using an Adams–Moulton finite difference scheme. The solution is found by marching axially through the biofilter. At each axial step, the flux, biofilm concentration profile, biofilm thickness, and packed-bed characteristics are evaluated. The dimensionless form of Equation (1) is solved with a second-order two-point boundary value problem direct method. Two levels of iteration are required in each axial step to handle the nonlinear term in Equation (1) and to calculate the biofilm thickness given by the linearization of the dimensionless form of Equation (4).

The thickness of the biofilm has two physical limits when the bed is clogged, i.e., when the porosity gets too small or the specific surface area is close to zero, whichever occurs first. Close to clogging, the assumption of spherical packing solids with a shell of biofilm is not valid. These two conditions define the maximum biofilm thickness enforced in each iteration.

RESULTS AND DISCUSSION

The validation of the mathematical model was carried out in two stages: estimation of the model parameters and testing of the model predictions. Thus, two different sets of data were needed. The first set was collected from an experiment conducted on a different biofilter run, which will be referred to as model biofilter (biofilter M) and the second one from the four biofilters during normal operation. Biofilter M was operated with an influent toluene concentration of 250 ppmv and an EBRT of 1 min.

Although this reactor does not have regular packing, \( n \) can be estimated using the previous equation and the value of the clean-bed porosity. For a porosity value of 0.34, the corresponding value of \( n \) is between 9 and 10. Because it yields better results in the validation of the model, \( n = 10 \) is used.

The packed-bed biofilter model is defined by Equations (1)–(7) and (10)–(13). These equations are solved in dimensionless form for mathematical simplicity and to reduce the number of model parameters. The new dimensionless variables are

\[
\begin{align*}
\epsilon &= 1.072 - 0.1193n + 0.004312n^2 \\
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\]  
(14)

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\[
\begin{align*}
z^* &= \frac{z}{L} \\
r^* &= \frac{r}{R} \\
t^* &= t(b_0^* + b_d) \\
C_g^* &= \frac{C_g}{C_{g0}} \\
C_f^* &= \frac{H C_f}{C_{g0}} \\
L_f^* &= \frac{L_f}{R} \\
L_f^{*0} &= \frac{L_f^{*0}}{R} \\
a_f^* &= \frac{a_f}{a_0} \\
b^* &= \frac{b}{b_0^* + b_d} \\
J^* &= J\left(\frac{R P M \epsilon H}{r_d D_w R_g T C_{g0}}\right) \\
\end{align*}
\]  
(15)

The value of \( n \) has been obtained from the value of the clean-bed porosity (Dullien, 1979). For regular packing, the number of points of contact between a given sphere and the adjacent spheres is called the coordination number. The most common regular packing systems are cubic, \( n = 6 \), and rhombohedral or compact, \( n = 12 \), with porosities 0.476 and 0.25, respectively. For regular packing, a relation exists between the value of the clean-bed porosity and the coordination number which can be expressed as follows (Dullien, 1979):

\[
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\[
\epsilon = 1.072 - 0.1193n + 0.004312n^2 \\
\]  
(14)
Eight unknown parameters were estimated: the yield coefficient, $Y$; biomass density, $X_f$; the decay rate coefficient, $b_d$; the Monod constant, $K_s$; the maximum growth rate, $m$; the default shear rate, $\mu_m$; the biomass density in the biofilm, $X_f$; and the initial biofilm thickness, $L_{0i}$. The parameter estimation technique used was the weighted least-squares errors method. The kinetic parameters that resulted in the best fit of the mathematical model solution to the experimental data and those parameters reported by Arcangeli and Arvin (1992) are given in Table III. The values of the toluene diffusivity in water and the toluene Henry constant used in the model are also included in Table III.

To determine the most critical parameters in the estimation process, a sensitivity analysis of the model was performed. Simulations were run for each estimated parameter modifying the nominal value by 10% and keeping the other parameters constant. The impact of the variation in the parameter value in the model predictions was measured as the difference between the nominal model predictions for the toluene removal efficiency and biofilm thickness along the biofilter (Fig. 7) and the results obtained when each parameter was modified. The measurement used as the difference between two functions was the numerically calculated area between them.

