Numerical Evaluation of Landfill Stabilization by Leachate Circulation

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Abstract: This study compares various leachate management scenarios using a biologically reactive transport model, which is proposed in this study. The proposed model can be used to predict the contribution of biodegradation to contaminant attenuation and contaminant concentration in leachate over time. It can also be used to assess the extent of landfill stabilization in terms of local mass per bulk volume of remaining refuse available for transfer. A sensitivity analysis shows that landfill stabilization has significant sensitivity to most biokinetic parameters, the fluid-phase saturation constant, and the dissolution rate, in addition to the half-saturation constant and the retardation factor. The proposed model is applied to assess landfill stabilization under two control scenarios: leachate recycling versus continued input of clean water with no recirculation. The simulation results indicate that leachate recirculation provides more favorable conditions for development of an active anaerobic bacterial population and, hence, accelerates landfill stabilization.

INTRODUCTION

A landfill is considered stabilized when two criteria are met: (1) maximum settlement has occurred; and (2) leachate does not constitute a pollution hazard (Leckie et al. 1979). In practice, it is desirable to quantify the rate of stabilization and, if possible, predict the time required for landfill site management. The stabilization of refuse and the change in quality of leachate depend on complex biological processes. This is especially the case in a sanitary landfill where the most important process is the biological decomposition of refuse materials. Modeling the biodegradation of organic contaminants in a sanitary landfill is a very useful tool for estimating the contribution of biodegradation processes to leachate attenuation and assessing the extent of landfill stabilization.

Carnes (1977) stated that the ratio of organic carbon to total carbon reflects the degree of biological stabilization of the landfill. A high value of this ratio indicates little organic degradation, whereas a low ratio reflects increasing stabilization. Other ratios such as chemical oxygen demands over total organic carbon (TOC), biochemical oxygen demand over TOC, and free volatile fatty acid over TOC have also been used. The present study, however, assumes that the degree of stabilization of the landfill can be measured by the remaining fraction of leachable solid waste as computed by the proposed model. The higher the remaining fraction of leachable solid waste, the lower the degree of stabilization.

Accelerated stabilization offers potential savings from lower postclosure costs and increases the potential for landfill reclamation. These advantages, and the opportunities for leachate management, have spurred many investigations using laboratory-scale experiments, landfill lysimeters, controlled landfill cells, and full-scale landfills (Pohland 1975; Leckie et al. 1979; Bogner 1990; Townsend et al. 1996). Some projects regarding leachate recirculation have focused on maximization of waste decomposition and/or the undesirable effects of forced leachate recirculation (Reinhart 1996; Reinhart and Al-Yousfi 1996). However, design and control methods for landfill stabilization are limited by lack of quantitative understanding of the interactions between microbial reactions and subsurface flow and transport. One way to develop this knowledge is to analyze the problem using numerical models that can incorporate many complex factors that are difficult to consider in simplified analytical models or laboratory tests.

This study compares various leachate management scenarios using a proposed biologically reactive transport model. Temporal changes in leachate quality and the remaining solid waste fraction are computed and compared for each scenario. This study is an extension of the work of Suk et al. (2000) and the numerical simulation model in this study modifies the previous model by incorporating a term associated with the waste leaching into the liquid phase.

GOVERNING EQUATION AND NUMERICAL SOLUTION PROCEDURE

The biologically reactive multispecies model is composed of two parts: variably saturated flow equations and a multispecies transport model. By solving the variably saturated flow equations, the velocity field and saturation for each phase can be computed. The velocity fields are then used to solve the multispecies transport problem. Finally, concentrations of multiple components in the liquid and gas phases are computed. Equations for variably saturated water flow and biologically reactive transport are solved numerically to calculate the concentrations of dissolved aqueous species (including oxygen), and aerobic and anaerobic bacteria. This study applies the formulation and numerical solution method of a preceding study (Suk et al. 2000).

Fluid Flow Equation

The governing equation for fluid flow in variably saturated porous media is given by Huyarkorn and Pinder (1983) as

$$\frac{\partial}{\partial z}\left(K_{zz}K_{rw}\left(\frac{\partial h}{\partial z}-1\right)\right)=C'\,\frac{\partial h}{\partial t} \tag{1}$$

where K_{zz} = saturated hydraulic conductivity in the z-direction (L T^{-1}); K_{rw} = relative permeability (dimensionless); h = pressure head (L); C' = specific moisture capacity (L^{-1}) defined by $C' = \phi(\partial S_w/\partial h)$ in which S_w = volumetric water content; ϕ = porosity; t = time (T); and z = soil depth, assumed to increase in a downward direction (L). van Genuchten (1980) gave the following equation to determine S_w from h:

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$$S_e = \frac{S_w - S_r}{S_s - S_r} = \left[\frac{1}{1 + (\alpha h)^n}\right]^m \quad (2)$$

where S_e = effective water saturation (dimensionless); S_r = irreducible water saturation (dimensionless); S_s = fully saturated volumetric saturation; α (L^{-1}) and n (dimensionless) = porous media parameters; and $m = 1 - 1/n$. The relative permeability K_{rw} is estimated from S_e (van Genuchten 1978). The specific moisture capacity is estimated from (2).

Contaminant Leaching

Contaminants are released from the refuse to passing leachate by physical, chemical, and microbial processes. The leachate percolates through the unsaturated environment, polluting the ground water with organic and inorganic matter. The model of leachate generation hinges on an understanding of the mechanisms of mass release from the solid to the liquid phase. In the present study the rate of mass transfer from the solid to the liquid phase, R_1, is postulated by Straub and Lynch (1982a,b) and Demetracopoulos et al. (1986) as

$$R_1 = \phi S_w k' \frac{S}{S_0}(c_{ST} - c_w) \quad (3)$$

where k' = dissolution rate (T^{-1}); S = local mass per bulk volume of refuse available for transfer at time ($M\ L^{-3}$); S_0 = same as S at initial time ($M\ L^{-3}$); c_{ST} = fluid-phase saturation constant ($M\ L^{-3}$); and C_w = concentration of dissolved aqueous species.

