

Oxygen flux limitation in aerobic fixed-film biotreatment of a hazardous landfill leachate

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Abstract

A bench-scale, continuous-flow submerged aerated biofilter was used to treat a RCRA-hazardous landfill leachate under steady- and non-steady hydraulic and organic loadings. At 1.37 day empty bed contact time and constant hydraulic flow rate, the biofilter achieved a removal efficiency of 76% for chemical oxygen demand (COD), equivalent to 95% of estimated biologically degradable COD in the influent. Aerobic fixed-film treatment achieved substantial removals of several RCRA-listed organic chemicals, but best demonstrated available technology standards were not met for all compounds. Biofilter COD removal efficiency decreased dramatically following hydraulic and organic shock loading. Mathematical modeling of oxygen and electron donor flux into attached biofilm suggests that oxygen diffusion may limit the ability of aerobic fixed-film reactors to respond to the sudden increases in flow rate which are common to landfill leachates.

Keywords: Bioreactor; Fixed film; Leachate; Perturbation; Oxygen flux; Model

1. Introduction

The design of biological systems to treat hazardous landfill leachates is confounded by variable hydraulic and organic loadings, the presence of toxic compounds, and the need to meet often stringent effluent limitations for specific organic compounds [1–4]. Packed-bed bioreactors offer economic advantages for leachate treatment because microorganism attachment to packing surfaces makes possible lower hydraulic retention times and smaller reactor volumes [5–9]. Biofilm reactor design is complicated by variable organic and hydraulic loading, the presence of multiple component wastestreams, and potential inhibition by toxic chemicals. Despite recent advances [10], limited information is currently available to design confidently fixed-film

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leachate treatment reactors that will reliably meet effluent quality goals under the varying conditions of full-scale application [11].

The microbial mass accumulation in a fixed-film bioreactor is controlled by the time-averaged organic loading to the treatment system, and an attached, steady-state biological mass may take weeks to become established [12]. The flow rate of landfill leachate, however, can change dramatically at a relatively short time scale due to incident precipitation on the landfill. Microbial films in fixed-film reactors, established in proportion to the average flow rate and organic concentrations, can be suddenly exposed to higher organic loadings when leachate flow rate increases, leading to possible deterioration in effluent quality. The ability of an aerobic fixed-film bioreactor to meet effluent quality goals in non-steady operation following applied shock loading is a critical design issue for biofilter application to landfill leachates, and exerts a strong influence on the required capacity of upstream equalization facilities.

This paper evaluates the ability of an aerobic biofilter to treat a hazardous waste landfill leachate under constant hydraulic loading (steady) and under applied hydraulic shock loading (non-steady) conditions. Leachate from a hazardous waste industrial landfill was biologically treated in a continuous-flow bench-scale upflow aerobic filter. The leachate contained volatile and semi-volatile organics which exceeded best-demonstrated-available-technology (BDAT) standards for multi-source leachate wastewater (MSLWW) established under the Resources Conservation and Recovery Act (RCRA). To gain insight into the operative factors which contribute to observed biofilter performance, a completely mixed biofilm reactor model was applied to analyze aerobic filter operation under steady-state and transient conditions. Modeling predictions of effluent chemical oxygen demand (COD) were compared to estimated concentrations of effluent biodegradable COD. In aerobic biological treatment, molecular oxygen is a terminal electron acceptor for bacterial metabolism. Both electron donor (COD) and oxygen must be transported to biofilm organisms to accomplish aerobic reactions. Electron donor and oxygen concentration profiles were modeled through the depth of the biofilm to gain quantitative insight into the effect of diffusion limitations on bioreactor performance [13]. Based on the model simulations, it appeared that the steady-state aerobic biofilter was on the verge of oxygen flux limitation. Modeling results suggest that the ability of the aerobic filter to respond to organic shock loading may be hampered by oxygen flux limitations under non-steady operation.

2. Materials and methods

The leachate was generated from an industrial waste landfill with a 3 ha active disposal area. Leachate samples for treatment were collected from a 1500 m³ collection tank into which the leachate currently flows and from which it is batch discharged. The leachate treated in this study was collected from the top meter of the water column of the collection tank, below the tank surface where a separate oily layer sometimes forms.

