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One of the most common types of onsite wastewater treatment and disposal systems in use today utilizes a septic tank followed by a soil absorption system (SAS). The SAS serves both to provide additional treatment and as a means to dispose of the applied wastewater. Physical/chemical treatment mechanisms in SASs include particulate filtration and sorption of select soluble wastewater constituents. The individual soil particles also provide a surface to which microorganisms can attach and grow, enabling further biological treatment of the wastewater to occur.

Biological oxidation processes can be broadly grouped into two categories: aerobic processes, in which the ultimate electron acceptor is elemental oxygen; and anaerobic processes, in which the ultimate electron acceptor is an organic or inorganic molecule other than oxygen. The results of past research suggest that when the unsaturated zone beneath a SAS is predominantly anaerobic, both the effluent quality and the infiltrative capacity of the system are adversely affected. For example, Magdoff et al. (1974) obtained essentially complete oxidation of the reduced carbonaceous and nitrogenous substrates present in wastewater applied to aerobic sand columns; in anaerobic columns, almost one-half of the reduced carbonaceous and nitrogenous substrates showed up in the column effluent. Furthermore, some complex organic compounds, including certain pesticides and aromatic hydrocarbons, are not appreciably degraded at all under anaerobic conditions (Sommers et al. 1980). An anaerobic unsaturated zone has also been linked to the accelerated development of so-called "clogging mats" on the infiltrative surfaces of SASs (Cotteral and Norris 1969). Recently, Siegrist (1986) proposed that the accumulation of organic matter by filtration and sorption followed by the synthesis of humic substances was the primary mechanism responsible for the soil clogging observed in his test cells. He then noted that restricted aeration within a SAS was one of the factors which stimulated this process.

The major objective of this research was to obtain a better understanding of gas transport associated with the biochemical transformations of pollutants occurring in the unsaturated zone of soil absorption systems. To meet this objective, research activity was conducted in two main areas: first, a two-dimensional mechanistic model of the transport of oxygen and other gases in SASs was developed; and second, bench-scale experiments were performed to collect quantitative data on these processes which could be used to verify the model. One potential application for a model such as this would be to estimate the composition of the soil-gas phase, and thus the potential for anaerobiosis to occur, within full-scale systems as a function of various system design parameters such as the physical characteristics of the underlying soil and the geometry of the SAS.

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GAS TRANSPORT MODEL

Fick's Law has often been used to model the diffusion of gases in porous media such as soils. However, Jaynes and Rogowski (1983) showed that multicomponent systems (i.e., systems in which three or more gases are present) are often poorly represented by Fickian-type models. Therefore, the so-called "dusty gas" model that was summarized in a paper by Thorstenson and Pollock (1989) was used to develop the multicomponent gas diffusion model used in this research along with the following assumptions:

- the soil atmosphere is predominantly aerobic and is comprised of a five-component mixture of oxygen, carbon dioxide, nitrogen, argon, and water vapor (at 100 percent relative humidity);
- nitrogen and argon are stagnant gases with no sources or sinks;
- for every one mole of oxygen consumed in the biochemical oxidation of carbonaceous organics, r moles of carbon dioxide are given off and leave the system via diffusion through the soil gas phase, resulting in the collinear, countercurrent diffusion of the two gases;
- the gas composition varies spatially within the unsaturated zone, but is time-invariant at any one point; and
- the system is isothermal and isobaric.

The governing equation derived from these assumptions was found to be

$$\nabla^2 x_i - \frac{k_{ai}\left(\nabla x_i\right)^2}{k_{ai}x_i + k_{bi}} + \frac{R_1}{cQD_{im}}\left(k_{ci}x_i + k_{di}\right) = 0 \qquad (1)$$

where x_i is the mole fraction of a component gas i (dimensionless)
R_1 is the production rate of oxygen per unit volume of the porous medium (mol/L^3/t)
c is the molar concentration of the gas mixture per unit void volume (mol/L^3)
D_{im} is the effective binary diffusivity of component gas i in a multicomponent gas mixture (L^2/t)
Q is the obstruction factor defined as the ratio of the binary diffusivity in a partially-saturated soil to the binary diffusivity in space (dimensionless)
$k_{a\text{-}d,i}$ are coefficients defined in Table 1

Equation (1) has no analytical solution. Instead, a finite difference approximation to the solution was obtained using Gauss-Seidel fixed-point iteration with successive overrelaxation. The solution methodology is presented elsewhere (Mahuta 1991) and will not be reproduced here.