Linking Transport and Biological Reactions

The governing equations that account for the combined effects of solid-to-liquid mass transfer and the biokinetics in the aqueous phase as the source/sink term are

$$\phi S_w R \frac{\partial c_w}{\partial t} = \frac{\partial}{\partial z}\left(\phi S_w D \frac{\partial c_w}{\partial z}\right) - v\frac{\partial c_w}{\partial z} - M_a b_a \phi S_w \frac{c_w}{(K_a + c_w)}\frac{o}{(K_o + o)} - M_{an} b_{an}\phi S_w \frac{c_w}{(K_{an} + c_w)}\left(1 - \frac{o}{K_o + o)}\right) + k'\phi S_w \frac{S}{S_0}(c_{ST} - c_w) \quad (4)$$

$$\sigma S_w \frac{\partial o}{\partial t} = \frac{\partial}{\partial z}\left(\sigma S_w D \frac{\partial o}{\partial z}\right) - v\frac{\partial o}{\partial z} + q'_{wv}(o^* - o) - \phi S_w M_a b_a F \frac{c_w}{(K_a + c_w)}\frac{o}{K_o + o)} \quad (5)$$

$$\frac{\partial(\phi S_w M_a)}{\partial t} = \phi S_w M_a b_a Y_a \frac{c_w}{(K_a + c_w)}\frac{o}{(K_o + o)} - \phi S_w \lambda_a M_a \quad (6)$$

$$\frac{\partial(\phi S_w M_{an})}{\partial t} = \phi S_w M_{an} b_{an} Y_{an}\frac{c_w}{(K_{an} + c_w)}\left(1 - \frac{o}{(K_o + o)}\right) - \phi S_w \lambda_{an} M_{an} \quad \text{when } \frac{o}{(K_o + o)} < 0.01 \quad (7)$$

$$\frac{\partial S}{\partial t} = -k'\phi S_w \frac{S}{S_0}(c_{ST} - c_w) \quad (8)$$

where the parameters and variables are defined in the Notations section at the end of the paper.

The two-step Crank-Nicolson finite-difference approximation is used for solving the water flow equation [(1)] in variably saturated porous media. The detailed solution procedure can be found in Suk et al. (2000).

Various types of numerical models have been suggested (Molz et al. 1986; Widdowson et al. 1988; Kindred and Celia 1989; Taylor and Jaffé 1990, 1991; Zysset et al. 1994) to solve the transport equations. However, the numerical solution procedures used are typically developed to solve a fixed form of microbial kinetics with a fixed number of reacting species, thereby limiting the user's ability to adapt the model to other microbial systems. In the present study, a combination of a finite-difference method and reaction-operator-split approaches (Kinzelbach et al. 1991; Valocchi and Malmstead 1992; Wheeler et al. 1992) is used to solve the reactive transport equations [(4) and (5)]. The advective-dispersion portion of transport is solved by the finite-difference method, and the reactive portion is reduced to an ordinary differential equation. Eqs. (4)–(8) are reduced to a system of ordinary differential equations and are solved simultaneously at every node by an explicit method. In the present study, the fourth-order Runge-Kutta method is used. The solution procedure is explained in more detail in Suk et al. (2000).

MODEL COMPARISON AND SENSITIVITY ANALYSIS

The numerical results of the proposed model are compared with analytical solutions and other numerical models such as HYDRUS (Kool and van Genuchten 1992) and BIOF&T (1996). HYDRUS is a Galerkin linear finite-element program for simulation of transient 1D flow and solute transport in variably saturated porous media. BIOF&T describes flow of water and multicomponent aqueous phase transport in variably saturated porous media. HYDRUS cannot simulate biodegradation of contaminant. BIOF&T does not consider mass transfer of soluble materials from refuse to percolating water, or compute local mass per bulk volume of refuse available for transfer. Therefore, BIOF&T and HYDRUS cannot be used to assess the degree of landfill stabilization in terms of reduction of leachable solid waste fraction.

For the first case, 1D nonreactive solute transport is considered here. It is assumed that the contaminant transport occurs along a 3-m-long 1D column, with no biodegradation. The upper boundary condition is specified at constant concentration, and the lower boundary has no concentration gradient. Input parameters are listed in Table 1. The analytical solution for the problem is

$$C(z, t) = \frac{C_0}{2}\left[\operatorname{erfc}\left(\frac{z - ut}{\sqrt{4Dt}}\right) + \operatorname{erfc}\left(\frac{z + ut}{\sqrt{4Dt}}\right)\exp\left(\frac{uz}{D}\right)\right] \quad (9)$$

where C_0 = specified constant concentration of contaminant at inflow boundary; and u = seepage velocity. Fig. 1 shows the results of the proposed numerical model and BIOF&T solution compared to the analytical solution generated by (9).

For the second case, solute transport under the condition of biodegradation is considered here. The input parameters are listed in Tables 1 and 2. The results are shown in Fig. 2. Numerical results match the BIOF&T solutions for upper and lower zones of flow domain, whereas for middle zones the proposed numerical solutions vary slightly from BIOF&T solutions. It also seems that the small discrepancies between the proposed numerical results and BIOF&T solutions stem from numerical dispersion, numerical oscillation, or discrepancy errors, as noted before. More details on model comparison and calibration to field data can be found in Suk et al. (2000).

TABLE 1. Input Parameters for Numerical Simulation of Nonreactive Transport

Parameter	Value
Initial contaminant concentration	0 mg/L
Grid length	5 m
Darcy velocity	0.0039 m/day
Porosity	0.26
Longitudinal dispersivity	50 m
Specified constant concentration for inflow boundary	4.5 mg/L
Simulation time	6,000 days

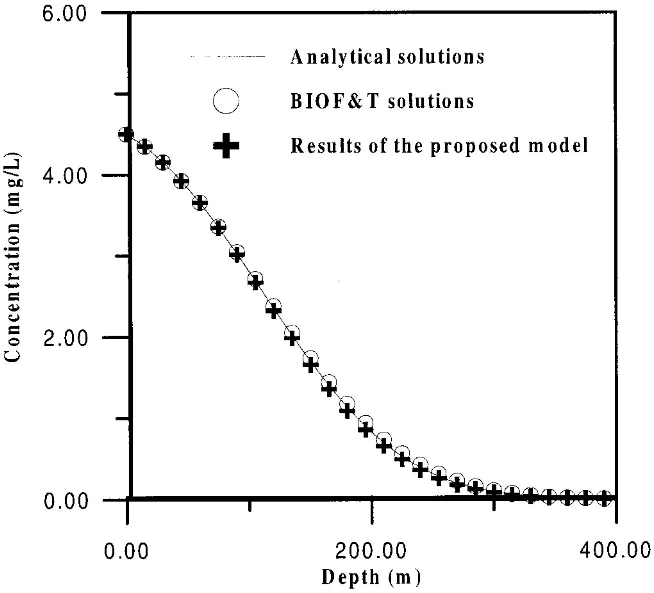

FIG. 1. Comparison of Solutions of Proposed Model with Analytical and BIOF&T Solutions in Nonreactive Transport Problem at 6,000 days (Specified Constant Concentration at Inflow Boundary = 4.5 mg/L)

TABLE 2. Additional Input Parameters for Numerical Simulation of Biologically Reactive Transport

Parameters	Value
Initial dissolved oxygen concentration	3 mg/L
Initial biomass concentration	0.1 mg/L
Specified constant concentration of dissolved oxygen for inflow boundary condition	3 mg/L
Maximum contaminant utilization rate per unit mass of microorganism	1.7 day^{-1}
Ratio of dissolved oxygen to contaminant consumed	3
Contaminant half-saturation constants	0.13 mg/L
Dissolved oxgyen half-saturation constant.	0.13 mg/L

The sensitivity analysis was performed for a hypothetical landfill where flow and solute transport conditions are shown in Fig. 3. Biological parameters are also shown in Fig. 3. In the sensitivity analysis discussed herein, only anaerobic utilization of the contaminant is emphasized because the aerobic phase is generally very short and negligible.