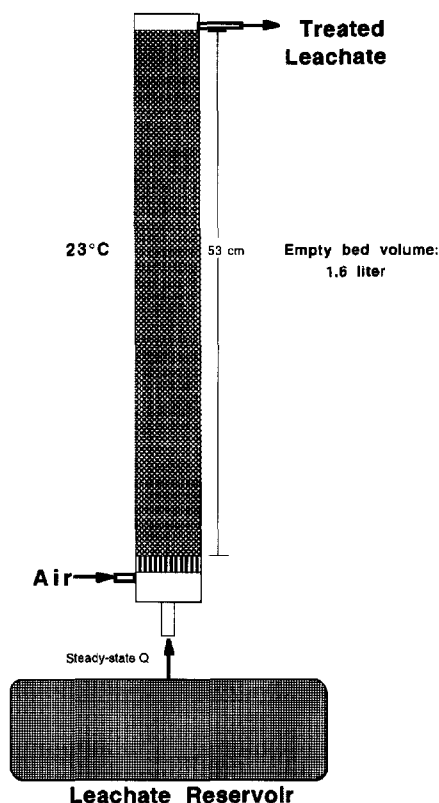


Fig. 1. Laboratory aerobic biofilter.

A submerged upflow aerobic biofilter without recycle was operated in the laboratory at 23 °C (Fig. 1). Leachate samples were collected weekly or biweekly and supplemented with nitrogen and phosphorous in excess of stoichiometric requirements for net biomass synthesis. The biofilter consisted of a 6.2 cm i.d. glass column packed with nominal 1.59 cm random polyethylene tower packing (NSW Corp.), with a packed bed height of 53 cm (Table 1). Aeration was accomplished by introducing laboratory air through a sparging stone located below a perforated liquid distribution and packing support plate. The filter was seeded from a thermophilic aerobic reactor treating industrial sludge. A variable speed peristaltic pump (Watson-Marlow, 202U/AA) delivered leachate continuously to the biofilter from a feed reservoir maintained at 4 °C. Leachate was applied for 226 days at a steady flow rate which resulted in an empty bed contact time (EBCT) of 1.37 days. The first 130 days of operation was a start-up period. The length of this period was determined by the time needed to develop the experimental system, operating procedures, and analytical methods, and not by the time needed for the bioreactor to reach steady-state operation. The next 96 days of constant flow rate operation was the period for which steady-state results are reported below. The biofilter effluent pH varied between 7.4

Table 1
Aerobic biofilter characteristics

Parameter	Value
<i>Packed column characteristics</i>	
Internal diameter (ID), cm	6.2
Length of packing, cm	53
Cross sectional area, cm ²	30.2
Empty bed volume, l	1.60
<i>Nor-Pack^a media characteristics</i>	
Surface area/reactor volume, m ² /m ³	331
Void ratio	0.88
Total surface area, m ²	0.530
<i>Operating conditions</i>	
Flow rate, l/d	1.17
Empty bed contact time (EBCT), d	1.37
Temperature, °C	23

^a Nor-Pack 1.59 cm random polyethylene tower packing (NSW Corp.).

and 8.3 during steady-state operation, and did not fluctuate outside this range during the applied shock loadings. Untreated and effluent samples were monitored throughout the study for unfiltered COD, total organic carbon, pH, and suspended solids using standard analytical methods [14]. Analyses for specific organic compounds were performed on samples of untreated leachate collected from the leachate feed reservoir and from reactor effluent, and analyzed by EPA methods 8240 (purge and trap, packed column GC/MS) for volatiles and methods 3520/8270 (MeCl₂ extraction, capillary column GC/MS) for semi-volatiles, respectively [15].

Leachate quality fluctuated over the steady-state period due to varying precipitation levels and dilution effect, with a mean unfiltered COD of 3628 mg/l and a standard deviation of 1580 mg/l. The steady-state COD removal results reported below represent averaged removals calculated from measured influent and effluent analyses during the 96-day steady-state period. At the conclusion of the steady-state operating period, the biofilter was subjected to two hydraulic shock loads. The first shock loading was a fivefold increase in flow rate over a period of 73 h (Transient I). Immediately following Transient I, the flow rate was further increased to 12 times the steady-state flow rate, decreasing the EBCT to 0.115 (Transient II). The second hydraulic surge was applied for 24 h, after which steady-state flow rates were resumed. Samples were collected hourly during the transients and analyzed for pH and bulk organic content.

