A fundamental model for trickling filter process design

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Wastewater treatment using biofilms grown on a support media was the first continuous flow bioreactor employed by sanitary engineers nearly a century ago. This wastewater treatment method became known as a trickling filter because wastewater dosed intermittently over support media such as slag or rocks "trickled" down through the media where organics were removed or "filtered" by microorganisms composing the biofilm. Process performance was measured by performing a biochemical oxygen demand (BOD) test on reactor influents and total process effluents. Rock filter heights were typically 1 to 2 m (3 to 6 ft).

The model predictions are based on substrate transport and first order kinetics.

Within recent years, plastic support media for biofilm growth have primarily been selected for use in newly constructed trickling filters. The plastic media are characterized by high voids (>90%) and low weight, which permit tower heights of 1 to 12 m (3 to 40 ft). Tall plastic media towers can produce efficient soluble BOD (sBOD) removal and are less subject to ponding than rock media trickling filters.

For many years design engineers have had limited success in applying fairly simple design equations to model plastic media trickling filters' performance. These existing trickling filter design approaches are reviewed below. Because it was found that different plastic media affect wastewater treatment efficiency, a computer model was developed to predict sBOD removal for various plastic trickling filter media geometries. In this paper, a trickling filter model is formulated and validated. Engineering implications of the trickling filter model are analyzed in a companion paper.5

REVIEW OF TRICKLING FILTER MODELS

Two approaches have been used to design trickling filters. The first is based on the performance of similar units, which is summarized by a correlation; an example is the NRC equation.6

\[ E = \frac{1}{1 + 0.014 \left( \frac{w}{V_m R_f} \right)^{0.3}} \]  

(1)

Where:

- \( E \) = the fraction of BOD removed,
- \( w \) = a loading [kg-BOD/day],
- \( V_m \) = the total volume of filter media [10^3 m^3], and
- \( R_f \) = a recycle factor defined by

\[ R_f = \frac{1 + r_s}{(1 + (1-p)R_s)^2} \]

(2)

Where:

- \( p \) = a weighting factor typically equal to 0.9, and
- \( R_s \) = the recycle ratio defined as the ratio of recirculation flow to the filter influent flow.

Equation predictions were unsuccessful for widespread use and are not intended for plastic media filters.

The second approach to trickling filter design was based on an attempt to define kinetic equations governing BOD removal. The first recognized attempt was made by Velz6 who suggested BOD removal is proportional to BOD via:

\[ \frac{dC}{dt} = -k_a C \]

(3)

Where:

- \( C \) = the concentration of biologically degradable substrate as measured by the BOD test,
- \( z \) = the filter depth, and
- \( k_a \) = a rate constant [m^-1].

Upon integration, Equation 3 yields:

\[ \frac{C_{out}}{C_{in}} = e^{-k_a z} \]

(4)

Where:

- \( H \) = the total height of the trickling filter,
- the subscripts \( in \) and \( out \) refer to substrate concentrations entering and leaving the filter, respectively.

Howland7 advanced trickling filter design by noting that the average hydraulic residence time (\( \tau \)) of fluid over an inclined plate was

\[ \tau = \left( \frac{3\nu}{g \sin \theta} \right)^{0.33} \frac{L}{q_w^{0.67}} \]

(5)

Where:

- \( \nu \) = kinematic viscosity [cm^2/s],
- \( g \) = acceleration of gravity [cm/sw^2],
- \( \theta \) = angle of plate inclination [cm/sw^2],
- \( L \) = length of flow [cm], and
- \( q_w \) = flow per unit width of plate [cm^3/cm^-2].

Howland's model specifies removal as a function of time as:

\[ \frac{dC}{dt} = -k_e C \]

(6)

December 1987
Where:

Rate constant \( k_2 \) [s\(^{-1}\)].

Integrating Equation 6 and incorporating an empirical temperature correction factor, \( \Theta \), typically equal to 1.035, Howland proposed:

\[
\frac{C_{out}}{C_{in}} = e^{-(k_2 \Theta \tau - 2a_t)}
\]  

(7)

In Howland’s approach, temperature affects the viscosity in Equation 5 and the kinetic constant \( k_{2,0} \) [s\(^{-1}\)], which is adjusted to 20°C through \( k_t = k_{2,0} \Theta ^{-20} \).

Various modifications\(^1,7\) have been made to Equation 7, but the most generally accepted model is a form of Equation 7 referred to as the modified Velz equation\(^4\), or

\[
\frac{C_{out}}{C_{in}} = \left( \frac{(R_t + 1) \exp \left[ \frac{k_{20} \Theta (T-20)}{Q_i (R_t + 1)^n} \right]}{n} - R_t \right)^{-1}
\]  

(8)

Where:

\( Q_i \) = the flow before recycle divided by filter cross sectional area [L/m\(^2\) s or gpm/ft\(^2\)],

\( A_s \) = the media specific surface area [m\(^2\)/m\(^3\) or ft\(^2\)/ft\(^3\)], and

\( n \) = an empirical flow coefficient typically set to 0.5.

The units of \( k_{20} \) are L/m\(^3\) s\(^{0.5}\) (gpm/ft\(^2\))\(^{0.5}\); this constant bears little resemblance to the kinetic constant of Equation 6 and is either designated as a treatability coefficient, or simply as a \( k_{20} \) factor in the rest of this work.

Some critical observations can be made of the modified Velz equation. Although Howland\(^d\) fundamentally derived that residence time is proportional to the hydraulic load raised to the 0.67 power, the power typically used to fit data is 0.5. Furthermore, temperature effects on residence time (through fluid viscosity) are not incorporated in the modified Velz equation although they appear in Howland’s original derivation. Media inclination effect (plastic sheet angle) is not incorporated in Equation 8. These factors may add to the confusion in the literature concerning residence time determination in trickling filter media.\(^1,3,10-12\)  Because the values of \( \Theta \) and \( n \) are usually fixed, the modified Velz equation has one adjustable coefficient (\( k_{20} \)) and is essentially a correlation model.

The major failure of the modified Velz equation is that the \( k_{20} \) value is often different for different media with the same specific surface area.\(^1,3\) Parker and Merrill\(^d\) found in side by side pilot plant studies treating domestic wastewaters that \( k_{20} \)’s for two different media configurations (cross-flow and vertical-flow media) with the same specific surface area were 0.0019 and 0.0011 L/m\(^2\) s\(^{-1}\) (2.3 × 10\(^{-3}\) and 1.7 × 10\(^{-3}\)[gpm/ft\(^2\)]\(^{0.5}\)), respectively. According to calculations using the modified Velz model, such differences in \( k_{20} \) require nearly 50% more vertical flow media to effect the same fraction BOD removal than cross flow media.\(^4\) At current installed trickling filter media costs of approximately $350 per cu m ($10 per cu ft), this may reflect a difference of $2 million for a 36 m (120 ft) diameter trickling filter 6 m (20 ft) deep filled with cross-flow media versus a filter requiring 50% more vertical flow media.