MATERIALS AND METHODS

A layout drawing of the bench-scale SAS and its associated apparatus is shown in Fig. 1. The dimensions of the unit were 122 cm (48 in) high by 48 cm (19 in) wide with a horizontal depth of 7.6 cm (3 in). The unit was designed to represent one-half of the vertical cross-section of a SAS trench, assuming that typical soil-gas concentration profiles generated in a field system would be symmetrical about the centerline of the trench or bed. Therefore, the infiltrative surface width of 20 cm (8 in) in the SAS unit would corresponded to a SAS trench width of 40 cm (16 in).

Table 1. Coefficients of Governing Equation (1)

Component	i	k_{ai}	k_{bi}	k_{ci}	k_{di}	D_{im}
Oxygen	1	r-1	1	r-1	1	Eq. (2)
Carbon Dioxide	2	r-1	-r	r-1	-r	Eq. (3)
Nitrogen	3	1	0	$(D_{13}/D_{23})r - 1$	0	D_{13}
Argon	4	1	0	$(D_{14}/D_{24})r - 1$	0	D_{14}

$$D_{1m} = \frac{1 - x_1\left(1 - r\right)}{\frac{\left(x_2 + rx_1\right)}{D_{12}} + \frac{x_3}{D_{13}} + \frac{x_4}{D_{14}}} \tag{2}$$

$$D_{2m} = \frac{-r - x_2\left(1 - r\right)}{\frac{\left(-x_2 - rx_1\right)}{D_{12}} - \frac{rx_3}{D_{23}} - \frac{rx_4}{D_{24}}} \tag{3}$$

A medium-to-fine sand was used as the porous medium in these studies. The feed wastewater was primary effluent from a municipal wastewater treatment plant. It had an average total organic carbon (TOC) content of 80 mg/L, total Kjeldahl nitrogen (TKN) content of 20 mg/L, and total phosphorus content of 6.5 mg/L. The feed was pumped to the SAS unit four times per day for two minutes per pumping event at an overall rate of 4.1 cm/d (1 gpd/ft^2 of infiltrative surface area). The system was operated in a constant temperature room at 20°C. Wastewater was added fresh to the feed tanks each day since biodegradation of the stored wastewater was relatively rapid at this temperature.

After a series of experiments were conducted under these conditions, the wastewater was spiked with glucose, ammonia, and other trace nutrients, and a second series of experiments were conducted in order to obtain larger oxygen and carbon dioxide gradients within the unsaturated zone. Glucose was added at a concentration of 1,300 mg/L, and nitrogen and phosphorus were added to maintain the same C:N:P ratio as the non-spiked primary effluent. These two sets of data were then used to calibrate and verify the multicomponent gas transport model.

Wastewater samples were collected from the feed, effluent, and from selected points within the unsaturated zone during testing. Soil-water samples were collected by applying a vacuum to porous ceramic cups placed at various depths beneath the infiltrative surface and collecting the extracted pore water. The concentrations of ultimate biochemical oxygen demand (BOD), TOC, TKN, and of various anions (chloride, nitrite, nitrate, and sulfate) were measured in the wastewater feed and effluent. The anions were measured using high-pressure liquid chromatography (HPLC). Only the TOC and anion concentrations were measured in the soil-water samples because of the low sample volumes obtained (3-5 mL). Correlations between ultimate BOD and TOC were used to estimate oxygen uptake rates (OURs) in the unsaturated zone from the oxidation of carbonaceous organics; OURs from the oxidation of nitrogenous substrates were estimated from changes in the nitrite/nitrate concentrations in the soil-water. The overall OUR and respiratory quotient of the system were measured from a mass balance on the component gases in the air inlet and outlet. This could then be compared with the estimated OUR from substrate removal considerations to check the accuracy of the latter estimate.