The results of this study are presented in the third and fourth columns of Table III, where the calculated area is termed as parameter impact. For each parameter the value of this area is given for the two predicted variables, the toluene removal efficiency and the biomass distribution with depth. The most influential parameters are the ones with the highest impact values. For the predicted toluene removal efficiency these parameters are the yield coefficient, $Y$, the biomass density, $X_f$, the ratio between VOC diffusivities in biofilm and water, $r_d$, and the maximum growth rate, $m$. The yield coefficient, $Y$, the biomass density, $X_f$, and the decay rate, $b_d$, are the ones that have more impact in the predicted biomass. This analysis indicates which parameters should be more accurately evaluated.

The model was then tested in the prediction of the dynamic performance of the four biofilters previously described. During the periods between backwashings, toluene removal efficiency in each reactor was regularly determined over 242 days of operation. Data were collected during the first 5 h and at the end of the first, second, and third day (if applicable) after the restart of the reactor for the three different backwash strategies. The averages of observed values at different times and the model predictions are shown in Figure 8. For backwashing strategy BW#3, the efficiency was measured only the first and the second day after the restart because the frequency of backwashing was every 48 h. The error bars in Figure 8 represent the percentage coefficient of variation of the sample, i.e., the standard deviation of the sample divided by the mean times 100.

Although the efficiency of the reactor was measured each day, the analysis of the first 5 h was done only on some selected days, so the number of observations is smaller. For backwashing strategy BW#2 in biofilter D, only one set of observations was taken, and, therefore, there are no error bars for this plot. The predictions of the model are in most cases within the error bars. As the biomass removal process was not included in the mathematical description of the system, the biomass density in the biofilm and the initial biofilm thickness were assumed to be variables dependent on the back-
Table III. Biofilter parameters.

<table>
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<tr>
<th>Parameter</th>
<th>Estimated value</th>
<th>Arcangeli and Arvin value</th>
<th>Parameter impact on efficiency</th>
<th>Parameter impact on biomass</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum growth rate, $\mu_m$</td>
<td>3.0 day$^{-1}$</td>
<td>3.5 day$^{-1}$</td>
<td>0.115</td>
<td>0.788</td>
</tr>
<tr>
<td>Decay rate coefficient, $b_d$</td>
<td>0.432 day$^{-1}$</td>
<td>0.6 day$^{-1}$</td>
<td>0.014</td>
<td>2.228</td>
</tr>
<tr>
<td>Default shear rate coefficient, $b_\sigma$</td>
<td>0.005 day$^{-1}$</td>
<td></td>
<td>0.001</td>
<td>0.106</td>
</tr>
<tr>
<td>Monod kinetic constant, $K_s$</td>
<td>0.15 mg VOC/L</td>
<td>0.47 mg COD/L</td>
<td>0.028</td>
<td>2.676</td>
</tr>
<tr>
<td>Yield coefficient, $Y$</td>
<td>0.4 mg COD/mg COD</td>
<td>0.5 mg COD/mg COD</td>
<td>0.019</td>
<td>1.547</td>
</tr>
<tr>
<td>Ratio between VOC diffusivities in</td>
<td>9.0</td>
<td>9.0</td>
<td>0.119</td>
<td>2.075</td>
</tr>
<tr>
<td>biofilm and water, $r_d$</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biomass density, $X_f$</td>
<td>17000 mg biomass/L</td>
<td>24140 mg COD/L</td>
<td>0.157</td>
<td>1.421</td>
</tr>
<tr>
<td>Initial biofilm thickness, $L_f$</td>
<td>0.0042 cm</td>
<td></td>
<td>0.014</td>
<td>2.516</td>
</tr>
<tr>
<td>Number of contacting spheres, $n$</td>
<td>10</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Toluene diffusivity in water, $D_w$</td>
<td>$10.8 \times 10^{-6}$ cm$^2$/s</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Toluene Henry constant, $H$</td>
<td>104.03 ppmv/(mg/L)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 8. Variation of the toluene removal efficiency after backwashing in four biofilters operating in different conditions and using three different backwashing strategies.
The estimated values of the biomass density and the initial biofilm thickness are presented in Table IV. Although these two parameters were selected to fit the data, correlations between them and the operational variables of the biofilter can be inferred from Table IV. There is a relationship between the influent contaminant concentration and the biomass density and initial biofilm thickness: Biofilters A and D with 500 ppmv influent concentration have lower values of biomass density and higher values of initial thickness than biofilters B and C with 250 ppmv influent concentration. When the biofilters with the same influent toluene concentration are compared, the biomass density and the initial biofilm thickness are higher in the biofilters loaded with 6.2 kg COD/m³-day than in the ones loaded with 4.1 kg COD/m³-day. Biomass density also appears to be related to the backwashing technique: it was found to be lower for backwashing strategy BW#1 than for BW#3. BW#1 is less aggressive than BW#3: 1 h every 84 h for BW#1 compared to 1 h every 48 h for BW#3. Figure 8 illustrates that the variation of the contaminant removal efficiency of the biofilter can be reasonably predicted over time. In all cases, the removal efficiency increases and then drops after the first day of operation. It also reveals that the differences in the backwashing strategy employed have a great influence on biofilter performance.