The sensitivity of landfill stabilization to dissolution rate was also analyzed. The time histories of the total leaching rate in refuse, bottom contaminant concentration, bottom biomass concentration, and total remaining solid waste fraction in refuse are shown in Fig. 4 for variable dissolution rates. An increase of dissolution rate leads to an increase of the maximum bottom contaminant concentration [Fig. 4(b)]. However, due to the finite amount of leachable mass, the bottom contaminant concentration corresponding to the larger dissolution rate decreases faster after maximum bottom contaminant concentration, so that the overall mass balance may be preserved. Fig. 4(b) shows that the history of the calculated bottom contaminant concentration resembles a hydrograph with a rising limb, a peak, a recession limb, and a near horizontal segment.

The rising limb (segment $\overline{AB}$) is the period for which the rate of mass transfer from the solid to liquid is larger than the rate of contaminant utilization by biomass because of low initial biomass concentration [Fig. 4(c)]. However, as the contaminant concentration increases [Fig. 4(b)], the rate of the mass transfer gradually decreases because the value of the gradient driving the mass transfer process in (4), $c_{ST} - c_w$, decreases. On the other hand, the rate of contaminant utilization by biomass increases because contaminant concentration actively increases [Fig. 4(b)]. A peak point (B) appears in Fig.

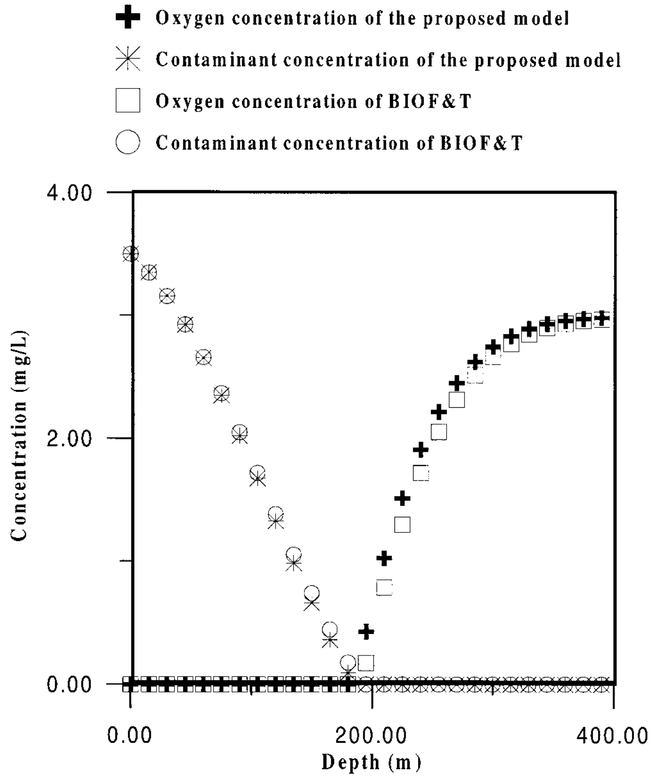

FIG. 2. Comparison of Solutions of Proposed Model with BIOF&T Solutions in Biologically Reactive Transport Problem at 6,000 Days (Specified Constant Concentrations of Contaminant and Dissolved Oxygen at Inflow Boundary = 4.5 and 3 mg/L, Respectively)

4(b), at which time the rate of the mass transfer is finally equal to the contaminant utilization rate. The recession limb (segment $\overline{BC}$) in Fig. 4(b) is the period for which the rate of mass transfer is smaller than the contaminant utilization rate because of excess increase of biomass [Fig. 4(c)]. Thus, for this period, the bottom contaminant concentration decreases [Fig. 4(b)].

The minimum point (C) in Fig. 4(b) is the point at which the biomass production rate by Monod kinetics is equal to the anaerobic microbial decay rate [Fig. 4(c)]. At this point, the biomass concentration is a maximum [Fig. 4(c)], and so the contaminant is consumed by maximum biomass utilization rate, thus the contaminant concentration reaches a local minimum [Fig. 4(b)]. Total leaching rates in refuse have a local peak at point C [Fig. 4(a)], where the value of the gradient driving the mass transfer process, $c_{ST} - c_w$, is a local maximum due to a local minimum of contaminant concentration [Fig. 4(b)]. The horizontal segment ($\overline{CD}$) in Fig. 4(b) is the period for which the rate of biomass production by Monod kinetics is smaller than the anaerobic microbial decay rate [Fig. 4(c)]. The decrease of bottom biomass concentration in segment ($\overline{CD}$) in Fig. 4(c) results from depletion of contaminant used as the electron donor in microbial metabolism. However, the value of the gradient driving the mass transfer process, $c_{ST} - c_w$, in (4) is actually large because of low contaminant concentration in segment ($\overline{CD}$) [Fig. 4(b)], and so the amount of soluble materials leached by mass transfer from refuse to leachate is significant in this segment. Despite the significant release of soluble materials, the significant amount of contaminant released is almost entirely used up as an electron donor in microbial metabolism, and so bottom contaminant concentration no longer increases and finally nearly reaches equilibrium. However, despite biomass utilization of a significant amount of released contaminant, bottom biomass concentration decreases with time in segment ($\overline{CD}$) [Fig. 4(c)]. This can be explained by the fact that the bottom contaminant