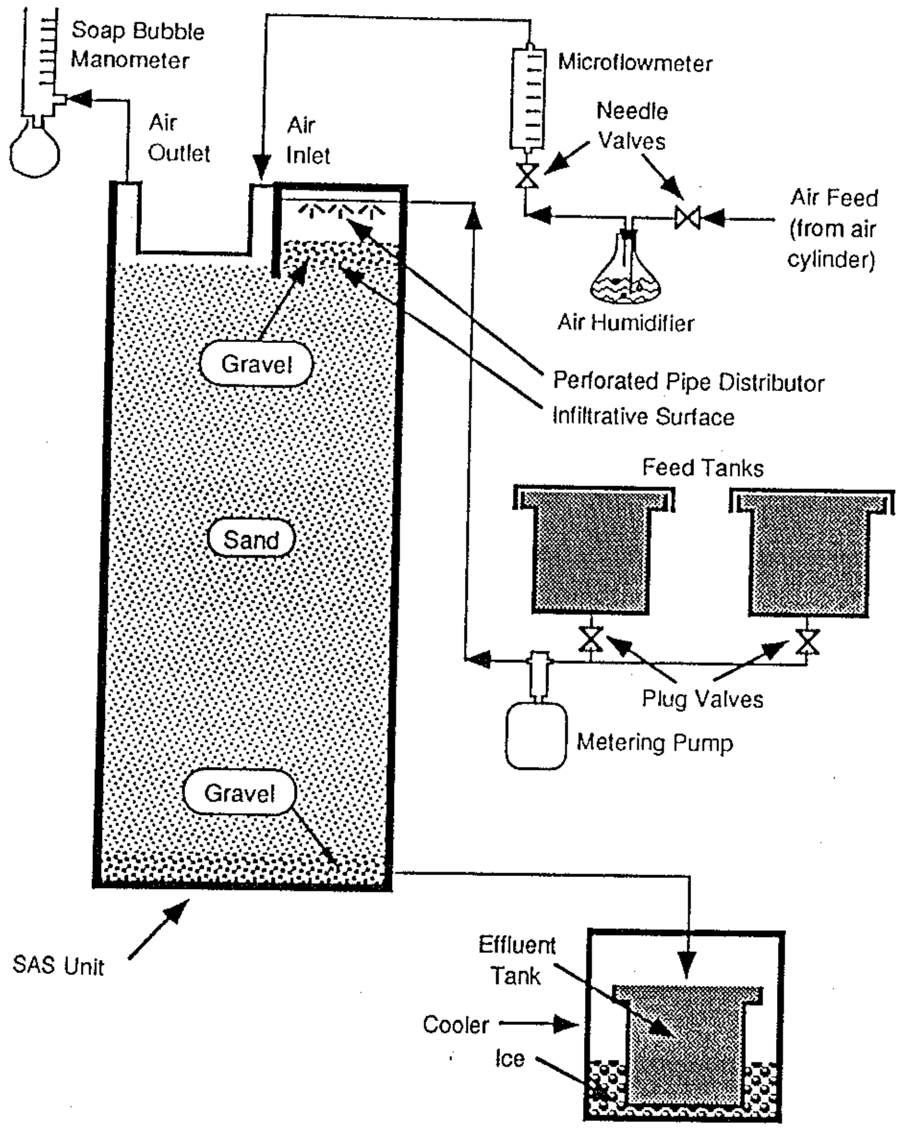


Fig. 1 Layout Drawing of the Bench-scale SAS and Associated Apparatus

Soil-gas samples were collected from the unsaturated zone in gas-tight syringes from sample points sealed with rubber septa. They were analyzed using a gas partitioner calibrated for O_2-Ar, CO_2, N_2, and CH_4.