3. Biofilter model

The aerobic biofilter was modeled using as completely stirred tank liquid phase reactor (CSTR) with attached microbial films. The long liquid residence time and

continuous mixing provided by aeration justified the inherent assumption of a completely mixed aqueous phase and uniform bulk substrate concentration. A steady-state biofilm model was used to predict the flux of substrate per cross-sectional area of support surface at the given bulk substrate concentration [12, 13]. The model includes substrate diffusion through an external stagnant liquid layer, biochemical reaction stoichiometry, reaction and diffusion of electron donor and acceptor in the biofilm, and a biofilm thickness calculation (Table 2). The kinetic coefficients applied in the biofilm model (Table 3) were typical values for aerobic heterotrophic growth. The

Table 2
CSTR biofilm model equations

Electron donor

$$\mathcal{D}_{fd} \frac{\partial^2 S_{fd}}{\partial z^2} = \frac{kX_f}{K_{sd} + S_{fd}} S_{fd} \left(\frac{C_{O_2}}{K_{SO_2} + C_{O_2}} \right) \quad (1)$$

$$J_d = \mathcal{D}_{wd} \frac{S_{bd} - S_{sd}}{L} \quad (2)$$

Electron acceptor: O₂

$$\mathcal{D}_{fa} \frac{\partial^2 S_{fa}}{\partial z^2} = \frac{kX_f}{K_{sd} + S_{fd}} S_{fd} \frac{v_a}{u_{th}} \left(\frac{C_{O_2}}{K_{SO_2} + C_{O_2}} \right) + bX_a f_a(1.42) \left(\frac{C_{O_2}}{K_{SO_2} + C_{O_2}} \right) \quad (3)$$

$$J_a = \mathcal{D}_{wa} \frac{S_{ba} - S_{sa}}{L} \quad (4)$$

Energy balance

$$L_f = \frac{J_d Y}{bX_f} \quad (5)$$

CSTR material balance

$$S_{bd} = S_d^0 - J_d a \theta_H \quad (6)$$

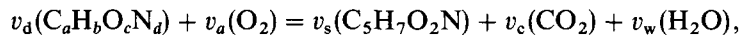
Table 3
Aerobic biofilm model parameters

Parameter, units	Value	Explanation
k , mg COD/mg VSS/d	10	Maximum substrate utilization rate
$K_{s,d}$, mg COD/cm ³	0.20	Half velocity constant for donor
K_{s,O_2} , mg/cm ³	0.0005	Half velocity constant for oxygen
Y , g VSS/g COD	0.45	Yield coefficient
b , d ⁻¹	0.15	Endogenous decay coefficient
X_i , mg/cm ³	40.0	Biofilm density
L , cm	0.0050	Stagnant layer thickness
$D_{w,d}$, cm ² /d	0.50	Substrate diffusion coefficient in water
$D_{f,d}$, cm ² /d	0.40	Substrate diffusion coefficient in biofilm
\mathcal{D}_{w,O_2} , cm ² /d	2.05	Oxygen diffusion coefficient in water
\mathcal{D}_{f,O_2} , cm ² /d	1.64	Oxygen diffusion coefficient in biofilm

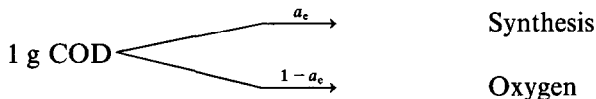
diffusion coefficient for electron donor was calculated by dividing the diffusion coefficient for acetic acid at 23 °C by 2.4 to account for the higher molecular weight of average leachate organic solutes. The diffusion coefficient for acetic acid was calculated by the Haduk Laudie equation [16]. The biofilm decay coefficient was selected to estimate both decay and detachment processes. The mass transfer coefficient from liquid phase to biofilm was estimated using the Fukuma correlation for three-phase, packed-bed reactors [17]. The Fukuma correlation predicted a mass transfer coefficient of 100 cm/day ($L = 50 \mu\text{m}$), which is less than reported values for fluidized-bed bioreactors but greater than those for non-aerated submerged biological filters [18]. The bulk liquid dissolved oxygen concentration was 8.3 mg/l, or 97% of solubility at 23 °C, in steady-state and transient simulations.