The value of the mean-square error (MSE) can be used to measure the accuracy of the model predictions. MSE values are presented in Table V for the 12 different experimental treatments, a treatment here being a biofilter with a backwashing strategy. The MSE is calculated as the sum of the squared residuals divided by the number of observations. MSE1 corresponds to the data collected in the first 5 h, and MSE2 to the data collected in the first, second, and third days. These values are included for comparison. The MSE for each biofilter (MSE biofilter in Table V) and for each backwashing strategy (MSE backwash in Table V) have also been calculated. The value of MSE backwash has been obtained grouping the data corresponding to each backwashing strategy.

The biofilters are ranked according to the value of the MSE: biofilter D (MSE = 2.63, EBRT = 2 min) being the best fit, followed by biofilter A (MSE = 6.9, EBRT = 1.33 min), biofilter C (MSE = 9.87, EBRT = 1 min), and biofilter B (MSE = 17.39, EBRT = 0.67 min). When the backwashing strategies are compared, the best fit corresponds to BW#3 (MSE = 7.08), followed by BW#1 (MSE = 10.67) and BW#2 (MSE = 13.87). It is interesting to note that lower EBRT corresponds with lower goodness of fit. Decreasing EBRT may raise the importance of other factors neglected in the model, such as oxygen limitations, mass transfer resistance of the liquid layer, and concentration gradient in the bulk gas. In general, removal efficiencies predicted by the model are higher than the measured ones, due probably to the aforementioned factors not incorporated in the model.

Overall, the model is more accurate in predicting the biofilter performance after the first 24 h rather than during the initial stages; in 8 cases of 12, MSE2 is lower than MSE1. This is because the system is disrupted after backwashing and takes time to stabilize. In fact, the points where the model is the furthest away from the data are after the first 20 and 60 min of operation following backwashing. Another factor to consider is that the backwashing process has not been included in the math-
ematical formulation of the model but has been incorpo-
rated in the estimation of the initial biofilm thickness
and biomass density.

CONCLUSIONS

A mathematical model that describes the biotreatment
of a waste gas in a trickle-bed reactor has been proposed
and validated. The model considers a two-phase system,
uniform bacterial population, one limiting substrate,
and quasi-steady-state processes. In the model analysis,
the unknown biofilter parameters have been estimated
and the model has been tested against experimental
data. A new approach describing the variation in the
biofilter specific surface area with microbial growth has
been included. Variations in specific surface area can
explain experimental results that exhibit a decrease in
contaminant removal efficiency in the biofilter while
biomass is growing. This fact leads to the question in
the title of the article—How much biomass is enough?—
which has to be answered considering not only the
amount of biomass in the system but also how accessible
is its biomass. The most important conclusion of this
article is that the performance of the biofilter depends
not only on the amount of biomass but also on the
amount of biomass that can be readily accessed by the
diffusing contaminant.

Four biofilters with different operating conditions
were used in the validation process. To improve biofilter
performance, biomass accumulated in the biofilter was
periodically removed using three different backwashing
techniques. Two sets of observations have been em-
ployed in the nonlinear estimation of the parameters,
the biofilter treatment efficiency profile and the biomass
concentration. These variables have been calculated and
compared with experimental data from a model bio-
filter, biofilter M. The biofilter was continuously oper-
dated during 9 days, and air scouring was used during
the backwash at the start and at the end of the period.
The estimated parameters are the kinetic constants: Mon-
don constant, maximum growth rate, yield coefficient,
decay and shear rate coefficients, the ratio between the
VOC diffusivity in the biofilm and water, the biomass
density, and the initial biofilm thickness. The biomass
density and the initial biofilm thickness are assumed to
be dependent on the biofilter operating conditions. The
model has been tested in the prediction of the variation
to toluene removal efficiency of the four biofilters with
time. The initial time is considered to be the restart of
the biofilter after backwashing.