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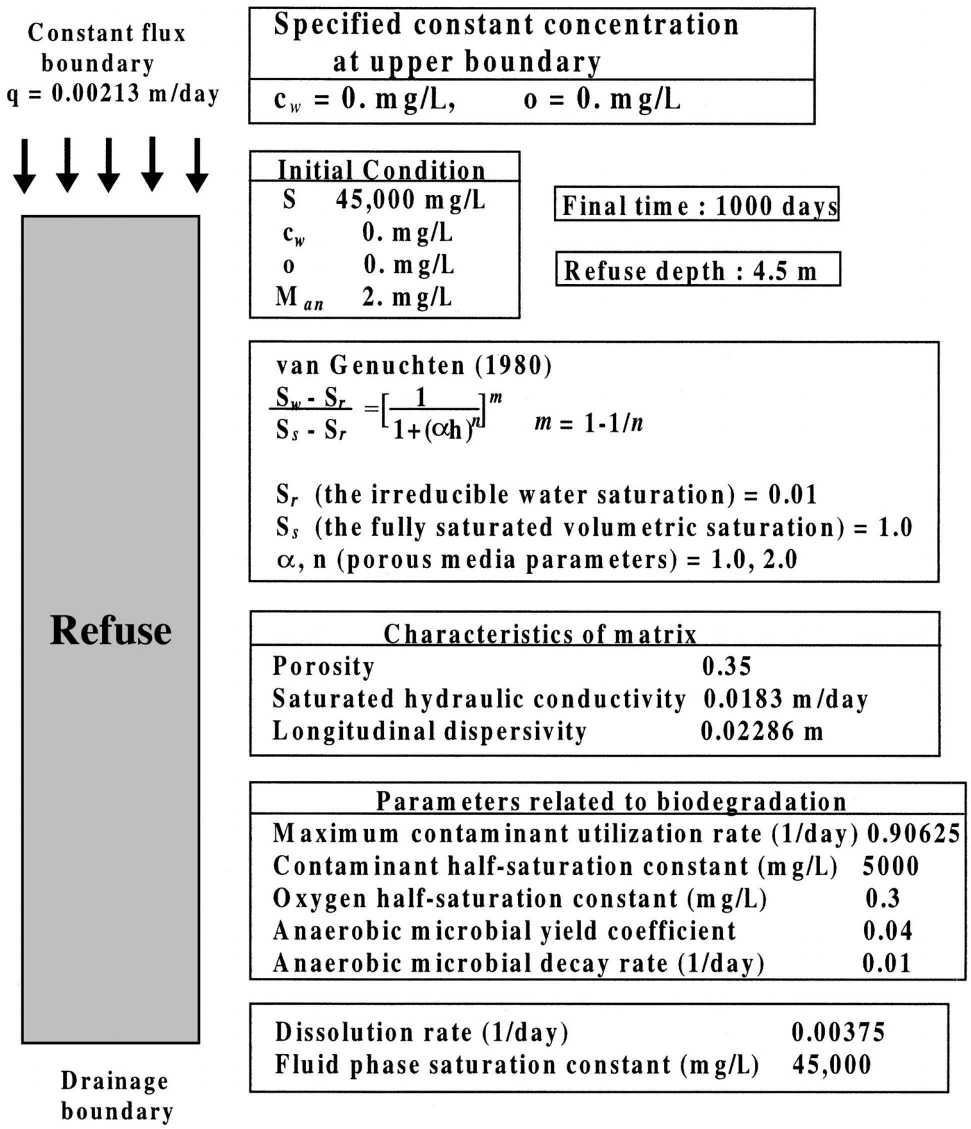

FIG. 3. Input Parameters and Conditions for Sensitivity Analysis

concentration is lower than $\lambda_{an}K_{an}/(b_{an}Y_{an} - \lambda_{an})$ in segment ($\overline{CD}$) [in this study $\lambda_{an}K_{an}/(b_{an}Y_{an} - \lambda_{an})$ is about 1,900 mg L^{-1}], and so the right side of (7) is lower than zero; thus biomass concentration decreases.

The extent of ultimate refuse stabilization of the landfill was assumed to be indicated by the total remaining leachable solid waste. In this simulation, 40% of total leachable solid waste has been removed from the landfill after approximately 600, 720, and >1,000 days for dissolution rates 0.00375, 0.003, and 0.002 day^{-1}, respectively [Fig. 4(d)]. This indicates that the higher the dissolution rate, the faster landfill stabilization is reached. Thus, it is found here that the dissolution rate has a significant effect on landfill stabilization.

Finally, the model shows significant sensitivity of landfill stabilization to most biokinetic parameters (yield coefficient, maximum utilization rate, and anaerobic microbial decay rate), the fluid-phase saturation constant, and the dissolution rate (also to half-saturation constant and retardation factor, although the latter results are not shown in this paper).

NUMERICAL RECIRCULATION EXPERIMENTS

Here, landfill scenarios were constructed that were simulated with leachate recirculation, continued input of clean water, and landfill control (no recirculation). Input parameters and a conceptual model for this application are shown in Fig. 5. The application of the proposed model can aid in deciding which scenario is more appropriate to landfill stabilization.

The traditional approach to a sanitary landfill design and operation is to encapsulate and store waste in a manner that minimizes entry of moisture into the landfill. In this approach, leachate production is minimized, but the rate of biological decomposition of the solid waste is slowed. In this study, this mode of operation is referred to as landfill control.

On the other hand, using a technique known as leachate recirculation, a landfill may be operated as a solid waste bioreactor treatment system rather than as a waste storage site. Thus, leachate recycling may be used to convert the landfill to a bioreactor treatment system and accelerate landfill stabilization (Pohland 1972, 1975; Leckie et al. 1975, 1979; Townsend et al. 1996).

The application of the numerical model to leachate recycling, continued input of clean water and landfill control demonstrates the effect of leachate recycling and continued input of clean water in accelerating landfill stabilization. The simulation for leachate recirculation assumed contaminant concentrations of input moisture to be equal, over time, to their respective concentrations in the flow from the bottom. Because leachate production is delayed until field capacity is reached, initial conditions are important in determining how quickly leachate is produced. As a result, assumptions of the entry of