The biofilm model was solved by an explicit, forward stepping finite difference numerical technique for electron donor and acceptor [19]. The model was first used to simulate steady-state substrate and oxygen profiles and the thickness of the attached biofilm established from the steady-state substrate flux. The model was then used to simulate experimental Transients 1 and 2 using the applied hydraulic flow rates and COD concentrations measured in the transients. The biofilm thickness predicted in the steady-state solution was applied to simulate Transients 1 and 2, resulting in pseudo-steady-state concentration profiles and effluent substrate concentrations for the non-steady-state biomass in the filter. The CSTR biofilm model was also solved for the cases of five- and tenfold flow rate increase applied to the steady-state reactor, but with the organic loading rate held constant. These computer simulations were designated as Transients 3 and 4, respectively.

The biochemical reaction stoichiometry was assumed to be adequately represented by the following general equation:



where $C_aH_bO_cN_d$ presents bulk organics and $C_5H_7O_2N$ represents synthesized biomass. The fraction of electron donor used for synthesis, a_e , was assumed to equal 0.64, which is a typical value for aerobic heterotrophic substrate utilization [20]. The fraction of electron equivalents that flow to oxygen for energy generation is then $1 - a_e$:



The CSTR biofilm model equations (Table 2) were applied to steady-state biofilter operation using typical parameters listed in Table 3 and the CSTR mass balance equation:

$$S_e = S_0 - \frac{aV}{Q} J,$$

where S_e is the effluent substrate concentration, mg/cm^3 , S_0 is the influent substrate concentration, mg/cm^3 , a is the specific surface area, cm^2/cm^3 , V is the volume, cm^3 , Q is the flow rate, cm^3/d and J is the flux, $\text{mg}/\text{cm}^2/\text{d}$.

4. Results and discussion

A performance summary for aerobic biofilter operation under steady-state and transient operation is shown in Table 4. The increase in organic loading rate in Transients 1 and 2 was not proportional to flow rate increase because the influent leachate was more dilute. Landfill leachates contain organic material, such as humic-type substances, which pass through membrane filters normally used to delineate suspended organic matter [1, 3, 21, 22]. Though these high molecular mass materials are not biologically degradable under practical treatment conditions, they exert a chemical oxygen demand which is included in chemical analyses of untreated leachate and effluent [3].

To provide more insight into the effectiveness and dynamics of bioprocess action, it was considered more useful to represent effluent quality in terms of degradable COD, and disregard the inert COD component in calculating removal efficiency. An estimate of the refractory fraction of untreated COD was needed to proceed with this analysis. Based on other studies with this leachate, 20% of untreated leachate COD was estimated to be non-degradable in the biological filter [23]. This estimate was derived from biotreatment of this landfill leachate in other reactor systems. One other reactor system was an anaerobic biofilm reactor followed in series by an aerobic biofilm reactor. This sequential system was operated on the same leachate in parallel with the aerobic biofilter, with the first reactor receiving the same volumetric COD loading as the aerobic biofilter. The organic loading rate to the second stage aerobic reactor was quite low (1.05 kg COD/m³ d), less than 50% on influent COD was removed (the rest being presumably non-degradable COD). The sum removal rate from this anaerobic/aerobic system was 80%. In another, separate study, aerobic-only biodegradation of the leachate stream in a suspended growth laboratory

Table 4
Performance summary of aerobic biofilter

Parameter	Steady state	Transient I	Transient II
Flow rate, l/d	1.17	5.69	13.9
EBCT, d	1.37	0.28	0.115
Organic loading rate, kg COD/m ³ d	2.65	8.12	14.3
Organic removal rate, kg COD/m ³ d	2.01	3.41	2.55
COD (Unfiltered), mg/l			
Influent	3628	2284	1643
Degradable influent COD ^a	2902	1827	1314
Non-degradable influent COD ^b	726	457	329
Effluent	872	1323	1349
Removal efficiency	76	42	18
<i>J</i> avg ^c , g/m ² /d	6.08	10.3	7.70

^a Estimated as 0.8 × (influent COD).