DEVELOPMENT OF MASS TRANSFER EQUATIONS

Previous biofilm model application to trickling filter design has been hindered for a variety of reasons. Virtually all fundamental biofilm models require estimating substrate diffusivity. Although using defined substrates with well known diffusivities such as glucose is desirable for research studies, no previous study has suggested appropriate BOD component diffusivities for use in biofilm models trickling filter design. Many biofilm models also assume completely mixed fluid conditions\(^11\) even though completely mixed conditions in the fluid film may not be justifiable from the system hydraulics examination.\(^14\) In this section a fundamental model for substrate removal is developed specifically for use in trickling filters design. This model is based on: a biofilm model based on first-order, uptake kinetics; the assumption of a laminar flow in a thin fluid film; and soluble organic substrates with a five-component diffusivity spectrum.

Substrate removal models in biofilms. Substrate transfer models and substrate removal biofilm generally use molecular diffusion to describe mass transfer in the biofilm, at steady state, or:

\[
D_e \frac{d^2 C}{dx^2} = R
\]  

(9)

Where:

\( R \) = the rate of disappearance of substrate, and

\( D_e \) = the effective diffusivity.

The reduction in substrate diffusivity in biofilms because of increased path length and biofilm density has been discussed by many authors.\(^15-18\) Effective diffusivity reduction is estimated to be only 80% of molecular diffusivity even for the relatively dense biobeds artificially created by Williamson and McCarty\(^19,20\) by filtration of cells onto a support media.

There is no generally accepted rate model for substrate removal by microorganisms. Vaughan and Holder\(^11\) have assumed zero order kinetics for substrate removal. Swilley\(^21\) used first order kinetics; but most researchers have used Michaelis-Menten or Monod-type relationships.\(^19,20,23-25\) Harremoes\(^28\) contends that the biofilm kinetics are essentially zero order and the diffusional limitations result in the apparent kinetics being a bulk substrate concentration function. As substrate concentrations decrease the observed kinetics span three ranges of zero-order, half-order, and first-order. A theoretical discussion of the effects of the assumed order uptake kinetics on concentration profiles and mass flux was presented by Atkinson and Daoud\(^27\) and Atkinson and Davies.\(^28\)

In this model, first order removal kinetics are assumed, or \( R = k C \), with first order kinetic coefficient, \( k \). The substrate flux into the biofilm is based on a mass balance to cells comprising the biofilm matrix, assuming Brownian collisions between substrate and cells, as:

\[
Flux = C_s D_s (4 \pi a_c N_e E_b)^{0.5}
\]  

(10)

Where:

\( C_s \) = the substrate concentration at the biofilm surface,

\( D_s \) = the substrate diffusivity,

\( a_c \) = the radius of a spherical cell comprising a biofilm,

\( N_e \) = the biofilm cell concentration, and

\( E_b \) = a collector efficiency of 0 ≤ \( E_b \) ≤ 1 that accounts for unsuccessful collisions between substrate and cells caused by solution chemistry, hydrodynamics, and incomplete coverage of cells with enzymes used for substrate metabolism.
The $E_b$ value must be determined from model calibration. Assuming all cells in the biofilm are viable, the cell concentration value can be obtained for the biofilm as a function of porosity as:

$$N_c = \frac{(1 - \epsilon)}{4} + \frac{3}{\epsilon^2}$$  \hspace{1cm} (11)

Where:

$\epsilon = \text{biofilm porosity}$.  

Combining Equations 10 and 11, the substrate flux to the biofilm becomes:

$$\text{Flux} = C_i D \left( \frac{3 E_b (1 - \epsilon)}{a^2} \right)^{0.5}$$  \hspace{1cm} (12)

Molecular diffusivity is used because the fractional reduction of the molecular diffusivity in the biofilm, as well as reduced cell viability, can be incorporated in the parameter $E_b$. In the first-order model, substrate flux is proportionate to the product $C_i D$, whereas in a zero order model substrate flux would be proportionate to $(C_i D)^{0.5}$. Equation 12 shows that there is a maximum flux to the biofilm that is limited by mass transfer to the cells and attained for a collector efficiency equal to unity. Substrate flux to a biofilm in the trickling filter model is, therefore, a function of cell size, biofilm porosity, and collector efficiency $(E_b)$ of cells. Calculations used to obtain the substrate concentration at the biofilm surface $(C_i)$ are developed below.

**Mass transfer substrate in thin fluid films.** It is assumed that wastewater flow over a biofilm is hydraulically identical to a Newtonian liquid flow down an inclined surface. This assumption is common to most biofilm and trickling filter models. As shown in Figure 1, fluid flows over biofilm length $L$ in the $z$ direction with a parabolic velocity profile $v(x)$. Such flow is classified as laminar for Reynolds numbers (Re) less than 6 and turbulent for Re greater than 250, where the Re is based on the fluid thickness $(\delta_f)$, or $Re = \frac{v \delta_f}{\nu}$. Intermediate flow for $6 \leq Re \leq 250$ results in crest formation in the fluid and is classified as wavy flow. The parabolic velocity profile for laminar flow is given as:

$$v(x) = v_{max} \left(1 - \left(\frac{x}{\delta_f}\right)^2\right)$$  \hspace{1cm} (13)

![Figure 1 — Definition of a theoretical plate.](image)

With film thickness $\delta_f$ calculated from:

$$\delta_f = \left(\frac{3 q v}{g W \cos \theta}\right)^{0.5}$$  \hspace{1cm} (14)

Where:

$q \text{ [cm}^3\text{/s]} = \text{the flow applied to a plate of width } W \text{[cm]}$.  