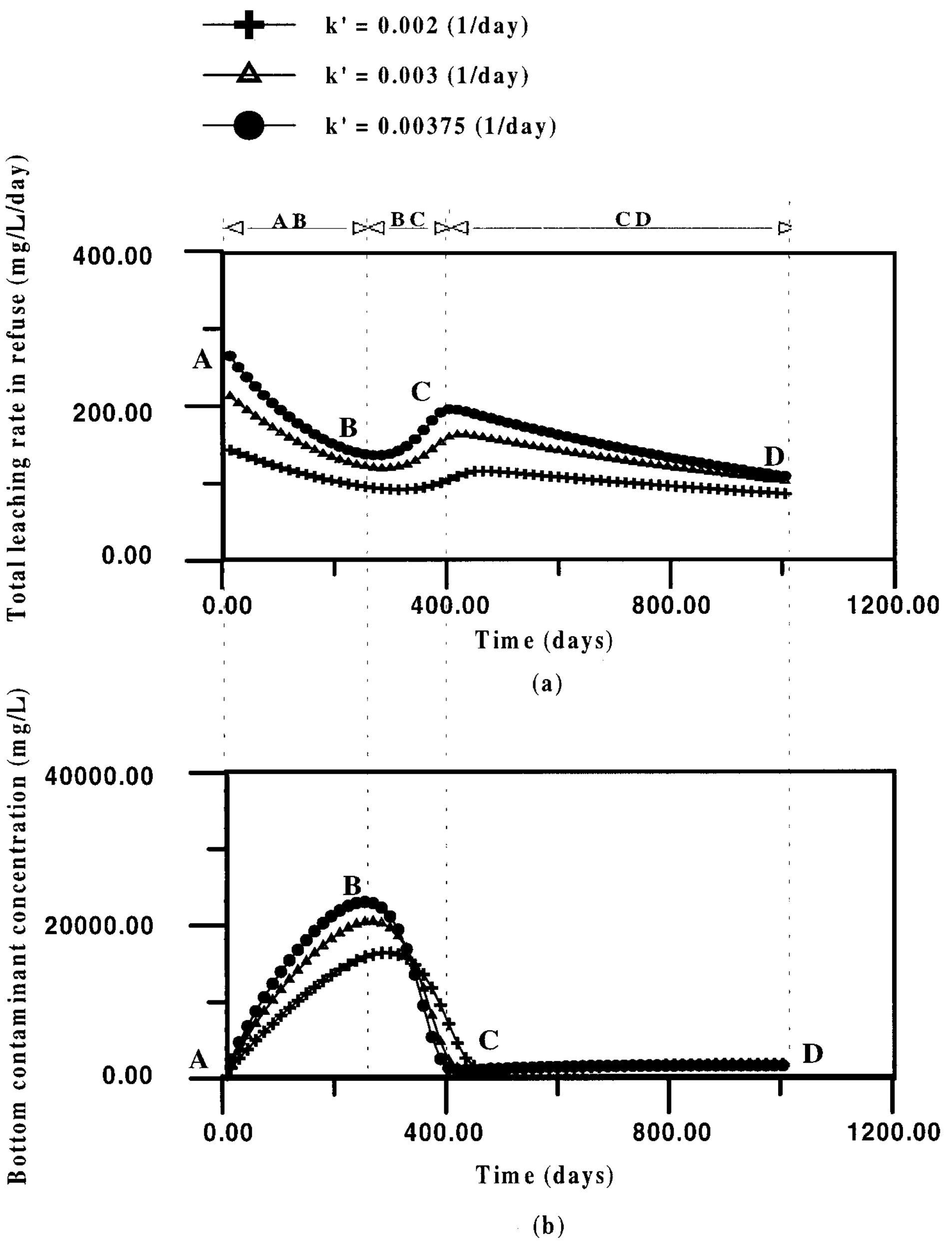

FIG. 4. Sensitivity Analysis of: (a) Total Leaching Rate in Refuse; (b) Bottom Contaminant Concentration; (c) Bottom Biomass Concentration; (d) Total Remaining Solid Waste Fraction in Refuse to Dissolution Rate

constant moisture into landfill (flow rate into top of landfill = flow rate out of bottom of landfill) and uniform initial conditions are necessary to avoid the effect of initial moisture conditions on the delay of leachate production. In this study, to compare effects of two modes of operation such as continual water application and leachate recirculation on landfill stabilization, it is assumed that the initial condition and rate of input water flow into landfill in continual water application are the same as in leachate recirculation. In practice, there is also leakage in the landfill control case, which this study assumes to occur at the rate of 2.13E-6 m/day. To investigate the effect of flow rate on landfill stabilization, both clean water and leachate in the two modes of operation mentioned above were applied at rates of 0.000526 m/day (low flow rate) and 0.00213 m/day (high flow rate).

The five modes of operation were marked with symbols as follows: A, B, B′, C, and C′ (Table 3). The A mode denotes landfill control with microbial activity. This operation provides a comparative standard basis for the other four managed operations and to investigate the effects of flow rate and microbial activity on landfill stabilization. The B and B′ modes denote continual through-flushing with clean water with microbial activity (B) and no microbial activity (B′), respectively. The C and C′ modes denote recirculation of leachate with microbial activity (C) and no microbial activity (C′), respectively.

In the B and C modes of operation under conditions of both relatively low and high flow rate [Figs. 6(a) and 7(a)], bottom contaminant concentrations in leachate increase rapidly and then decrease to the same low level. The abrupt decreases after the peak are accounted for by the relationship between substrate/microbial dynamics and the leaching process [Figs. 6(a) and 7(a)]. This behavior was mentioned above in the sensitivity analysis. In contrast, in the B′ mode of operation under conditions of low and high flow rates, the bottom contaminant concentrations continue to increase, and reach pseudoequilibrium within the limited time span considered [Figs. 6(a) and 7(a)]. In the B′ mode of operation, not considering microbial activity, no recession limb appears because there is no growth of biomass, and so bottom contaminant concentration contin-