Prediction of biofilter performance over time is suc-
cessful when the variation of the specific surface area
with biofilm growth is included in the model. To evalu-
ate the accuracy of the model, the MSE values were
used as a measure of goodness of fit. It was found that
the accuracy of the predictions depended upon the ini-
tial contaminant concentration, the EBRT, and the
backwashing technique. Time after backwashing was
also a factor. The model agreement was better (lower
values of MSE) after 1 day when the biofilter was stable.
Simultaneously, some problems have been identified.
The method used to remove excess biomass in the sys-
tem had a big influence on the performance of the bio-
filter but currently is not included in the mathematical
description. However, it is considered in the determina-
tion of the biomass density and the initial biomass thick-
ness. Future models should mathematically account for
biofilter backwashing. They should also include more
than one limiting substrate, such as oxygen and nitrate,
and nonhomogeneous biomass species, such as active
and inactive biomass. The model presented in this article
provides a greater understanding of the VOC degrada-
tion process in a biofilter and identifies fundamental
questions to be addressed in the development of fu-
ture models.

NOMENCLATURE

\( a_{0} \) clean bed surface area per unit volume (\( \text{cm}^{-1} \))
\( a_{1} \) surface area per unit volume when there is biofilm growth
in the bed (\( \text{cm}^{-1} \))
\( A_{1} \) biofilm area lost in each contact point between equivalent
spheres (\( \text{cm}^{2} \))
\( b \) shear/decay rate coefficient (\( \text{s}^{-1} \))
\( b_{d} \) default shear/decay rate coefficient (\( \text{s}^{-1} \))
\( b_{f} \) decay rate coefficient (\( \text{s}^{-1} \))
\( b_{g} \) shear rate coefficient (\( \text{s}^{-1} \))
\( b_{o} \) default shear rate coefficient (\( \text{s}^{-1} \))
\( C_{f} \) biofilm VOC concentration (\( \text{mg/L} \))
\( C_{g} \) gas phase VOC concentration (ppmv)
\( C_{d0} \) inlet gas phase VOC concentration (ppmv)
\( D_{1} \) VOC diffusivity in the biofilm (\( \text{cm}^{2}/\text{s} \))
\( D_{w} \) VOC diffusivity in water (\( \text{cm}^{2}/\text{s} \))
\( H \) Henry’s law constant [ppmv/(mg/L)]
\( J \) VOC flux in gas phase (ppmv \( \text{cm} \))
\( J_{f} \) VOC flux in biofilm (mg \( \text{cm}/\text{L} s \))
\( K_{s} \) Monod saturation constant (\( \text{mg/L} \))
\( L \) biofilter packing media length (cm)
\( L_{1} \) biofilm thickness (cm)
\( M/\text{L}^{3} \) mass/length\(^{3} \)
\( M_{a} \) VOC molecular weight (g/mol)
\( n \) number of characteristic packing spheres in contact with a
given sphere
\( p \) pressure in the biofilter (atm)
\( r \) radial coordinate (cm)
\( r_{d} \) ratio between VOC diffusivities in biofilm and water
\( R \) characteristic packing sphere radius (cm)
\( R_{g} \) universal gas constant (\( \text{cm}^{3} \text{ atm/mol K} \))
\( t \) time (s)
\( T \) system temperature (K)
\( u \) interpore velocity (\( \text{cm}/\text{s} \))
\( u_{0} \) approach velocity to the biofilter (\( \text{cm}/\text{s} \))
\( V_{1} \) volume of biofilm lost in each contact point between equiva-
lent spheres (\( \text{cm}^{3} \))
\( X_{f} \) biomass density (\( \text{mg/L} \))
\( X_{b} \) biomass concentration in the biofilter (mg/L)
\( Y \) yield coefficient (mg biomass/mg VOC)
\( z \) axial coordinate (cm)

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Greek letters

\[ \beta \] stress proportionally constant
\[ \epsilon \] porosity of a packed bed
\[ \epsilon_0 \] clean bed porosity
\[ \epsilon_f \] porosity in bed with biofilm
\[ \phi \] sphericity of packing solids
\[ \mu_m \] maximum growth rate (s\(^{-1}\))
\[ \tau \] shear stress (dyn/cm\(^2\))

References