^b Estimated by difference.

^c COD removal rate/total reactor surface area.

reactor (activated sludge) yielded very close to 80% total COD removal when organic loading was much lower than that to the aerobic biofilter. COD removal efficiencies of greater than 80% did not result when low COD loading rates were applied. Previous studies on other landfill leachates have reported similar and greater non-degradable organics fractions, and values of COD removal through biotreatment of as low as 10% have been reported for old leachates [3, 7]. The 20% estimated fraction of COD which is non-degradable is highly specific to this leachate. For other leachates, estimates of the non-degradable fraction of COD must be made with data specific to each leachate.

The COD removal efficiency of the biofilter was 76% at steady state, based on total COD. By correcting for the estimated 20% non-degradable COD in untreated leachate, the steady-state effluent degradable COD was 146 mg/l, or only 5% of the influent degradable COD of 2902 mg/l (Table 4). A much higher removal efficiency was calculated when inert material was subtracted from the COD balance. The non-degradable material comprised 83% of effluent COD at steady state. Similar calculations were performed for the two transients using measured analytical results.

Total COD removal efficiencies declined dramatically from 76% at steady state to 42% in Transient 1, and further to 18% in Transient 2 (Table 4). The estimated removal efficiency for biodegradable COD decreased from 95% at steady state to 53% and 22% in Transients 1 and 2, respectively. Though organic loading rate in the transients increased by over three and five times the steady-state loading rate, organic removal rates increased by factors of only 1.6 and 1.2 times in Transients 1 and 2, respectively. The relatively small increase in organics utilization in the transients suggests that some factor may limit the ability of the aerobic biofilter to respond to sudden loading increases.

Removal of several specific organic compounds by the biofilter during the steady state operating period is shown in Table 5. These data are limited, but suggest substantial removal of all of the listed organic compounds. Even with substantial fractional removals, however, effluent levels of many of the compounds were above

Table 5
Aerobic removal of RCRA-listed organics

Compound	Untreated leachate	Aerobic biofilter effluent ^a	Percent removal	BDAT standard
Acetone, mg/l	2.21	0.586	73	0.162
Acetonitrile, mg/l	0.430	0.104	76	0.097
Aniline, mg/l	0.011	0.045	—	0.81
Bis(2-ethyl-hexyl)phthalate, mg/l	0.430	0.081	81	0.278
Naphthalene, mg/l	0.170	ND ^b	100	0.059
Phenol, mg/l	3.04	0.264	91	0.026
Toluene, mg/l	0.043	ND ^b	100	0.08

^a Based on a single grab sample under steady-state operation.

^b Below analytical detection limit.

the BDAT standards. Though specific organic compounds were not analyzed in reactor effluent following shock loading, higher transient levels of specific organic compounds can be inferred from the decline in COD removal efficiency. The exact relationship between the removal efficiency of bulk organics (COD) and that of specific organic compounds is not well established for either steady-state or transient operation.

The CSTR biofilm model was applied to aerobic filter operation, first to simulate steady-state and then the results of the experimental transients. The model was used to predict the biofilm thickness that was achieved by bacterial utilization of the average supply of substrate entering the biofilter during the steady state operating period. The biofilm thickness predicted for steady state was used to model transient biofilter operation, based on the assumption that attached biomass in the filter would remain constant during the limited time duration of the shock loadings. Transient modeling of shock loadings produced quasi-steady state substrate concentration profiles and effluent organics levels. Since biofilter action operates only on degradable organics, the influent COD used in the CSTR biofilm model was 80% of the analytically measured influent COD in all cases. The following discussions of modeling results are based on degradable chemical oxygen demand except where noted.

The steady state CSTR biofilm model result is shown in Fig. 2. The energy balance assumption (Table 2, Eq. (5)) predicts that a biofilm thickness of 500 μm would be supported by steady state substrate flux. The electron donor profile shows that degradable COD approaches a low value of 2 mg/l, suggesting a biofilm that is practically deep [12]. Dissolved oxygen (DO) was predicted to decline precipitously with depth but to remain above 1 mg/l throughout the biofilm. External oxygen diffusion resistance caused a significant decline in the oxygen level from bulk liquid to the biofilm surface. For the steady state case, the CSTR biofilm model predicted an effluent degradable COD of 35 mg/l, versus 147 mg/l which was the difference between the total measured effluent COD and the non-degradable component (Table 6).