For trickling filters typical average flow velocities are 1 to 10 cm/s and fluid thicknesses are around 100 µm. This results in Re of 1 to 10 and flow conditions that will be assumed to be laminar, even though laminar conditions are not rigorously justifiable for Re $\geq 6$. Lechví11 has shown wavy flow can increase oxygen transfer by about 15% to thin films, and Swilley22 has shown that wavy flow can increase substrate removal by biofilms; it is expected that substrate removals for flow conditions with high Re may be underestimated by assuming laminar flow conditions throughout this study. A further wavy film discussion was provided by Brauner and Maron32 and Brauner,33 a comprehensive thin film review was provided by Fulsord.34

Two factors that might affect the proposed hydraulic model for inclined surfaces covered with biofilms are: the entrance length over which a parabolic velocity profile is established and the surface roughness. Considerable research has been done on laminar fluid entrance regions over inclined surfaces.35-38 According to calculations by Stucheli and Ozisik,39 undisturbed laminar flow is developed within an entrance length of $L_{ent}$, given by

$$L_{ent} = 0.3 \text{ Re } \delta_f$$  \hspace{1cm} (15)

Where:

$\delta_f = \text{the fluid thickness under fully developed laminar flow conditions}$.  

Because Re for fluid flows are around 10 in this study, entrance effects should be restricted to within three fluid thicknesses along the flow path, of 0.03 cm. For laminar flow, surface roughness should not affect the flow profile. It is generally accepted that viscous effects at low Re damp out flow irregularities caused by surface protrusions.39 Laminar flow conditions therefore, can be assumed for rough biofilms. This assumption concerning biofilm roughness would not be true for turbulent flow conditions as shown by Picolouglo et al.40

Assuming a substrate reaction absence in the thin fluid film, the non-steady substrate transfer in the fluid film with respect to depth $(z)$ and flow length along a plate $(x)$ from a mass balance is described by

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial x^2} v(x) \frac{\partial C}{\partial z}$$  \hspace{1cm} (16)

The appropriate boundary conditions are:

$$z = 0 \quad C = C_0$$  \hspace{1cm} (17)

$$x = 0 \quad \text{Oxygen: } C = C^*$$  \hspace{1cm} (18)

$$\text{BOD: } \frac{\partial C}{\partial x} = 0$$  \hspace{1cm} (19)

$$x = \delta_f \quad D \frac{\partial C}{\partial x} = C_i D \left( \frac{3 E_b (1 - \epsilon)}{a^2} \right)^{0.5}$$  \hspace{1cm} (20)

At the plate entrance, the thin fluid film is assumed completely vertically mixed and uniform in concentration $C_0$ (Equation 17). Substrate is either biodegradable organics as currently mea-
sured by the soluble BOD test (as discussed below), or dissolved oxygen; therefore, two different boundary conditions are necessary for these two substrates at the air-liquid interface (x = 0). Oxygen concentration at the air-liquid interface is assumed to be the equilibrium saturation concentration of oxygen C* (Equation 18); for sBOD, it is assumed that there is no mass flux into air, or that there is no BOD constituents volatilization (Equation 19). The final boundary condition (Equation 20) specifies that substrate flux out of the fluid is determined according to biofilm kinetics specified in Equation 12. A biofilm porosity of 0.8, and a cell radius of \( a_r = 1 \mu m \) are assumed. Temperature corrections to diffusivities are incorporated using a common correction term,\(^{39}\) or

\[
D_T = D_{20} \frac{T}{293} \frac{\mu_{20}}{\mu_T}
\]  

(21)

Where:

the subscript 20 indicates the parameter value of 20°C (293°K), and temperature is in Kelvin.

No empirical corrections for temperature are made to the kinetics in Equation 12 other than those that result from changes in substrate diffusivity or fluid viscosity.

Similar biofilm model applications have been investigated by other researchers in the laboratory. Maier et al.\(^{41}\) for example, developed a model to describe substrate removal by biofilms and postulated instantaneous substrate removal at the biofilm surface, or that substrate removal is completely controlled by mass transfer limitations of diffusion to the biofilm surface. It will be shown that although diffusion limitations are more important than reaction limitations, reaction cannot be neglected in analysis mass transfer to biofilms. Vaughan and Holder\(^{21}\) used a biofilm model similar to the model proposed in this work, but based on zero order kinetics; their model and laboratory data were not in agreement, and the glucose concentrations used in their study were not mentioned. Justifying zero order kinetics is difficult because of the bulk concentration at cell surfaces which is reduced at the liquid-biofilm boundary layer and within the biofilm itself to concentrations below those that would cause zero order kinetics.

A similar model was examined by Swilley\(^{22}\) and coworkers.\(^{52,43}\) Unfortunately, Swilley did not develop solutions for oxygen transfer, and his solutions for substrate removal are presented in graphical form, which yields imprecise readings. Moreover, model applications for substrates such as domestic wastewater was not addressed. Swilley's experimental work was carried out for an inclined plate using glucose for an energy and carbon source and nitrate for an oxygen source. Reynolds numbers tested were 10 to 100, which suggests flow was best characterized as wavy, and not laminar. Atkinson et al.\(^{45}\) reported good model agreement with laboratory data until biofilm growth became excessive and pools and channels were created along the biofilm. Glucose concentrations of 100 to 200 mg/L were used, which is roughly 100 to 200 mg-BOD/L. These concentrations may be too large to justify first order kinetics because of glucose diffusivity, and the substrate cell-to-rate transfer which can exceed the maximum microorganisms growth rate. Logan\(^{28}\) has suggested that glucose concentrations be kept less than 85 mg/L for biofilm experiments, to assume the simple first order uptake kinetics case. Glucose concentrations used in studies by Swilley\(^{22}\) and others\(^{41,44,45}\) may similarly result in mass transfer of glucose exceeding the maximum predicted uptake rate of microorganisms.

THE TRICKLING FILTER MODEL

Using the substrate transport equations, developed in the previous sections, a computer model predicted sBOD removal and oxygen transfer in trickling filters. Methods are developed below to characterize different plastic media geometries and domestic wastewater components, this information is used in a numerical scheme to solve the mass transfer equations.

Plastic media geometry. Trickling filter media can be separated into four categories of: cross-flow (XF); vertical flow (VF); tubular (TB); and random. The first three types of media surfaces are presented in Figure 2. Cross-flow media consists of ridged sheets, joined so that adjacent ridges form an x pattern. Vertical media contains both wavy patterned sheets and straight vertical sheets. Tubular media contains only straight vertical tubes. These three plastic media are constructed as modules of varying volume, although VF and XF media are usually 60 cm (2 ft) in depth. The only tubular media examined in this study had a depth of 220 cm (7.2 ft). Random media, such as rocks or small plastic modules, were not considered.