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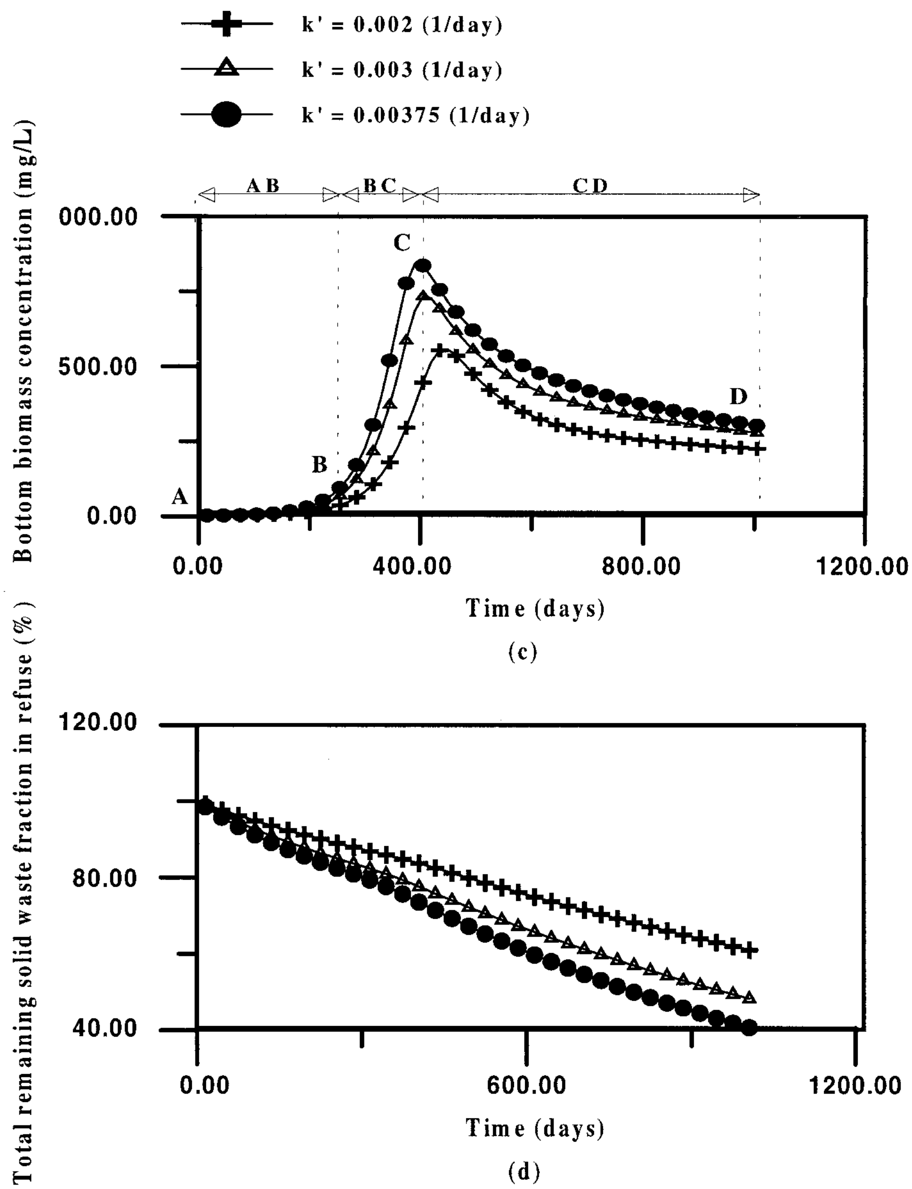

FIG. 4. (*Continued*)

ues to increase until simulation time approximately reaches travel time for passing through the refuse. After the travel time, because percolating water has the same contact time with refuse, the same mass is able to leach into percolating water. Hence, bottom contaminant concentrations reach pseudoequilibrium after the travel time in the B′ mode of operation. Especially, Figs. 6(a) and 7(a) indicate that in the B′ mode, the higher the flow rate, the lower the bottom contaminant concentration at pseudoequilibrium. This is because higher flow rate leads to less contact time, offering decreased opportunities for solubilization of leachable materials within refuse. In addition, Figs 6(a) and 7(a) indicate that while the bottom contaminant concentration in the B′ mode reaches a pseudoequilibrium value less than the fluid-phase saturation constant, in the C′ mode the bottom contaminant concentration attains an equilibrium value equal to the fluid-phase saturation constant. The reason for the difference is that the addition of contaminant by leachate circulated back to landfill surface in the C′ mode increases contaminant concentration more than in the B′ mode.

According to Figs. 6(b) and 7(b), the extent of stabilization is greater in the B′ mode than in C′. This is because the increase of contaminant concentration through leachate recirculation reduces the value of the gradient driving the mass transfer process, $c_{ST} - c_w$, in (4); the leaching rate is slowed; and thus the extent of landfill stabilization in the C′ mode is less than that in B′. Moreover, Figs. 6(b) and 7(b) show that in the B′ mode, the greater the flow rate the greater the extent of landfill stabilization. This can be explained almost completely on the basis of increasing the quantity of moisture. The higher flow rate in steady state results in greater moisture content within refuse. This allows extraction of more soluble materials within refuse because increasing moisture content increases the last term on the right side of (4), which describes the force driving mass transfer of soluble materials from solid to liquid phase. In the same manner, this explanation can also be applied to the C′ mode as well as the B and C modes.

In the cases of the B and C modes under conditions of both relatively low and high flow rate, the degree of landfill stabilization in C is nearly the same as in B [Figs. 6(b) and 7(b)]. These results suggest that because addition of contaminants through leachate recirculation promotes more vigorous microbial activity, growth of biomass is significantly more rapid. Thus, the contaminants added by recirculation as well as contaminants leached from refuse are consumed rapidly by the enhanced microbial utilization. In addition, this fact suggests

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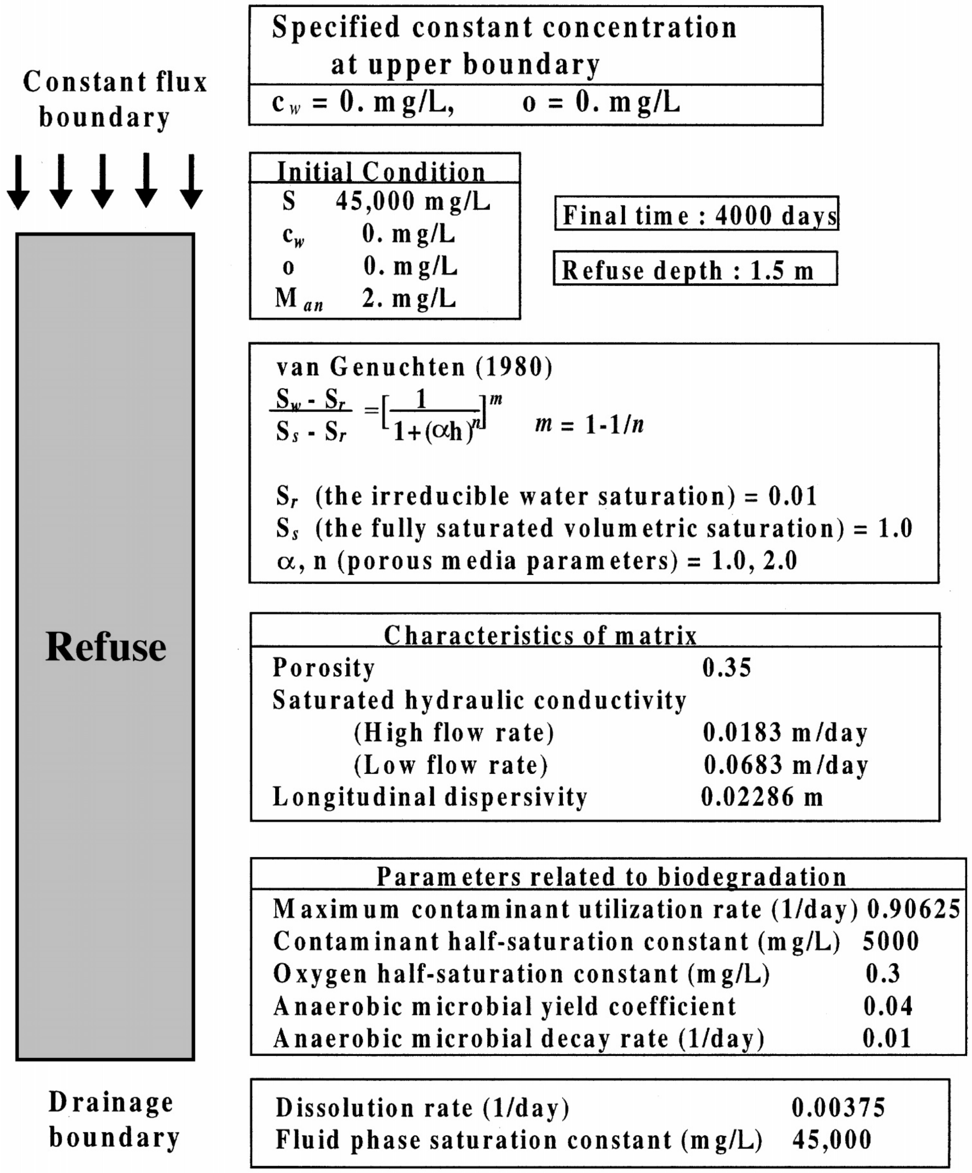