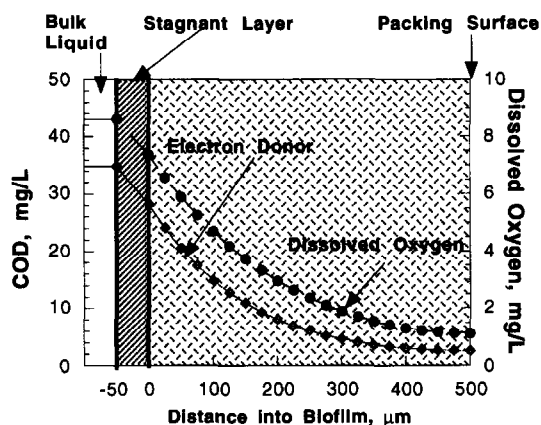


Fig. 2. Steady-state electron donor and oxygen concentration profiles.

Table 6
Aerobic biofilter effluent COD

Effluent COD, mg/l	Steady state	Transient I	Transient II
Total measured	872	1323	1349
Non-degradable estimated ^a	726	457	329
Degradable estimated ^b	146	866	1020
Degradable model	35	130	204

^a Estimated as $0.2 \times$ (influent COD)

^b Estimated as $0.8 \times$ (influent COD)

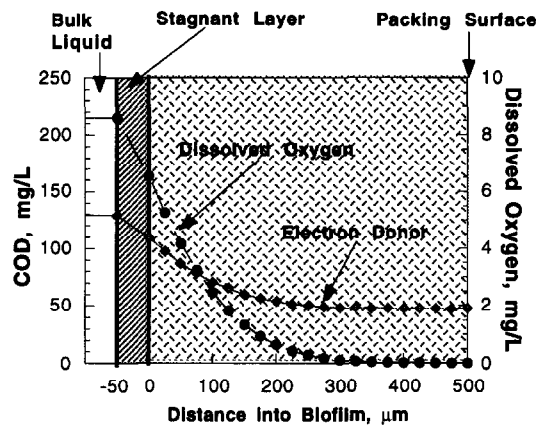


Fig. 3. Transient 1 electron donor and oxygen concentration profiles.

This discrepancy is not surprising considering the highly heterogeneous nature of the leachate organics and the actual complex kinetic and diffusion interactions. Additionally, the use of a non-degradable COD fraction of 20% in the untreated leachate, though useful for assessing the dynamics of the bioprocess, is of limited usefulness for accurate prediction of effluent COD. The modeling results are more useful for comparing performance under steady state and shock loading conditions and to assess the causative factors contributing to reactor response, rather than as precise predictions of biofilter performance.

The non-steady state simulations for Transients 1 and 2 are shown in Figs. 3 and 4, respectively. In both transients, dissolved oxygen was predicted to penetrate only a portion of the biofilm, and aerobic substrate utilization was stopped in the deeper regions where oxygen was depleted. Oxygen penetration was more limited in Transient 2, which had a higher organic loading. Higher effluent degradable COD was predicted in Transient 2 than Transient 1, but for both transients the experimental estimated effluent biodegradable COD was much higher than model predictions (Table 6). The flux of electron donor at steady state and in the transients was

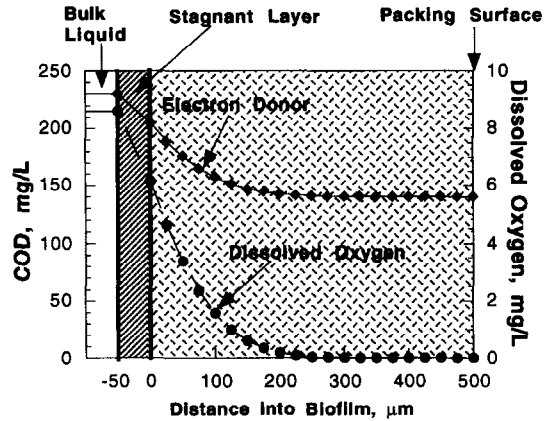


Fig. 4. Transient 2 electron donor and oxygen concentration profiles.