Tensiometers were used to measure the matric suction of the soil-water in the unsaturated zone. A soil-moisture characteristic curve (SMCC) was developed to estimate the air-filled porosity of the medium from the measured values of matric suction at various points within the system. However, this method was rather imprecise because of the hysteresis exhibited by the SMCC.

RESULTS AND DISCUSSION

Experimental Work

The bench-scale SAS was operated using primary effluent as the feedstock for approximately 6-1/2 months until it "failed"; that is, the applied wastewater was continuously ponded between doses to a depth of 13 cm above the infiltrative surface. Ponding development was gradual. After about three months of operation, applied wastewater began ponding during a dosing cycle,

but would rapidly infiltrate within the first few minutes after the cycle ended. In contrast, system failure when spiked feed was used occurred within 2-3 weeks following system start-up. Following the first system failure using spiked feed, the SAS unit was emptied and refilled with clean, dry sand. Nevertheless, system failure again occurred less than a month after start-up. These observations lend some support to the conclusion of Siegrist (1986) that the rate of soil clogging in SASs is proportional to the total mass loading of organic material applied. However, the sand in the latter spiked feed run was packed to a higher bulk density than was the sand in the first spiked feed run, possibly contributing to the rapid failure as well.

Substrate Transformations: Using primary effluent as the feedstock, between 66 to 76 percent of the applied TOC was removed within the first 30 cm (1 ft) of unsaturated zone. These removal rates were close to the 85 percent observed by Swed (1985) in a field system over a similar depth of soil. Overall, between 86 to 95 percent of the applied TOC was removed by the system which, along with the virtually complete oxidation of TKN observed, corresponded favorably with the 90 percent removal of chemical oxygen demand (COD) in the aerobic columns of Magdoff et al. (1974). Soil-gas data showed a maximum drop in oxygen content within the unsaturated zone of less than 2 percentage points by volume, indicating that the system was predominantly aerobic.

The spiked feed experimental data showed higher TOC removal rates than were observed in the non-spiked feed runs. Percent removals of TOC over the first foot of unsaturated soil ranged between 84 to 90 percent with overall removals approaching 100 percent. The higher removal rates likely resulted from the higher proportion of soluble substrate in the spiked feed. In their study of rapid infiltration systems, Zaghloul et al. (1987) predicted that all of the soluble carbonaceous substrate in primary effluent applied to soils would be removed in the first 1.2 cm of soil whereas the particulate TOC removal was depth dependent; in fact, their predicted percent TOC removals of 77 percent over a 30 cm depth and 88 percent over a 107 cm depth in a sandy soil correlated very well with the percent removals measured in the non-spiked feed experiments.

However, the spiked feed data also showed that not all of the soluble substrate was removed as quickly as predicted by Zaghloul et al. (1987). A possible explanation is that at the high TOC concentrations present in the spiked feed (approximately 675 mg/L), the rate of substrate utilization is expected to be flux and substrate limited by oxygen (Williamson and McCarty 1976; Parker and Merrill 1984). However, even in this case, an unsaturated zone depth of approximately 107 cm was sufficient to obtain a relatively high level of treatment.

Soil-gas oxygen contents in the spiked feed experiments dropped to 10 percent by volume immediately beneath the infiltrative surface. However, no methane was detected anywhere within the system, and a mass balance on nitrogen over the system did not suggest that any significant amount of denitrification was occurring therein.