FIG. 5. Input Parameters and Conceptual Model for Application

TABLE 3. Five Modes of Stabilization Operation for Landfill

Mode	A	B	B′	C	C′
Landfill control	O	X	X	X	X
Microbial activity	O	O	X	O	X
Clean water flushing	X	O	O	X	X
Leachate recirculation	X	X	X	O	O

that if the C mode is used to stabilize the landfill, there is no need to expend additional cost for leachate treatment or to obtain water for the B mode. Hence, C has advantages over B for landfill stabilization in terms of leachate control.

In addition, it is found here that by comparison of the differences between the C and C′ modes of operation, and between B and B′ [Figs. 6(b) and 7(b)], the biological process has a more pronounced influence on landfill stabilization in the C mode than in B. Also, it is found here that by comparison of the differences between B′ and C′ (dilution effect) and between C′ and C (biodegradation effect), the effect on landfill stabilization of biodegradation in leachate recirculation is dominant over the effect of dilution [Figs. 6(b) and 7(b)].

CONCLUSIONS

This study used the model of Suk et al. (2000) to compare various leachate management scenarios by estimations of the landfill stabilization extents. The present study extended the original model by adding a transport numerical model that estimates the extent of landfill stabilization. The model incorporates water flow, contaminant solute transport, the leaching process of soluble materials from refuse to percolating water, adsorption, and aerobic and anaerobic biodegradation.

To define which parameters have the most effect on landfill stabilization, a sensitivity analysis was performed. The sensitivity analysis showed that the extent of landfill stabilization has significant sensitivity to most biokinetic parameters (yield coefficient, maximum utilization rate, and anaerobic microbial decay rate), the dissolution rate, and the fluid-phase saturation constant.

The application of the proposed numerical model to leachate recycling, continued input of clean water and landfill control demonstrated the influence of leachate recycling and continued input of clean water on landfill stabilization. It can be concluded through numerical experiments that leachate recircu-

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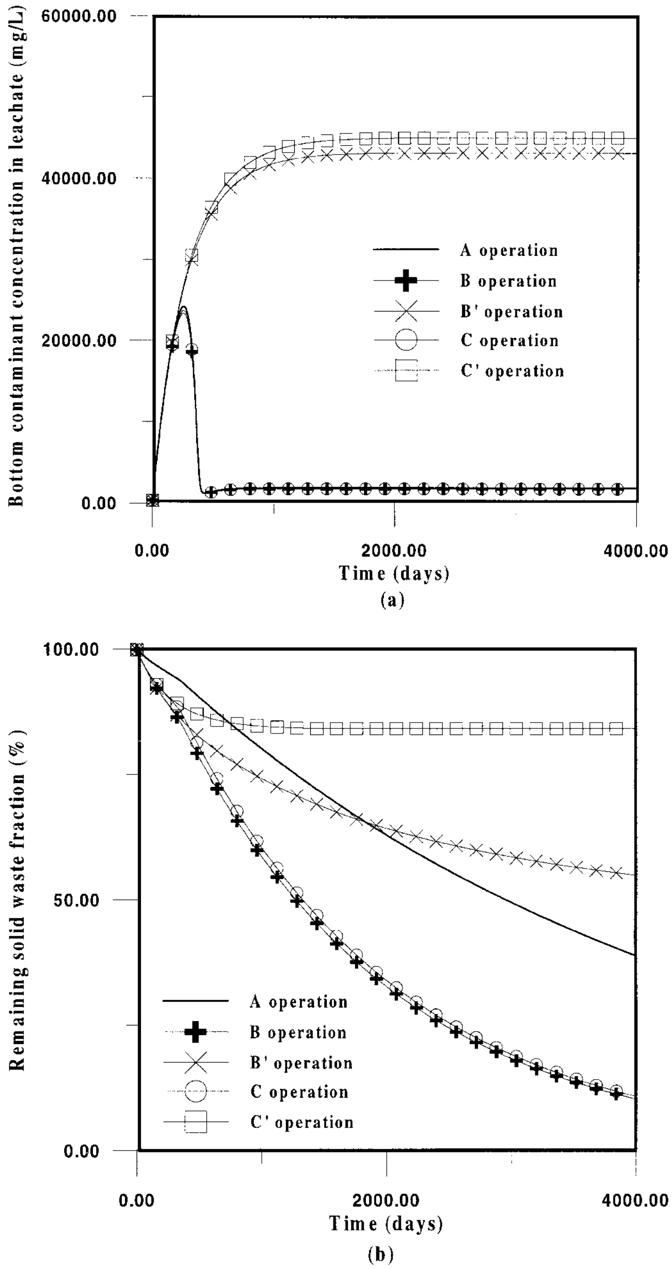

FIG. 6. History of: (a) Simulated Bottom Leachate Concentrations; (b) Simulated Total Remaining Leachable Solid Waste Fractions with Time for Leachate Recirculation and Continued Input of Pure Water with Relatively Low Flow Rate (5.26E-4 m/day) and Landfill Control with Leakage Rate (2.13E-6 m/day), Considering Microbial and No Microbial Activity