Table 7
Experimental and modeled electron donor and oxygen flux

	Flux, g/m ² /d		
	Steady state	Transient I	Transient II
Donor, experimental	6.43	10.3	7.70
Donor, model	6.67	19.7	25.2
Oxygen, model	5.10	8.43	10.0

calculated as the experimentally measured volumetric COD utilization rate divided by the specific surface area of packing media (Table 7). The flux of electron donor and oxygen predicted by the CSTR biofilm model are also shown in Table 7. Though the modeled oxygen flux increased and was double the steady state flux in Transient 2, modeled electron donor flux increased to almost four times that at steady state. The experimental donor flux also increased in the transients, but followed the modeled oxygen flux more closely than the donor flux predictions.

The CSTR model predicted a substantial increase in donor flux during the transients as compared to steady state, even though oxygen depletion shut down aerobic substrate utilization in substantial portions of the biofilm. This is explained by the higher levels of substrate to which organisms in the aerobic portions of the biofilm are exposed (Figs. 4 and 5). Though fewer biofilm organisms participate in aerobic substrate utilization in the transients, those organisms in the aerobic zone experience higher substrate levels and have higher specific substrate utilization rates (Table 2, Eq. (1)). The result is an increase in overall donor utilization in the transients over that at steady state.

The application of steady state biofilm thickness to transient modeling makes the assumption that the filter biomass would not adjust to the new, higher organic loadings in the limited time duration of the transients. The transient substrate flux, estimated from measured influent and effluent COD concentrations, was used to

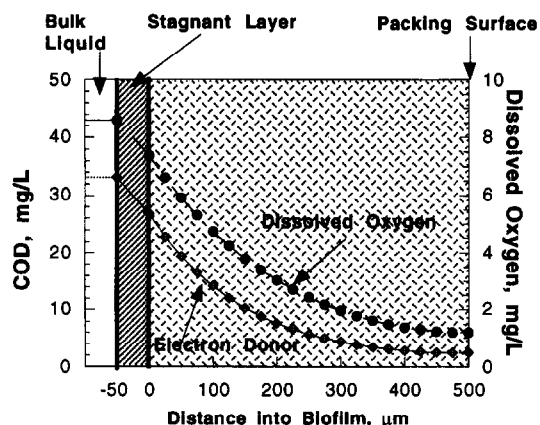


Fig. 5. Transient 3 electron donor and oxygen concentration profiles.

calculate the increase in biofilm thickness during the transients by applying the equation: $\Delta L = (JY - bX_f L)\Delta t/X$. The calculation indicates that biofilm thickness would increase significantly by 124 and 12 μm during Transients 1 and 2, respectively. However, the substrate profiles indicate that much of the deeper portion of the biofilm is non-aerobic during Transients 1 and 2, and substrate utilization occurs in only the outer biofilm layer. Even though the predicted increase in biofilm is significant, its effect on predicted substrate flux during transient operation is limited.

Several factors may help explain why measured increases in donor flux were less than model predictions. The landfill leachate treated in this study was a complex, multicomponent mixture of hazardous and non-hazardous organics. The CSTR biofilm model considers only a single electron donor substrate and cannot account for the complexities of a multicomponent mixture. Biochemical reaction rates can differ markedly for different chemicals, which makes difficult the accurate prediction of biofilter performance under transient conditions. Biofilms grown on multicomponent substrates would not be expected to be homogeneous with depth; more readily utilized substrates are perhaps preferentially utilized by specific microbial species which reside in layers closer to the bulk liquid. The model also uses a single diffusion coefficient to represent multiple diffusion electron donors; the limitations of this modeling simplification may be more apparent under transient than under steady state operation. Additionally, hazardous organics in the leachate could inhibit microbial substrate utilization by decreasing the specific utilization rates of substrates during applied transients. Soluble microbial product formation has been found to contribute to effluent COD in suspended and fixed film reactors, although the fraction of COD utilized in SMP formation is relatively small. SMP was not included in the model simulations because of the already large non-degradable COD component, and because inclusion of this added complexity would not substantially effect the implications of the modeling results. Treatment models have a limited ability to predict non-steady state performance when these complicating factors are present. The experimental results and model simulations cannot adequately assess these added complexities.

When considering the impact of hydraulic surges on fixed-film biotreatment of landfill leachates, another factor to consider is dilution of leachate by rainwater. Incident precipitation can result in overland flow or other water parcels that have no direct contact with waste materials and do not become contaminated. Following high rainfall events, the elevated leachate flow rates can be relatively dilute. The effect of dilution is decreased stress on the biofilter because the organic loading would not increase in proportion to the hydraulic flow rate. Two model simulations were performed to examine the effect of leachate dilution on the response of a steady state biofilter to applied hydraulic surges. Instantaneous flow rate increases of five- and tenfold over steady state were balanced by decreasing the influent COD to maintain a constant organic loading (Table 8). Simulated chemical profile and bulk substrate concentrations are shown in Figs. 5 and 6 for Transients 3 and 4, respectively. For

Table 8
Predicted response of aerobic biofilter to hydraulic surges at constant organic loading

Parameter, units	Steady state	Flowrate Increase	
		Fivefold ^a	Tenfold ^b
EBCT, d	1.3	0.26	0.13
Influent COD, mg/l	2900	580	290
Organic loading rate, kg COD/m ³ /d	2.23	2.23	2.23
<i>Model predictions</i>			
Effluent COD, mg/l	35	33	31
Donor flux, g/m ² /d	6.67	6.4	6.0
Oxygen flux, g/m ² /d	5.10	5.0	4.9

^a Modeled as Transient III.

^b Modeled as Transient IV.

^c Degradable COD.

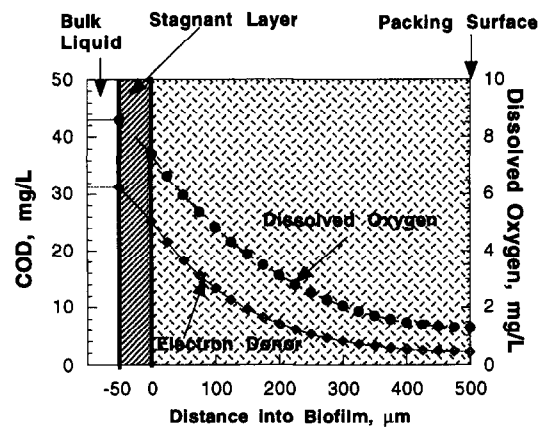


Fig. 6. Transient 4 electron donor and oxygen concentration profiles.

both transients, simulated profiles of oxygen, electron donor, and effluent COD were similar to steady state predictions. Sudden increases in flow rate appear less likely than organic shock loadings to result in a deterioration of effluent quality. Design of integrated equalization/aerobic biofilter systems for landfill leachate treatment should account for the concentration of bulk organics in the leachate during shock loadings as well as the magnitude of the flow rate increase.

5. Conclusions

The major conclusions from this study are listed below.

1. A steady state bench scale aerobic biofilter operated at a COD loading of 2.65 kg/m³/d removed 76% COD and 95% degradable COD from an industrial hazardous waste landfill leachate.
2. The aerobic biofilter responded poorly to organic shock loading, with transient COD removal efficiencies decreasing to as low as 18% at 14.3 kg/m³/day organic loading.
3. The increase in effluent COD during shock loading was partially composed of degradable but non-degraded organics.
5. Aerobic biofilter treatment substantially reduced the concentrations of several RCRA-listed organic chemicals under steady state conditions, but best demonstrated available technology standards were not met for all compounds.
6. Model and experimental results implicate oxygen diffusion as a key factor limiting aerobic activity in the biofilter during applied transients.
7. Model results suggest that effluent COD would not increase during shock hydraulic loading provided the organic loading rate remained constant.
8. The design of integrated equalization/biotreatment systems for landfill leachates should prevent organic shock loadings to aerobic biofilm reactors as well as limiting the hydraulic surges.

Acknowledgements

Dr. Daniel P. Smith is presently Professor in the Department of Civil and Environmental Engineering at Utah State University, Logan, Utah. At the time of this study, Dr. Smith was Senior Environmental Engineer with Environmental Process Design, Cranbury, New Jersey.

The experimental reactor studies were performed by Mr. Miceal Campbell.

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