Respiratory Quotient Calculations: The respiratory quotient (RQ) is defined as the ratio of the carbon dioxide evolution rate (CER) over the oxygen uptake rate (OUR). The total system OUR results from the oxidation of both carbonaceous and nitrogenous substrates. However, the oxidation of reduced nitrogen (e.g., ammonia) to nitrate does not produce any carbon dioxide, thereby tending to lower the measured RQ of the system. In the spiked feed experiments, the system OUR was 45 mmol O_2/day, the CER was 8 mmol CO_2/day, and the calculated RQ was 0.18. These figures were calculated from a mass balance on the component gases in the air inlet and outlet (Fig. 1). Assuming that all of the TKN removed within the system was in the form of ammonia, the nitrogenous OUR was calculated to be 11 mmol O_2/day, giving a carbonaceous OUR of 34 mmol O_2/day and a RQ of 0.24. This RQ is still considerably less than the RQ of 1.0 expected for glucose oxidation. Equilibrium calculations indicate

that more than 60 percent of the CO_2 evolved in biochemical reactions remained in the soil-water as $CO_2(aq)$ and HCO_3^- because of the high CO_2 concentration in the soil-gas phase. The pH of the percolating wastewater also plays a role in determining the magnitude of this effect. The impact of CO_2 dissolution on gas transport in SASs will be discussed in the next section.

Modeling Work

Model Calibration and Verification: The input data required for the model can be categorized as follows:

- physical constants, including the trench/bed width, spacing between adjacent trenches/beds, depth to infiltrative surface, depth to groundwater, hydraulic loading rate, atmospheric gas composition, binary diffusivities, and temperature;
- constituent concentrations (TOC and nitrate-nitrogen) at various depths within the unsaturated zone;
- the total porosity and air-filled porosities at various depths within the unsaturated zone;
- fitting parameters, including the mathematical relationship for the obstruction factor Q as a function of air-filled and total soil porosity, and the respiratory quotient r; and
- numerical model parameters, including node spacing and the desired error tolerance.

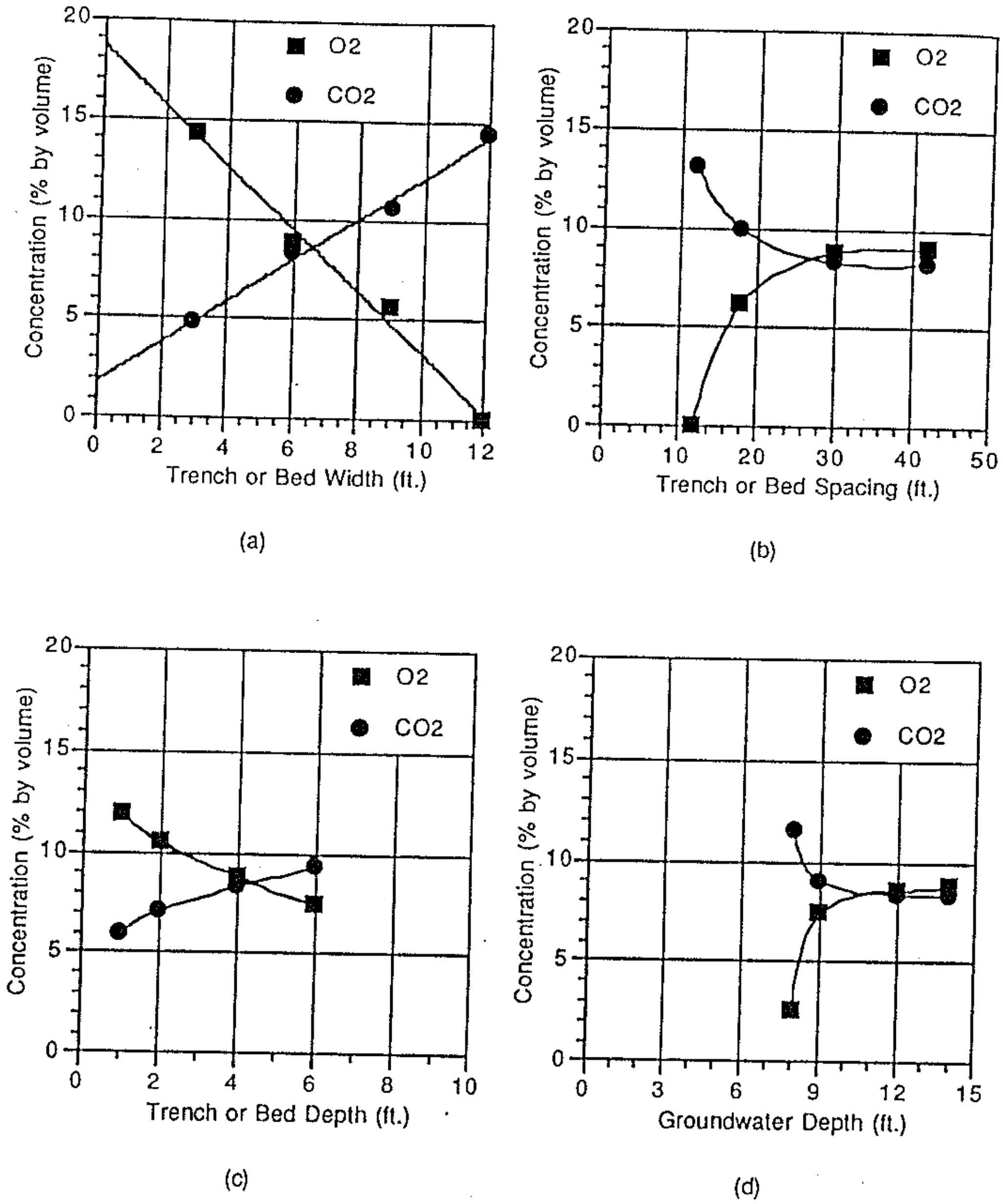
The model was calibrated using the data obtained from the spiked feed experiments. The best fit was determined by comparing the experimental data with the model data for each of the four soil gas components (CO_2, O_2, N_2, and Ar) at each sampling point in the unsaturated zone. In the case of CO_2, the average difference found between the model predictions and experimental data was 0.8 percentage points by volume; for O_2, the average difference was 1.5 percentage points; for N_2, the average difference was 1.2 percentage points. By way of comparison, the measured CO_2 concentrations in the unsaturated zone varied from 0.6 to 9.2 percent; for O_2, the concentrations varied from 17.1 to 7.0 percent; and for N_2, the concentrations varied from 80.4 to 84.2 percent.