Plastic media geometry and specific surface area differences result in different lengths of uninterrupted fluid flow, (flow length during which the liquid film velocity profile is not disrupted) and different fluid film thicknesses. Characterizing plastic support media requires a further explanation of uninterrupted flow for different media geometries. In Figure 3, two flow paths that con-

Figure 2.—Media used in analysis: (a) cross-flow, (b) vertical-flow, and (c) tubular.
fluid rippling or wavy flow in the fluid—and not a wavy or discontinuous surface in support media—could explain observed mass transfer rates. Since Reynolds numbers of 10 to 100 were used in their study, their wavy fluid flow explanation is reasonable. Therefore, it is assumed that convex or concave discontinuities, or waves or protrusions in the plastic media, are insufficient to generate mixing in the thin fluid film. Two flow path configurations shown in Figure 3, which are identical in length, are assumed to be hydrodynamically equivalent.

Two situations are hypothesized to result in thin fluid film mixing: fluid separation from support media and stream interception. Fluid at the module bottom may separate from the module end as a free surface stream (either a jet stream or a sheet flow) or as drops; therefore, it is assumed for all plastic media, that at the plastic module end, the concentration profile predicted by the biofilm model is disrupted and that fluid entering the next module is completely mixed and of homogeneous concentration. In XE-media, fluid flowing along one inclined surface intercepts the fluid flowing along the opposing sheet surface at several points along the module. This fluid stream interception is assumed to cause mixing in the two fluid streams. For example, in Figure 2, seven fluid interruptions are diagrammed for the XF media versus only a single fluid interruption in a VF media that occurs at the module end.

Eight plastic media modules geometries are summarized in Table 1. The media code represents the media type, media manufacturer, and specific surface area. For instance, XFa-138 indicates XF media produced by corporation a, with specific surface area of 138 m²/m³. For each module the length of uninterrupted fluid flow is defined as the plate length. The plate width and plate numbers are calculated from the manufacturers' specified surface area and module geometry such as 60 degree XF, 90 degree tube flow, and so on. Since each manufacturer provides media with slightly different geometry, it can not be assumed a priori that media of identical surface area and type (that is, vertical-flow) will perform identically to another media of the same general type.

Wastewater components diffusivities. Substances in wastewater can be characterized into four size categories as soluble, colloidal, supercolloidal, and particulate.\textsuperscript{47-49} Approximately one-third of the total organic carbon in municipal wastewater is soluble. A common method of measuring the wastewater component concentration, which comprise soluble substrates for microorganisms, is to perform a BOD test on a wastewater sample filtrate using glass fiber filter with an average pore diameter of 1.0 μm. It is likely that this sBOD test is actually a measurement of both soluble and colloidal wastewater com-

![Figure 3—Comparison of media configurations relative to fluid mixing: (a) idealized wavy media surface as a series of ridges and (b) inclined plate.](image)

TABLE 1—Summary of media geometries.

<table>
<thead>
<tr>
<th>Media code</th>
<th>Specific surface, m²/m³</th>
<th>Plate width, cm</th>
<th>Plate length, cm</th>
<th>Angle</th>
<th>Disruptions per module</th>
<th>Plates, m⁻²</th>
</tr>
</thead>
<tbody>
<tr>
<td>XFa-99</td>
<td>89</td>
<td>5.1</td>
<td>8.6</td>
<td>60</td>
<td>8</td>
<td>1 560</td>
</tr>
<tr>
<td>XFa-98</td>
<td>98</td>
<td>4.3</td>
<td>7.5</td>
<td>60</td>
<td>9</td>
<td>1 960</td>
</tr>
<tr>
<td>XFa-138</td>
<td>138</td>
<td>2.8</td>
<td>5.0</td>
<td>60</td>
<td>14</td>
<td>4 270</td>
</tr>
<tr>
<td>VFa-89</td>
<td>89</td>
<td>6.1</td>
<td>62</td>
<td>90</td>
<td>1</td>
<td>1 410</td>
</tr>
<tr>
<td>VFe-100</td>
<td>100</td>
<td>7.0</td>
<td>63</td>
<td>90</td>
<td>1</td>
<td>1 400</td>
</tr>
<tr>
<td>VFe-92</td>
<td>92</td>
<td>10.0</td>
<td>66</td>
<td>60</td>
<td>1</td>
<td>670</td>
</tr>
<tr>
<td>VFe-92</td>
<td>92</td>
<td>4.8</td>
<td>69</td>
<td>60</td>
<td>1</td>
<td>1 620</td>
</tr>
<tr>
<td>TBe-217</td>
<td>217</td>
<td>0.92</td>
<td>220</td>
<td>90</td>
<td>1</td>
<td>23 400</td>
</tr>
</tbody>
</table>
ponents, which is a mixture of cell fragments, viruses, and macromolecules such as polysaccharides, proteins, lipids, and so on.49

In Table 2, a five-component diffusivity spectrum is proposed to classify the soluble components in municipal wastewaters for this study. Using the Levine et al.49 data (their Figure 14b) and a correlation proposed by Polson,50 an average diffusivity of 112 \( \times 10^{-8} \) cm\(^2\)/s is assigned to the molecular weight distribution in 3000 to 30 000 amu. Other diffusivities in Table 2 were determined using an average diffusivity for the indicated molecular weight ranges. These diffusivities represent substrate diameters of 0.001 to 0.01 \( \mu \)m. Levine et al.49 found total organic carbon (TOC) absence in the particle size of 0.01 to 0.1 \( \mu \)m. Levine51 estimated an insubstantial soluble TOC fraction from a typical domestic wastewater to be less than 3000 amu. Particles larger than 0.1 \( \mu \)m cannot be considered in the analysis based on Brownian diffusion used in deriving the first order biofilm model (Equation 12), and are neglected for wastewater classification. Because sBOD data likely incorporates organic particles in the region of 0.1 to 1.0 \( \mu \)m in size, there will be some error involved in characterizing wastewater according to Table 2. Laboratory methods such as chromatography or membrane filtration needed to characterize wastewater components within the five diffusivity spectrum according to TOC, have been questioned by a number of researchers.72-78 relative to laboratory apparatus type. As an approximate description of a generic domestic wastewater, an equal sBOD distribution within the five components listed in Table 2 will be used. It is recommended that more accurate and wastewater specific data be obtained for use in future studies.