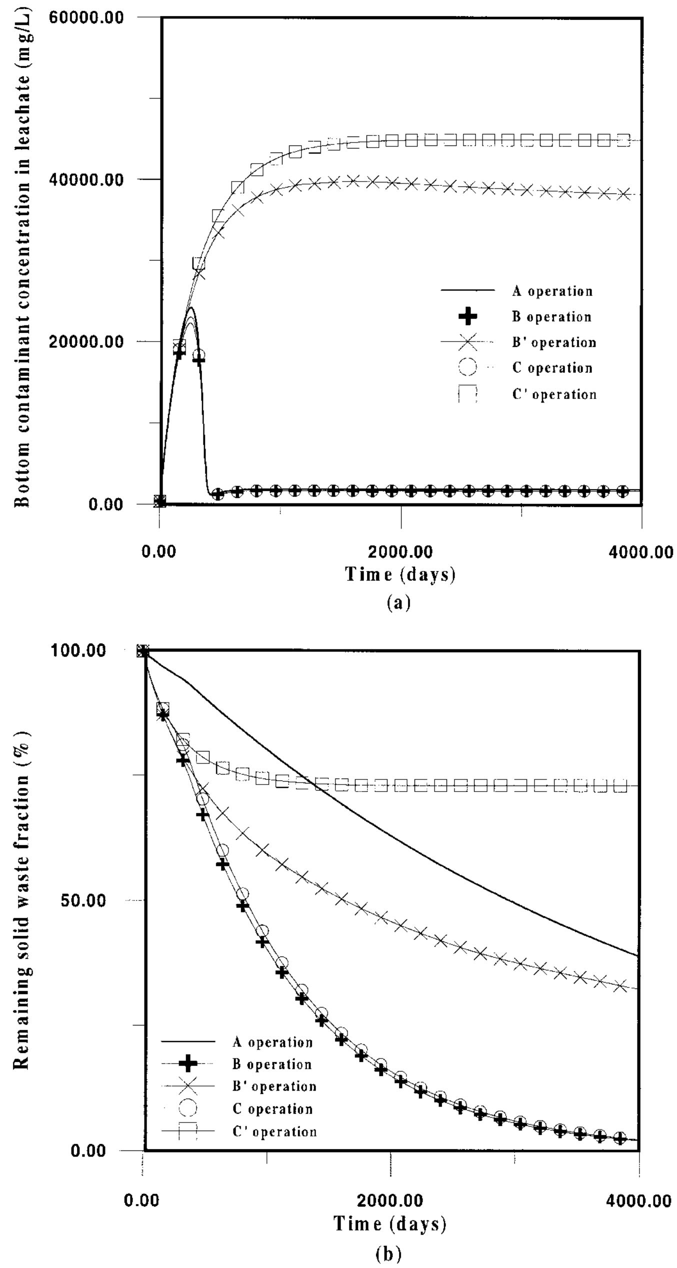

FIG. 7. History of: (a) Simulated Bottom Leachate Concentrations; (b) Simulated Total Remaining Leachable Solid Waste Fractions with Time for Leachate Recirculation and Continued Input of Pure Water with Relatively High Flow Rate (2.13E-3 m/day) and Landfill Control with Leakage Rate (2.13E-6 m/day), Considering Microbial and No Microbial Activity

lation provides more favorable conditions for development of an active anaerobic bacterial population, and since the increase of anaerobic bacteria enhances the rate of removal of contaminants, the leachate recirculation accelerates landfill stabilization.

Moreover, it is found here that in the leachate recirculation mode, the effect on landfill stabilization of biodegradation in leachate recirculation is dominant over the effect of dilution. These results suggest that the leachate recirculation performance uses the landfill as a bioreactor treatment system, and so the leachate recirculation accelerates landfill stabilization. This advantage of leachate recirculation performance has also been reported in experimental works (Pohland 1972, 1975; Leckie et al. 1975, 1979; Townsend et al. 1996). Finally, through numerical experiments it is found here that higher flow rates lead to faster landfill stabilization.

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NOTATION

The following symbols are used in this paper:

b_a = maximum contaminant utilization rate per unit mass of aerobic microorganisms (T^{-1});
b_{an} = maximum contaminant utilization rate per unit mass of anaerobic microorganisms (T^{-1});
C' = specific moisture capacity (L^{-1}) defined by $C' = \phi(\partial S_w/\partial h)$;
c_{ST} = fluid phase saturation constant ($M\ L^{-3}$);
c_w = concentration of dissolved aqueous species ($M\ L^{-3}$);
C_0 = specified constant concentration of contaminant at inflow boundary ($M\ L^{-3}$);
D = dispersion coefficient ($L^2\ T^{-1}$);
F = ratio of oxygen to contaminants consumed (dimensionless);
h = pressure head (L);
K_a = contaminant half-saturation constant for aerobic biodegradation ($M\ L^{-3}$);
K_{an} = contaminant half-saturation constant for anaerobic biodegradation ($M\ L^{-3}$);
K_d = solid-liquid phase partitioning coefficient ($M^{-1}\ L^3$);
K_o = oxygen half-saturation constant ($M\ L^{-3}$);
K_{rw} = relative permeability (dimensionless);
K_{zz} = saturated hydraulic conductivity in z-direction ($L\ T^{-1}$);
k' = dissolution rate (T^{-1});
M_a = aerobic microbial concentration ($M\ L^{-3}$);
M_{an} = anaerobic microbial concentration ($M\ L^{-3}$);
m = $1 - 1/n$;
n = porous media parameter (dimensionless);
o = dissolved oxygen concentration ($M\ L^{-3}$);
o^* = dissolved oxygen concentration in source/sink fluid ($M\ L^{-3}$);
q'_{wv} = volumetric flow rate of fluid injection (or withdrawal) per unit volume of porous medium (T^{-1});
R = retardation factor (dimensionless) defined by $R = 1 + K_d\rho_b/\phi S_w$;
R_1 = rate of mass transfer from solid to liquid phase ($M\ L^{-3}\ T^{-1}$);
S = local mass per bulk volume of refuse available for transfer at time ($M\ L^{-3}$);
S_e = effective water saturation (dimensionless);
S_r = irreducible water saturation (dimensionless);
S_s = fully saturated volumetric saturation (dimensionless);
S_w = volumetric water content (dimensionless);
S_0 = same as S at initial time ($M\ L^{-3}$);
t = time (T);
u = seepage velocity ($L\ T^{-1}$);
v = Darcy velocity ($L\ T^{-1}$);
Y_a = aerobic microbial yield coefficient (dimensionless);
Y_{an} = yield coefficient (ratio of anaerobic microorganisms produced to contaminants biodegraded) of anaerobic biomass (dimensionless);
z = soil depth, assumed to increase in downward direction (L);
α = porous media parameter (L^{-1});
λ_a = aerobic microbial decay rate (T^{-1});
λ_{an} = anaerobic microbial decay rate (T^{-1});
ρ_b = soil bulk density ($M\ L^{-3}$); and
ϕ = porosity (dimensionless).

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