The model was subsequently verified using the data from the non-spiked feed runs. The only input parameters that were changed from the spiked feed runs conducted under similar conditions were the TOC and nitrate profiles. The results were very similar to those just discussed for the calibration run.

Sensitivity Analysis: This analysis was performed as follows. First, a base case set of input data and model results were established. Then while holding all other input parameters constant, one parameter was varied over a range of values that might possibly be encountered in an actual field system. In the base case run, the total porosity of the soil was assumed to be 0.40, the hydraulic loading rate used was 1.2 cm/d, and the influent TKN was set equal to 50 mg/L.

Figs. 2a-d show the sensitivity of the model results to changes in the physical dimensions of the SAS. In these and all other analyses presented in this chapter, the O_2 and CO_2 concentrations given are the smallest and greatest values, respectively, predicted by the model. The point at which this occurred was located on the trench centerline immediately beneath the infiltrative surface.

The results for trench/bed width (base case = 1.8 m = 6 ft) indicate that anaerobic conditions would be expected for bed widths greater than 3.7 m (12



Figs. 2a-d Model Sensitivity to Changes in Physical Dimensions of the SAS

ft). Field evidence to support this finding was obtained by Swed (1985) in his study of a 30 m wide SAS bed under similar loading conditions.

The results for trench/bed spacing (base case = 9.1 m = 30 ft centerline-to-centerline) show that increasing the bed spacing beyond approximately 9 m (30 ft) yields no measurable improvement in oxygen supply to the system. Conversely, significant oxygen depletion within the unsaturated zone can be expected when the bed spacing is less than about 6.1 m (20 ft). This reflects the reduced surface area available for oxygen diffusion into the soil. Therefore, for full-scale systems under similar conditions, the optimum bed spacing should be between 6 and 9 m.