Computer model for trickling filter design. Equations 16 to 20 could not be solved in closed form and were solved using a finite difference numerical scheme. The mass balance in Equation 16 can be written in terms of differences as:

\[
\frac{\Delta C}{\Delta t} = \frac{D}{\Delta x^2} \left[ C_{x-1,z} + C_{x+1,z} - 2C_{x,z} \right] - \frac{v_x}{\Delta x} \left[ C_{x,z} - C_{x-1,z} \right] \tag{22}
\]

Where:

\[ C = \text{the concentration}; \]
\[ D = \text{the diffusivity of either component of sBOD or oxygen}; \]
\[ v_x = \text{the velocity of fluid at the node, and the subscripts \( x \) and \( z \) refer to the node locations}. \]

A 10 \( \times \) 10 grid of nodes was used for all analyses, resulting in:

\[ \Delta z = L/10 \quad \text{and} \quad \Delta x = \delta_f/10. \]

Where:

\[ L = \text{the plate length of various media defined in Table 1, and} \]
\[ \delta_f = \text{the fluid film thickness}. \]

Table 2—Diffusivity spectrum of soluble BOD in wastewater.

<table>
<thead>
<tr>
<th>Molecular weight 10^3, amu</th>
<th>Diffusivity × 10^5 (cm²/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>3-30</td>
<td>112</td>
</tr>
<tr>
<td>30-50</td>
<td>85</td>
</tr>
<tr>
<td>50-100</td>
<td>65</td>
</tr>
<tr>
<td>100-500</td>
<td>50</td>
</tr>
<tr>
<td>500-1000</td>
<td>30</td>
</tr>
</tbody>
</table>

Table 3—Effect of \( E_B \) factor on percent substrate removal.

<table>
<thead>
<tr>
<th>( E_B )</th>
<th>Cross-flow media, XFe-98</th>
<th>Vertical-flow media, VFb-100</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.01</td>
<td>82</td>
<td>67</td>
</tr>
<tr>
<td>0.005</td>
<td>77</td>
<td>63</td>
</tr>
<tr>
<td>0.0035</td>
<td>73</td>
<td>61</td>
</tr>
<tr>
<td>0.001</td>
<td>60</td>
<td>52</td>
</tr>
</tbody>
</table>

Boundary conditions (Equations 17 to 20) were used in the finite difference program to define substrate concentrations at nodes or concentration gradients at boundaries. For example, fluid at the plate entrance is of uniform concentration \( C_0 \), and defining \( C_{x,\text{exit}} = C_0 \), satisfies the boundary condition in Equation 17. At the air-liquid interface, \( C_{x,\text{air}} = C^* \), where \( C^* \) is the equilibrium oxygen concentration of the liquid. Because there is no sBOD flux across the air-liquid interface, the terms \( D/\Delta x^2(C_{x-1,z} - C_{x,z}) \) are defined to be zero in Equation 22 to satisfy the boundary condition in Equation 19.

Equation 22 represents a non-steady state mass balance. Steady state, substrate flux into biofilm was calculated by successive iteration of Equation 22 until the following convergence criteria was met:

\[ |\tilde{C}(t) - \tilde{C}(t + \bar{\tau})| < 0.0001 \tag{23} \]

Where:

\[ \tilde{C} = \text{average substrate concentration leaving the plate at times} \]
\[ t \text{ and } t + \bar{\tau}, \text{ and} \]
\[ \bar{\tau} = \text{average residence time of fluid over a plate, defined as} \]
\[ \text{plate length divided by average fluid velocity}. \]

Steady-state substrate removal was defined as obtained when the substrate concentration at the plate exit did not change by more than 0.0001. This condition occurs when the substrate flux based on a concentration gradient in the liquid at the liquid-biofilm interface is equal to the substrate flux calculated using the boundary condition in Equation 20.

Using the above numerical scheme, the sBOD removal percent was calculated for one, uninterrupted flow plate. The overall sBOD removal in the filter is:

\[ f = \sum_{i=1}^{5} (1 - f)^n \tag{24} \]

Where:

\[ f = \text{the fraction of sBOD component removed along a plate}; \]
\[ n_p = \text{the total number of plates of a filter of height } H \text{ for each specific media geometry}; \]
\[ f = \text{the combined fraction of all sBOD components remaining in the trickling filter effluent}. \]

Full computer program details used to model sBOD removal are in Reference 29.

RESULTS

Model calibration. With the plastic support media geometries and substrate diffusivities defined as previously discussed, only the value of the collector efficiency, \( E_B \), was required for model calibration. In Table 3, the sensitivity of substrate removal per-
cent in a 10 module (6 m) tower to the value of the collector efficiency is presented for XF and VF trickling filter media. As the collector efficiency increases, the total substrate removal increases. For collector efficiencies from 0.001 to 0.01, substrate removal increases from 60–82% in the XF media, and from 52–67% for VF media, respectively. A ten-fold increase in collector efficiency increases microbial uptake kinetics and substrate flux into biofilm less than 30%. A larger percentage increase in removal exists for the XF media than VF media for these values of \( E_B \) because of more fluid disruptions in XF media. A value of \( E_B = 0.0035 \) was determined by comparing model predictions with XF media (XFa-98) trickling filter data of Brown and Caldwell\(^{59} \) for the Garland, Tex., plant.

Collector efficiency calibration depends on five diffusivities that were defined for the sBOD components. Shown in Table 4, are the percent removals for each of the sBOD components in a 10 module (6 m) trickling filter assuming \( E_B = 0.0035 \). As substrate diffusivity increases, the total percent removal of the substrate is also predicted to increase. For substrate diffusivities of 30 and \( 80 \times 10^{-8} \) cm\(^2\)/s, substrate removals are 54 and 81%, respectively, in XF media filter. Total percentage sBOD mass removed for the two media are shown to be 73 and 61% for XF and VF media, respectively.

Variability in wastewater composition results in different sBOD percent removal predictions for a trickling filter. For example, in XF media trickling filter (XFa-98), 81% sBOD removal is predicted when wastewater is largely composed of sBOD with a diffusivity of \( 80 \times 10^{-8} \) cm\(^2\)/s; if wastewater composition changes so that the sBOD substrate diffusivities are mostly composed of organics with \( D = 50 \times 10^{-8} \) cm\(^2\)/s, only a 68% sBOD removal is predicted. Variability in wastewater composition may have a large effect on the observed sBOD removal in a trickling filter. Applying the trickling filter model is, therefore, restricted to wastewaters that have similar compositions to wastewater for which this model was calibrated.

**Data for model validation.** The proposed trickling filter model was tested for accuracy by comparing model predictions with several sets of laboratory, pilot plant, and full scale data. A data set summary used in this study is presented in Table 5. Of these studies, one was a laboratory study and three were full-scale studies. The remainder are pilot plant studies of which three contain parallel tests of different plastic media. Using this data, various aspects of trickling filter performance were further analyzed.

**Fluid disruptions effect.** The overall performance of cross flow (XFa-98) and VF (VFb-100) media is compared in Figure 4 for substrate removal in a 10 module (6 m) tower. sBOD percent removed is shown as a loading rate function, assuming oxygen does not limit the sBOD removal rate, at a wastewater temperature of 20°C. At a hydraulic loading rate of 0.68 L/m\(^2\)/s (1.0 gpm/ft\(^2\)), the vertical media is predicted to remove 60% of the

<table>
<thead>
<tr>
<th>Substrate diffusivity ( D \times 10^{-6} ) [cm(^2)/s]</th>
<th>Cross-flow media, XFa-98</th>
<th>Vertical flow media, VFb-100</th>
</tr>
</thead>
<tbody>
<tr>
<td>112</td>
<td>89</td>
<td>78</td>
</tr>
<tr>
<td>80</td>
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<td>30</td>
<td>54</td>
<td>41</td>
</tr>
<tr>
<td>All substrates</td>
<td>73</td>
<td>61</td>
</tr>
</tbody>
</table>

**Table 4—Effect of Diffusivity on percent substrate removal.**

**Figure 4—Predicted percent of sBOD removals for 10 module (6 m) trickling filters using cross-flow (XFa-98) and vertical-flow (VFb-100) media.**
applied sBOD after flow through 10 modules versus 75% sBOD by the XF media. Figure 4 can also estimate that a XF media load could be increased by 50% to 1.0 L/m²s (1.5 gpm/ft²) to achieve an equivalent sBOD removal as vertical-flow media loaded at 0.68 L/m²s. Parker and Merrill similarly predicted that the superior XF media performance could result in a 50% reduced XF media requirement compared to VF media. Media performance prediction in Figure 4 is different from previous trickling filter models in that the increased substrate removal is ascribed to a physical mechanism—fluid disruption. In a modified Velz analysis, differences in media result in different treatability coefficients, which implies that the microbial kinetics are different in biofilms grown from the same wastewater but on different support media. In this model, kinetic constants are unchanged when comparing all media performance; differences in substrate removal for different media are because of differences in media geometries.

**Temperature effect.** Temperature effect on sBOD removal is shown in Figure 5 for a 10 module tower of XF (XFa-98) media. At a hydraulic loading of 0.68 L/m²s (1.0 gpm/ft²), for example, sBOD removal is predicted to increase from 73 to 80% for a wastewater temperature increase of 20° to 30°C. Using the modified Velz equation, 73 and 84% removals are calculated, assuming a \( k_T = 0.0019 \) (L/m²s)² (2.3 × 10⁻³ gpm/ft²)⁹. Increased removals for the proposed model result from changes in physical parameters because no biological corrections to the kinetics, such as the \( \theta_{7-20} \) factor in the modified Velz equation, have been used. As fluid temperature increases, fluid viscosity decreases resulting in an increase in the average fluid film velocity (decreased fluid residence time). However, increases in substrate diffusivities (Equation 12) result in a larger substrate flux to the biofilm, and an overall increase in sBOD removal amount is predicted.

### Table 6—Glucose removals in inclined plate laboratory experiments.

<table>
<thead>
<tr>
<th>Glucose feed concentration, mg/L</th>
<th>Observed</th>
<th>Maier et al.⁴¹</th>
<th>This study</th>
</tr>
</thead>
<tbody>
<tr>
<td>27</td>
<td>14</td>
<td>16</td>
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<tr>
<td>277</td>
<td>7</td>
<td>16</td>
<td>15</td>
</tr>
</tbody>
</table>

**Loading rate effect.** The main factor predicted to affect sBOD removal for a specific trickling filter media, once the substrate characteristics are fixed, is hydraulic loading rate. In Table 6, the glucose removals observed and predicted by Maier et al.⁴¹ for a laboratory inclined plate apparatus 11 cm wide and 60 cm long with a 0.018 cm/cm slope are compared with removals predicted by the trickling filter model, for hydraulic load of 1.33 cm²/s. Glucose (\( D = 600 \times 10^{-6} \) cm²/s) was enzymatically measured. The Maier et al. model was based on instantaneous substrate use at the biofilm surface. They claimed good agreement between model predictions and laboratory data for glucose concentrations below 100 mg/L. The trickling filter model predicts similar removals for these glucose concentrations. Above an inlet glucose concentration of 100 mg/L, neither model agrees well with the observed removals. This lack of agreement is expected from the first order kinetic model for glucose concentrations less than 85 mg/L.

In Figures 6 and 7, the predicted sBOD percent removed as a loading rate function is compared with pilot plant data⁶ for VF (VFb-100) and XF (XFa-98) media. Model predictions are
Figure 7—Predicted effect of hydraulic load on sBOD removal for cross-flow (XFa-98) media compared with the data of Richards.40

Figure 8—Predicted hydraulic load effect on sBOD removal for vertical-flow (VFb-92) media compared with the data of Hutchinson.62

Figure 9—Predicted sBOD profiles for cross-flow (XFa-98) and vertical-flow (VFb-100) trickling filters versus the data of Richards,40 for an average hydraulic load of 0.48 L/m²s (0.7 gpm/ft²).

compared with the average of approximately 15 data points at three hydraulic loadings; error bars are based on one standard deviation. Five of the six comparisons shown in these two figures lie within one standard deviation of the observed data indicating the model predictions accuracy. Some data scatter in these two figures may be caused by temperature fluctuations because only an average wastewater temperature was available for each data point, which averaged several weeks of performance. Model predictions were made assuming an average wastewater temperature for all data at each hydraulic loading.

Model predictions were also compared with data reported by Hutchinson42 over slightly larger loading rates of 0.21 to 1.11 L/m²s (0.3–1.6 gpm/ft²). One such comparison is shown for sBOD removal as a function of loading for another VF media type (VFb-92) in Figure 8. Model and data agreement is good. These results are particularly important because influent sBOD’s were around 200 mg/L versus an average influent sBOD of 60 mg/L in the previous two figures. This supports a model assumption that influent sBOD magnitude is not a factor in predicting sBOD removals as long as the influent sBODs are within the range where first order uptake kinetics can be assumed.

sBOD profiles. Several researchers have investigated sBOD concentrations at multiple depths in trickling filters. This research provides valuable data to compare with the proposed model on the sBOD removal rate plus overall removals. In Figure 9, the observed sBOD concentrations for every 1.2 m (2 modules) is shown for XF (XFa-98) and VF (VFb-100) media using data from a pilot plant study in Atlanta, Ga.40 Model predictions are shown at every module and are connected by a line for visual comparison purposes. For the XF-media profile, the observed data shows the same sBOD removal rate decreasing with depth as the model. The predicted effluent sBOD and the sBOD at various depths are seen to fit the data. There seems to be much
more scatter in the data for the VF media than for the XF media; according to the VF media data, sBOD actually increases for the last measurement. The model predictions fall at intermediate values and are considered to fit the observed data reasonably well.

Predicted sBOD profiles for trickling filters at higher organic loading rates also showed good agreement with observed removals using pilot plant data reported by Hutchinson for a loading rate of 0.21 to 1.11 L/m².s. In Figure 10, an sBOD profile for a VF (VFc-92) media loaded at 0.41 L/m².s (0.6 gpm/ft²) with an influent sBOD in excess of 200 mg/L is shown; both profile and effluent sBOD predictions agree with observed data.

Recycle effect on sBOD profiles is examined using data from the Garland, Tex., pilot plant. Hydraulic loading rates at this site averaged 0.4 L/m².s (0.6 gpm/ft²). An sBOD profile for a recycle ratio (R) of 0.62 is shown in Figure 11. The recycle ratio is defined here, in the model, and in subsequent discussions, as the fraction of total applied hydraulic filter load that is recycled effluent. Good agreement is shown for model and data comparisons at the filter bottom. Unfortunately, an sBOD at the intermediate depth of 1.2 m (4 ft) was not sampled. One confusing characteristic of this data set that is exhibited in all profiles examined is the consistently low sBOD predicted from mixed recycle and primary effluent. In Figure 11, for example, if primary effluent of 130 mg-sBOD/L was mixed with water containing no sBOD at a ratio of 0.38 primary effluent to 0.62 water, the resulting sBOD should be approximately 50 mg/L. Because the recycled effluent contains some sBOD, it does not follow from a mass balance that the mixture contains 39 mg/L sBOD as indicated in Figure 11. This consistently low influent sBOD may be a result of one or a combination of factors, including: sampling error that can occur from incompletely mixing recycle and primary effluent; suspended cells in the recycle that may remove sBOD, although this uptake is expected to be negligible; and sBOD analysis error, which is discussed below. It is unknown to what extent these previous factors may have contributed to the reported data.

Overall change in sBOD. The proposed model's usefulness is dependent on the agreement between predicted trickling filter effluent sBOD concentrations and observed data. One way to evaluate model accuracy over a wide influent sBOD range is to compare the overall observed change in sBOD with its overall predicted change. The Stockton data, based on VF media (VFb-99), is shown in this comparison type in Figure 12. This data set is characterized by high recycle ratios (0.7 to 0.8) and influent sBODs up to 200 mg/L. The agreement between observed and modelled changes in sBOD for this study is very good. If predicted removal agrees exactly with the observed removal, the points would lie on the 45 degree line drawn on the graph. Because sBOD test precision is only 10 to 20%, agreement of observed data with model predictions by more than this amount is fortuitous.

Effluent sBOD comparisons. Another method used to evaluate overall model predictions accuracy is plotting average influent and effluent sBODs compared to predicted effluent sBODs. As in the previous comparison method, the intrinsic operating effects on recycle ratio or loading are not apparent; these plots reflect the total parameter result on observed and predicted sBOD in trickling filter effluent. In Figure 13, the consecutive weekly data are shown for a pilot plant composed of XF (XFa-98) media.
The predicted sBOD in filter effluents is similar to observed effluent sBOD except for a data point at week 6.

DISCUSSION

For the last 20 years, engineers have used simple design equations, such as the modified Velz equation, to design trickling filters\textsuperscript{2,4,53} that are based on kinetically limited substrate removal. Model applications to different plastic media types used in trickling filters has resulted in different constants to predict sBOD removal for identical wastewaters. The proposed model defines hydraulic flow through trickling filters better than previous models and has been validated through comparison with a variety of pilot plant and full scale trickling filter studies. There was good agreement between model predictions and observed data for sBOD removal as a function of hydraulic loading. There were also generally accurate sBOD concentrations predictions at various depth within the filter. Model accuracy was good enough that in one case, when a data set differed substantially from the model predictions, it was checked with the investigator and found to be mis-labelled.\textsuperscript{70}

The increased XF media performance as compared to VF media results from disrupting the thin film boundary layer above the biofilm, and not different microbial kinetics as implied by the modified Velz equation, because the kinetic constants were fixed throughout this study. Filter performance is predominantly attributed to mass transfer limited substrate removal rather than microbial kinetics because fluid mixing greatly affects efficient sBOD removal in different trickling filter media. Flow disruption was accomplished in XF media through fluid stream interception; wavy patterns on the plastic media were not considered sufficient to induce fluid layer mixing. Because fluid near the biofilm surface lacks substrate, disrupting the fluid layer brings substrate rich fluid into contact with the biofilm. To accomplish mixing through turbulent fluid flow, the \( \text{Re} \) is so large (\( \text{Re} > 250 \)) that less overall treatment is accomplished because of insufficient hydraulic detention time in the filter. The XF media, therefore, accomplishes intermittent mixing without requiring turbulent flow rates.

The most important factor affecting the prediction of the performance of a trickling filter is wastewater composition, as was shown in Table 4. The current substrate concentration evaluation method is the indirect sBOD test method to determine the oxygen demand of all filtered organic wastewater components. sBOD analysis is a significant improvement over the total BOD test because the task of modelling substrate transport is hopefully confined to soluble components. However, the sBOD test is only 10% to 20% reproducible and colloidal organics are likely present in the filtrate. Sarner\textsuperscript{4} compared the sBOD measurements at two laboratories for the same wastewater and reported a 10% to 20% variation in sBOD. Model sBOD removal predictions were compared to observed changes in sBOD in Figure 12 using a 45 degree line. The Sarner\textsuperscript{4} results indicate that model predictions accuracy can only be related to data within 10% to 20% sBOD; therefore, a difference of 5 mg/L or less between model predictions and data. An sBOD of 25 mg/L, for example, is acceptable within the sBOD test analysis accuracy.

There are two inherent difficulties with the sBOD test precision. The first, widely recognized problem is that particles larger than the nominal filter pore diameter may pass through the filter. A Whatman glass fiber filter with nominal pore diameter of 1.0 \( \mu \text{m} \) is typically used in the sBOD analysis; voids between wound glass fibers could easily allow passage of 1.0 \( \mu \text{m} \) or larger particles, although on the average particles of 1.0 \( \mu \text{m} \) or larger are captured. The second difficulty with a filtered sample, which is not generally recognized, is that filter efficiency increases with surface loading. Particle accumulation on the filter may decrease the nominal
pore size and increase the filter surface area for particle capture. Particles with substantially less size than the original or intended pore diameter can be captured. This may result in the unintentional removal of sBOD components. Johnson and Wangersky\textsuperscript{71} found that a glass fiber filter with nominal pore diameter of 1.2 µm retained up to 20% of beads 0.176 µm in diameter at high filter loadings. Danielsson\textsuperscript{72} examined particle and color removal by membranes and glass fiber filters and found the membranes removed significant amounts of soluble and colloidal components smaller than nominal pore diameter. Membranes were judged to be unsuitable for separation of particulate and dissolved material. Danielsson found that a 0.7 µm Whatman glass fiber filter achieved satisfactory passage of particles smaller than the filter pore size, but that a similarly sized Spectrograde glass fiber filter resulted in excessive small particle filtration. These results indicate that heavy filter loadings should be avoided to achieve sBOD separations from wastewater samples.

A general trend shows decreasing percent removal with increased loading for all data sets and is predicted by the trickling filter model. This trend results from hydrodynamic considerations and occurs as long as wastewater composition is constant. Calibrating the model to predict a particular sBOD removal implicitly assumes that all wastewater components were identical. This may not be true in all cases, and might account for the observed wide variations in sBOD removal in some trickling filters. Sarner,\textsuperscript{1} for example, reported sBOD removal in a 4.5 µm (15 ft) deep trickling filter filled with XF media (XFa-138). The observed sBOD removals for hydraulic loads of 0.5 to 3.4 L/m²s (0.8 to 5 gpm/ft²) are compared with model predictions in Figure 14. Sarner’s data is characterized by large variability in percent sBOD removed at a particular hydraulic load. For example, at 0.68 L/m² (1 gpm/ft²) between 50 to 85% of sBOD was observed to be removed, whereas less than 10% variation is predicted using the model for reported temperature variations. In general, the predicted removals were less than observed removals, especially at higher hydraulic loads of 2 to 3.4 L/m²s (3–5 gpm/ft²). The Reynolds numbers for the higher loadings were 18 to 25, which indicates flow was no longer laminar. These high Reynolds numbers, plus variability in wastewater composition, could account for the greater observed removals at higher hydraulic loadings.

Improvement in the hydrodynamic model may be warranted. Assuming a parabolic velocity profile in thin falling films has been supported by research on mass transfer from air to liquids. The uncertainty in the hydrodynamic model relates not to laminar flow assumption at low Reynolds numbers, but to identifying the biofilm-liquid interface. There are numerous indications that microorganisms can extend into moving fluid or that there is advective flow through some biofilm part. Siegrist and Gujer\textsuperscript{73} found biofilm was relatively flat and dense when grown on substrate such as peptone, but biofilm was loose and contained many protruding filaments when substrate was primary effluent. They attributed calculated diffusivities that were 140% larger than molecular diffusivities to eddy penetration into the biofilm matrix. It is well recognized that biofilms absorb water like a sponge; this observation does not necessarily support the concept of flow through the biofilm. However, Picologlou\textsuperscript{40} found that head losses in pipes containing biofilms were substantially larger than could be accounted for by a decrease in pipe diameter because of biofilm growth. The increased head losses were attributed to filament protrusion through the laminar sublayer into the turbulent boundary layer. Their results support advective flow past microorganisms comprising a biofilm.

Adective flow through some biofilm part may explain the findings of many researchers investigating mass transfer to biofilms, which have resulted in apparent diffusivities that are calculated to be greater than molecular diffusivities. Experiments by Ho et al.\textsuperscript{18} found apparent diffusivities 1.6 times the molecular diffusivities. Tomlinson and Snaddon\textsuperscript{74} found that substrate diffusivities fitted to a diffuse transport model for fixed films ranged from 67 to 1000% of the corresponding diffusivities in water. A model based on advective flow through all biofilm is not justified because mass transfer models based on a completely mixed biofilm and fluid layer, or a homogeneous mass transfer model, have failed to describe substrate removal by biofilms.\textsuperscript{22} Flow may exist through the upper biofilm portion exposed to the bulk fluid. One way researchers have incorporated a thin layer with both fluid and biofilm is through constructing a stagnant fluid layer. According to the model of Williamson and McCarty,\textsuperscript{19,20} the stagnant layer is composed of two depths L1 and L2, where L2 = bioplus stagnant fluid and L1 = stagnant mass transfer boundary layer. Fluid above this layer, L1 + L2, is defined as consisting of a bulk substrate concentration. Although such stagnant layers do not really exist, they may serve as conceptually useful constructs; they are not, however, physically accurate models. Logan\textsuperscript{25} has shown that the boundary layer changes in thickness during flow over a theoretical plate and can extend across the full fluid film thickness. Describing this situation as a stagnant boundary layer is inappropriate, and should be avoided for biofilms in trickling filters.

Although reported average biofilm densities (for example, Williamson and McCarty\textsuperscript{19,20}) do not result in the prediction of

![Figure 14](image_url)
advective flow through the biofilm, filaments extending into the fluid results in reduced biofilm densities in comparison to the bulk biofilm densities. Uptake by the filament increased as a result of higher substrate concentrations and flow past the filament in comparison to filaments within the denser biofilm. As more filaments grew into the moving fluid the microbial density increases and advective flow through the extended filaments decreases. This process continually thickens the biofilm and raises the advective flow area above the support surface. Eventually the biofilm sloughs, and the process repeats itself. This advective flow model requires unavailable information on biofilm densities as a biofilm depth function, and is not further pursued at this time.

CONCLUSIONS

The trickling filter model successfully predicted sBOD removal in plastic media trickling filters. It assumed that substances in wastewater consisted of soluble organics as measured by a sBOD test, and that these substances could be classified according to five diffusivities. Trickling filter performance was predicted by modelling substrate transport in a thin fluid film and substrate removal by the biofilm. The model success indicates that the soluble organic removal rate in trickling filters is primarily controlled by diffusion of components to the biofilm and not uptake kinetics within the biofilm. The superior XF media performance compared to VF media is a result of concentration gradient disruption in the thin fluid film. It is likely the proposed model underestimates sBOD removals for hydraulic loadings above 2 L/m² s (3 gpm/ft²) because of the development of wavy flow versus the assumed laminar flow conditions in the thin fluid films. Comparing model predictions with observed data suggests that sBOD test is an imprecise biodegradable organics measurement. A new test is needed to classify biodegradable wastewater components according to size or diffusivity. Further refining the proposed model is not warranted until more precise data sets can be obtained.

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