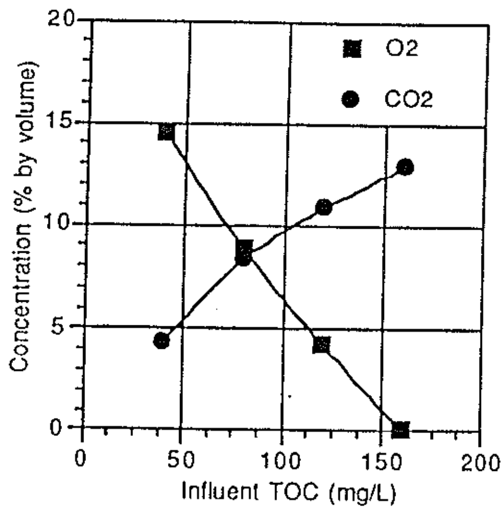
In comparison with the other physical dimensions, variations in O_2 and CO_2 resulting from changes in the trench/bed depth below the ground surface (base case = 1.2 m = 4 ft) are minor. The minimum depth necessary to prevent

freezing and surface seepage of raw wastewater should be the primary consideration in this case.

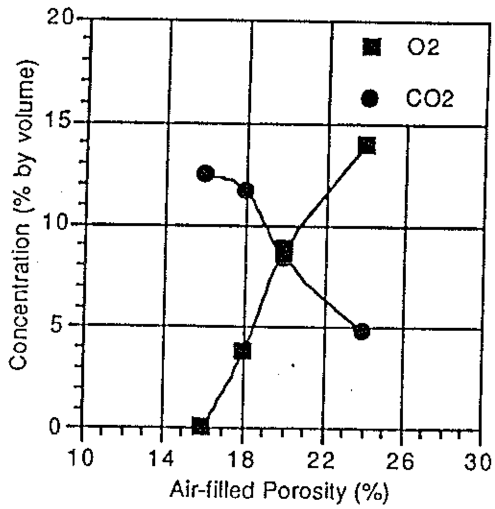
Shallow depths to groundwater (base case = 3.7 m = 12 feet) can severely restrict the diffusion of oxygen into the SAS. In this case, depths of less than 2.7 m or 9 ft (corresponding to unsaturated zone depths beneath the infiltrative surface of less than 1.5 m or 5 ft) result in rapid depletions of oxygen therein. The exact depth at which this occurs is also dependent upon the air-filled porosity of the soil. However, the data does indicate that the minimum groundwater depths of 0.6 to 1.2 m (2 to 4 ft) beneath the infiltrative surface recommended by the EPA (1980) may not be sufficient to maintain aerobic soil conditions within these systems.

The organic content of the applied wastewater and the air-filled porosity of the soil can also significantly affect the oxygen concentration in the soil-gas phase (Figs. 3a,b). The model shows that doubling the TOC concentration of the applied wastewater (base case = 80 mg/L) results in the complete depletion of oxygen immediately beneath the infiltrative surface of the system. This indicates that system design criteria used to size SASs for domestic wastewater cannot be indiscriminately applied to the design of systems handling commercial or industrial wastewaters that can contain much higher substrate concentrations. In these cases, lower hydraulic loading rates, narrower bed widths, and/or other system modifications should be considered.

The model results are also very sensitive to the air-filled porosity of the soil (base case = 20 percent) over a relatively small range of porosities. This implies that the texture of the soil will significantly affect the supply of oxygen to the system, and may be the most important factor considered thus far. This also may explain much of the error observed between the experimental data and the model calibration and verification results, since the potential error incurred in estimating the air-filled porosity from the SMCC was of the same order of magnitude as the range of air-filled porosities shown in Fig. 3b. Another method (e.g., gamma-ray scanning) should be used to more accurately measure the air-filled porosities in future studies of this type.



(a)



(b)

Figs. 3a,b Model Sensitivity to Changes in the Organic Content of the Applied Wastewater and the Air-filled Porosity of the Soil

CONCLUSIONS

The major conclusions drawn from this work are summarized below: