Effects of Aeration on Water Quality from Septic System Leachfields

David A. Potts, Josef H. Görres, Erika L. Nicosia, and José A. Amador*

ABSTRACT

We conducted a pilot-scale study at a research facility in southeastern Connecticut to assess the effects of leachfield aeration on removal of nutrients and pathogens from septic system effluent. Treatments consisted of lysimeters periodically aerated to maintain a headspace O_2 concentration of 0.209 mol mol^{-1} (AIR) or vented to an adjacent leachfield trench (LEACH) and were replicated three times. All lysimeters were dosed with effluent from a septic tank for 24 mo at a rate of 12 cm d^{-1} and subsequently for 2 mo at 4 cm d^{-1}. LEACH lysimeters had developed a clogging mat, or biomat, 20 mo before the beginning of our study. The level of aeration in the AIR treatment was held constant regardless of loading rate. No conventional biomat developed in the AIR treatment, whereas a biomat was present in the LEACH lysimeters. The headspace of LEACH lysimeters was considerably depleted in O_2 and enriched in CH_4, CO_2, and H_2S relative to AIR lysimeters. Drainage water from AIR lysimeters was saturated with O_2 and had significantly lower pH, five-day biological oxygen demand (BOD_5), and ammonium, and higher levels of nitrate and sulfate than LEACH lysimeters regardless of dosing rate. By contrast, significantly lower levels of total N and fecal coliform bacteria were observed in AIR than in LEACH lysimeters only at the higher dosing rate. No significant differences in total P removal were observed. Our results suggest that aeration may improve the removal of nitrogen, BOD_5, and fecal coliforms in leachfield soil, even in the absence of a biomat.

Approximately 23% of households in the United States rely on onsite wastewater treatment systems for disposal of domestic sewage (United States Census Bureau, 2003). Conventional septic systems are designed for removal of solids in the septic tank, and dispersal of wastewater in the associated leachfield. The passage of effluent through leachfield soil results in the removal of pathogens and biodegradable organic carbon at rates generally exceeding 90% (USEPA, 2002). Removal rates for N and P in the leachfield of conventional septic systems are more modest, ranging from 50 to 85% for P and 0 to 40% for N (Kaplan, 1987; USEPA, 2002). The water quality enhancement functions of leachfields are thought to be associated with the biological and hydraulic processes that take place in the clogging mat, or biomat, at the infiltration surface of the leachfield trench (USEPA, 2002). There is growing concern among water management and regulatory agencies that failing or improperly installed septic systems cause contamination of ground and surface waters with pathogens, nutrients, and biologically active compounds (Canter and Knox, 1985; Yates, 1985). In response to more stringent regulation of the quality of effluent delivered by septic systems, new technologies that lower nutrient and pathogen emissions have been developed.

The capacity of leachfield soils to enhance water quality can vary substantially with environmental conditions. For instance, fluctuations in depth to the water table and in soil temperature can affect leachfield functioning (Bouma et al., 1975; Cogger and Carlile, 1984; Viraraghavan and Dickenson, 1991). One approach to improving the quality of water coming out of septic systems is to promote conditions that enhance contaminant removal and/or retention in the leachfield. The biogeochemical transformations in leachfield soil are controlled by the type and availability of electron acceptors and donors. For example, sufficiently high levels of oxygen are necessary for microbial oxidation of ammonium to nitrate. Nitrate may then be removed by denitrification, but only if the amount of organic carbon is sufficient to support the activities of denitrifying bacteria. In addition, most of the soil meso- and microfauna thought to be involved in pathogen removal use O_2 as an electron acceptor. Levels of O_2 may be suboptimal below the biomat due to the combined effects of high rates of microbial activity and low gas diffusion rates through leachfield soil. Enhanced aeration of leachfield soil may thus improve its ability to remove nutrients and inactivate pathogens via abiotic and biological processes that require oxygen. It may also promote anaerobic processes in microsites, such as denitrification, that rely on the products of aerobic processes. Finally, aeration may be expected to enhance wastewater infiltration by reducing or eliminating the biomat (e.g., Erickson and Tyler, 2001).

A patented process for the rejuvenation of leachfields using aeration (Potts, 2000) has been successful in restoring hydraulic function in more than 60 failed onsite wastewater treatment systems in the eastern United States. The effects of this process on water quality leaving the leachfield, however, are not known. We conducted a pilot-scale study to evaluate the effects of aeration levels on the quality of water coming out of leachfields. The effluent from a household septic tank was passed through lysimeters filled with sand to a depth of 30 cm. Silica sand with a high uniformity coefficient was used because it is chemically inert and it represents the shortest retention time and thus the case with the least effluent treatment. Experimental treatments consisted of lysimeters vented to the leachfield, representing the conditions in a conventional system, and lysimeters aerated to maintain an oxygen concentration of 0.20 to 0.21 mol mol^{-1}. We measured the concentration of total N and P, pH, ammonium, nitrate, sulfate, phosphate, reduced iron, BOD_5, fecal coliforms, and *Escherichia coli* in the septic tank effluent coming into the lysimeters and in the drainage water from aerated and leachfield lysimeters. We

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Abbreviations: AIR, aerated lysimeters; BOD_5, five-day biological oxygen demand; LEACH, lysimeters vented to leachfield.

also determined the composition of the atmosphere in the headspace in both treatments. The effects of aeration were determined at loading rates of 12 cm d^{-1} (approximately 3 gallons ft^{-2} d^{-1}) and 4 cm d^{-1} (approximately 1 gallon ft^{-2} d^{-1}).

MATERIALS AND METHODS

Experimental Facility

The study was conducted in a laboratory facility built adjacent to a two-story, two-family home in southeastern Connecticut, USA. The home was built in 1983 and was fitted with a new conventional septic system in 1996. The septic tank had a maximum capacity of 4733 L (1250 gallons) and was not pumped during the course of the study. The home was inhabited continuously by three to six people.

A schematic diagram of the experimental setup is shown in Fig. 1. All of the effluent from the septic tank was diverted to a pump station and stored in a high-density polyethylene (HDPE) tank (1325-L [350-gallon] maximum capacity) housed in a climate-controlled (17–19°C) room above the laboratory. The contents of the tank were mixed at regular intervals using a pump. Wastewater from the tank was pumped through a 3.75-cm-diameter (1.5-in) PVC manifold to a series of dosing tanks in the laboratory. Cylindrical, HDPE dosing tanks (30.5-cm [12-in] i.d., 45.7-cm [18-in] height) had a maximum capacity of 38 L (10 gallons) and were dosed every 6 h. Dosing was regulated using electronically actuated valves. Dosing tank overflow was allowed to drain completely until only the desired dose volume was retained.

Wastewater from the dosing tank flowed by gravity into a lysimeter that consisted of a HDPE cylinder (43.2-cm [17-in] i.d., 45.7-cm [19-in] height) fitted with a drainage fitting 1.91 cm (0.75 in) from the bottom, and water and gas input fittings and inspection port on top. Wastewater was delivered to the surface of the soil through a horizontal, 1.91-cm-diameter (0.75-in) PVC pipe in which 0.64-cm-diameter (0.25-in) holes were drilled into the top of 0.1-cm (0.04-in) slotted well screen mesh. The bottom of the lysimeter was filled with 7.5 cm of no. 4 silica sand (diameter = 4.75 to 1.40 mm; uniformity coefficient <1.8), on top of which was placed 30 cm (12 in) of no. 00 silica sand (diameter = 0.71 to 0.21 mm; uniformity coefficient <1.6) (U.S. Silica Co., Berkeley Springs, WV), with headspace constituting the volume above the sand.

Lysimeters were dosed with wastewater at a rate of 12 cm d^{-1} (approximately 3.0 gallons ft^{-2} d^{-1}) for the first 24 mo of the experiment. LEACH lysimeters had developed a clogging mat, or biomat, approximately 20 mo before the beginning of our study, as indicated by periodic visual inspection. On 21 Apr. 2003 the dosing rate was changed to 4 cm d^{-1} (approximately 1.0 gallon ft^{-2} d^{-1}) to determine the extent to which the effects of aeration on leachfield water quality were affected by dosing rate. The aeration level in the AIR treatment was kept constant regardless of dosing rate.

Treatments

Treatments consisted of lysimeters with aerated (AIR) or unaerated (LEACH) headspace, with each treatment replicated three times. Aeration was accomplished using a patented process developed by Geomatrix, LLC (Potts, 2000). This process has been used successfully in hydraulic rejuvenation of failed septic system leachfields, but its effects on water quality are not known. Ambient air was pumped at regular intervals into the headspace of AIR lysimeters to maintain an O_2 level of 0.20 to 0.21 mol mol^{-1} using a piston pump. This resulted in a slight (approximately 2.5–6.7 kPa) positive pressure within AIR lysimeters. To mimic the in situ composition of the atmosphere found in the leachfield, the headspace and the gravel bed below the soil of LEACH lysimeters were vented to the septic system leachfield.

Sampling and Analyses

Sampling and Processing

Raw wastewater samples were collected from a valve in the input stream (Fig. 1) and placed in autoclaved, 125-mL polypropylene screw-cap bottles. One to two liters of wastewater were allowed to flow through the valve before sample collection. Water samples from the lysimeters were collected in 3-L Tedlar bags (2-mil thick; SKC, Eighty Four, PA) at ambient temperature (17–19°C). The bag was connected to the lysimeter outlet using Tygon tubing. To ensure that water samples were exposed to an atmosphere with the same composition as that found in the lysimeters, a connection was made from the headspace of each lysimeter to the drainline connected to the sampling bag. Sampling of water from the lysimeters started coincident with a dosing event, and continued for 60 to 90 min. Between 700 and 900 mL of water were collected from each lysimeter on each sampling date.

A portion of the raw wastewater and lysimeter water samples was analyzed immediately for dissolved oxygen and the presence of Fe^{2+}. The remaining sample was kept on ice during transport to the laboratory in Kingston, RI (approximately 1 h). Unfiltered samples were assayed immediately on arrival to the laboratory for BOD_5, pH, and total fecal coliforms and *E. coli*. A portion of the unfiltered sample was frozen for subsequent determination of total N and total P content. The remaining sample was filtered by passing through a glass microfiber filter (GF/F, 25-mm diameter; Whatman, Maidstone, UK). The filtered samples were stored in plastic, screw-cap scintillation vials at 4°C and analyzed for NH_4^+ and NO_3^- within 24 h, for PO_4^{3-} within 48 h, and for SO_4^{-2} within 72 h of collection.

Leachfield gases venting into LEACH lysimeters and headspace gases from the AIR and LEACH lysimeters were sampled using a portable soil gas monitor (SoilAir Technology, East Longmeadow, MA). Carbon dioxide, CH_4, O_2, and H_2S were determined using infrared, catalytic bead, galvanic, and electrochemical sensors, respectively. Measurements were made approximately 2 h after dosing of lysimeters with septic tank effluent. Gas samples were drawn at a rate of approximately 0.05663 m^3 h^{-1} (2.0 standard ft^3 h^{-1}) for 30 to 60 s and the maximum values detected during that sampling period are reported for all gases except O_2, for which the minimum value is reported.

Analyses

Constituent analyses were performed according to methods of the American Public Health Association (1998). Dissolved oxygen was measured using the azide modification of the Winkler titration method. The concentration of Fe^{2+} in water was determined using EM Quant iron (Fe^{2+}) test strips (EM Industries, Gibbstown, NJ). The pH of water samples was determined using a combination pH electrode and a Model UB-10 pH meter (Denver Instruments, Denver, CO). The concentration of sulfate was measured using the barium chloride turbidimetric method. Nitrate, ammonium, and phosphate concentrations of water samples were determined colorimetrically using an automated nutrient analyzer (Flow Solution IV; Alp-

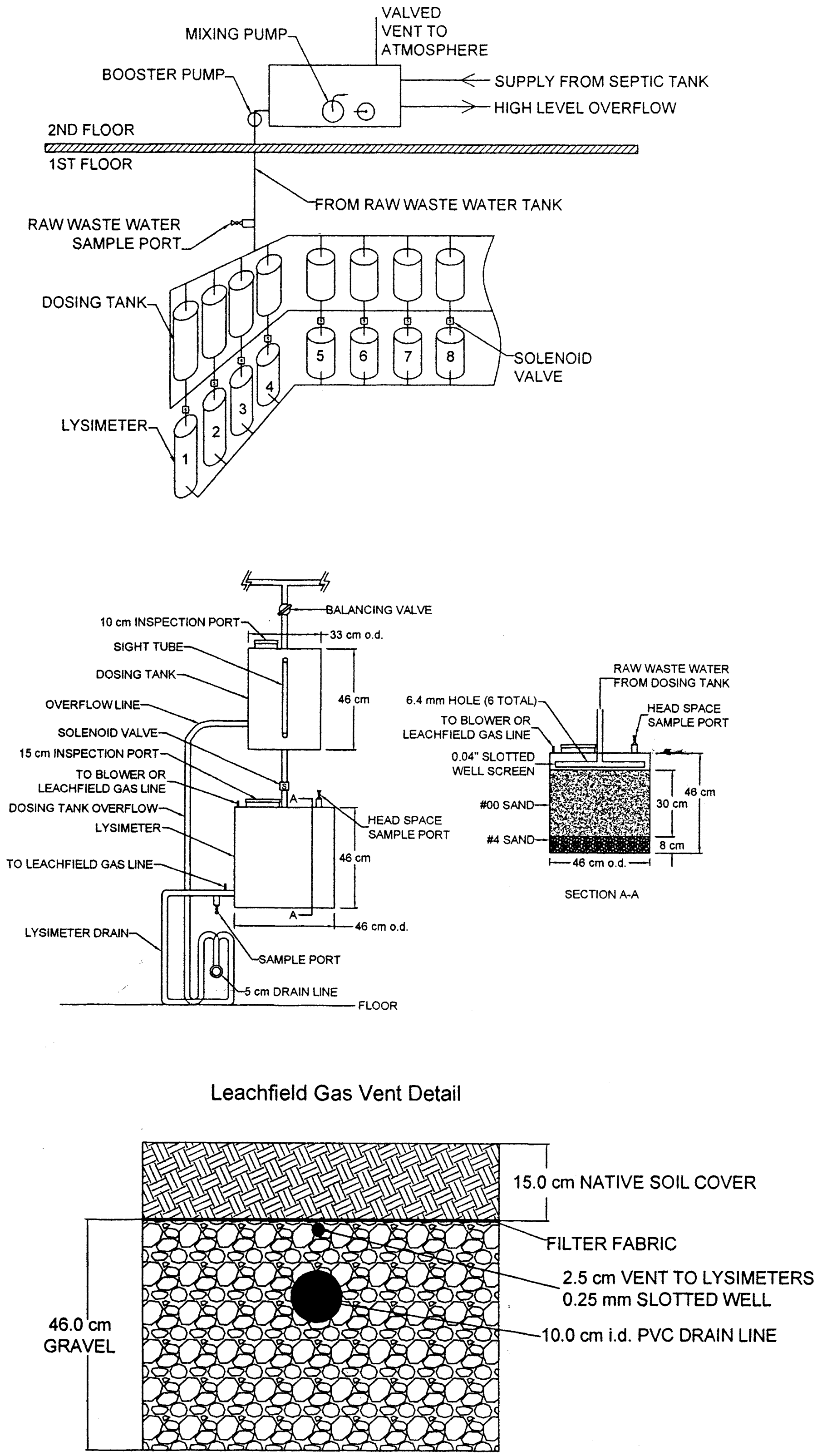

Fig. 1. Schematic diagram of experimental facility (top), lysimeters (middle), and detail of leachfield gas intake (bottom). Drawings are not to scale.

kem, College Station, TX). The total N and total P content of water samples was determined using the persulfate digestion method. Samples were digested by autoclaving at 121°C and analyzed colorimetrically for NO_3^- and PO_4^{3-} as described above. Fecal coliforms and *E. coli* were assayed using the standard fecal coliform membrane filtration procedure. The BOD_5 was measured on undiluted, unamended samples by manometric respirometry using an OxiTop BOD system

(WTW, Fort Myers, FL) at 21 ± 1°C. Volumes were 250 mL for raw and LEACH samples, and 432 mL for AIR samples.

Statistical Analyses

Differences between AIR and LEACH treatments were evaluated using Student's *t* test at the 95% confidence level (SigmaStat for Windows, Version 2.03; SPSS, 1995). Statistical analyses were performed on untransformed data except for total coliform bacteria, where data were log-transformed.

RESULTS

Operating Conditions

LEACH lysimeters had developed a clogging mat, or biomat, approximately 20 mo before the beginning of our study, as indicated by periodic visual inspection. Ponding in LEACH lysimeters was observed throughout the sampling period, with the volume of ponded effluent increasing when the loading rate decreased from 12 to 4 cm d^{-1}, probably as a result of the reduced hydraulic head at the lower loading rate. By contrast, visual inspection of AIR lysimeters indicated that a biomat had not formed at the soil surface before the beginning of the study, and none formed by the end of the sampling period. Ponding was not observed in AIR lysimeters regardless of dose. The temperature of septic system effluent coming into the lysimeters was generally 2 to 3°C lower than that of drainage water from the lysimeters (Fig. 2). Both values increased with time, and were close to ambient temperature (17–19°C) by the final sampling date. We did not observe treatment differences in the temperature of lysimeter drainage water.

Headspace Gases

Levels of O_2 in the gases coming into the LEACH lysimeters ranged from 0.06 to 0.160 mol mol^{-1} during the period examined. Oxygen in the headspace of the LEACH treatment ranged from 0.078 to 0.197 mol mol^{-1}, whereas in the AIR treatment the concentration of oxygen was 0.209 mol mol^{-1} on all sampling dates (Table 1). The concentration of CO_2 in leachfield gases ranged from 0.013 to greater than 0.05 mol mol^{-1}. Carbon dioxide levels were one to two orders of magnitude higher in LEACH than AIR treatments throughout the measurement period. Methane levels in leachfield gases ranged from 2 to $>50\,000 \times 10^{-6}$ mol mol^{-1}. Levels of CH_4 in the headspace of LEACH lysimeters varied from 750 to $>50\,000 \times 10^{-6}$ mol mol^{-1}. By contrast, methane concentrations in the AIR treatment ranged from 0 to 65×10^{-6} mol mol^{-1}. No hydrogen sulfide was detected in the AIR lysimeters on any sampling date, whereas the concentration of H_2S in LEACH lysimeters ranged from 0 to $>100 \times 10^{-6}$ mol mol^{-1}.

Water Quality Parameters

The pH of water from LEACH lysimeters was indistinguishable from that of the raw wastewater, regardless of sampling date, ranging between 6.2 and 7.0 (Fig. 2). By contrast, water from AIR lysimeters had pH values that were consistently and significantly lower than in the LEACH treatment, with values ranging from 3.9 to 4.4.

Dissolved oxygen levels in incoming wastewater ranged from 0 to 0.9 mg O_2 L^{-1} (Fig. 2). Aeration enhanced dissolved oxygen levels, with values ranging from 8.1 to 11.9 mg O_2 L^{-1} in water from AIR treatments. Dissolved oxygen in the LEACH treatment was significantly lower than in the AIR treatment on all sampling dates, with values ranging from 0 to 3.3 mg O_2 L^{-1}.

The BOD_5 of incoming wastewater ranged from 73 to 198 mg L^{-1} (Fig. 2). Values of BOD_5 in LEACH lysimeters ranged from 38 to 168 mg L^{-1}, with average removal rates of 29.6 and 60.9% at loading rates of 12 and 4 cm d^{-1}, respectively (Table 2). Values of BOD_5 in AIR lysimeters were significantly lower than in the LEACH treatment, and were below the detection limit of 1 mg L^{-1}, on all sampling dates, with removal rates higher than 99% (Table 2).

Values for fecal coliforms and *E. coli* were identical in all instances, and only levels of fecal coliforms are reported (Fig. 3). Levels of fecal coliforms and *E. coli* in wastewater ranged from 10^3 to 10^6 colony-forming units (CFU) 100 mL^{-1} (Fig. 3). Numbers of fecal coliforms and *E. coli* were significantly lower in AIR than in LEACH treatment on all sampling dates. Reduction in fecal coliforms and *E. coli* ranged from 98.0 to 98.6% (1 to 3 log units) in the LEACH treatment and 99.2 to 99.9% (2 to 6 log units) in the AIR treatment (Table 2).

The total N concentration in incoming wastewater ranged from 22 to 48 mg N L^{-1} with inorganic N making up 60 to 80% of the total N pool (Fig. 4). The concentration of total N was significantly lower in AIR than in LEACH lysimeters only on the first three sampling dates, when the nominal loading rate was 12 cm d^{-1}. After the wastewater loading rate was reduced to 4 cm d^{-1} (while holding the aeration level constant) there were no statistically significant differences in total N between treatments. Removal of nitrogen in LEACH lysimeters at the high loading rate was 1.3%, whereas in AIR lysimeters it was 23.6% (Table 2). Little to no net removal of total N was observed in either treatment at the low dosing rate (Table 2). Inorganic nitrogen constituted between 60 and 90% of total N in water from AIR lysimeters, and 90 to 100% of total N in LEACH lysimeters. Differences in NO_3^- and NH_4^+ concentrations between treatments were statistically significant on all sampling dates, with NO_3^- dominating the inorganic N pool in the AIR lysimeters and NH_4^+ constituting the bulk of the inorganic N pool in LEACH lysimeters (Fig. 4).

Levels of total P in incoming wastewater ranged from 2.8 to 11.8 mg L^{-1}, with phosphate constituting 30 to 50% of total P (Fig. 5). No significant differences between treatments were observed in the concentration of total P or phosphate coming out of lysimeters on any sampling date (Fig. 4). Furthermore, there was no net removal of total P in either treatment (Table 2); rather, there was a net increase of 1 to 13.4% in total P lysimeter drainage water relative to effluent inputs.

The concentration of sulfate in incoming wastewater

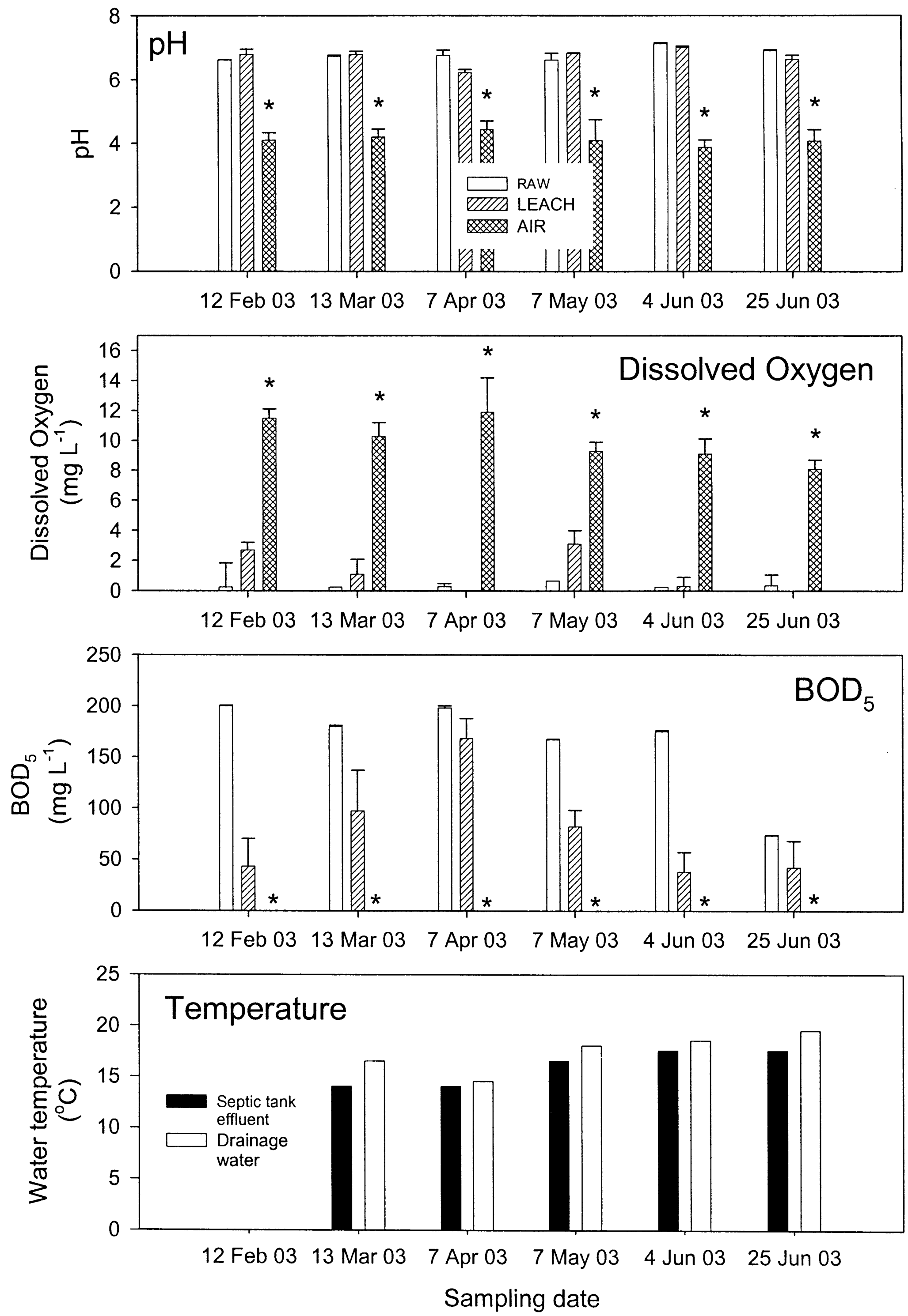

Fig. 2. Time course of pH, dissolved oxygen, five-day biological oxygen demand (BOD_5), and temperature of raw wastewater input and of drainage water from aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH). The wastewater loading rate was changed from 12 to 4 cm d^{-1} on 21 Apr. 2003. Temperature was not measured on 12 Feb. 2003. Values are means (n = 3). Bars represent one standard deviation. Significant differences between AIR and LEACH treatments ($P < 0.05$) for a particular sampling date are indicated with an asterisk (*).

ranged from 2.9 to 7.4 mg L^{-1} (Fig. 6). Water from LEACH lysimeters had significantly lower concentrations of SO_4^{2-} than AIR lysimeters. Sulfate levels were reduced 40 to 50% in LEACH lysimeter water relative to incoming values, whereas the concentration of sulfate increased by a factor of 2 to 3 after passage through AIR lysimeters. No Fe^{2+} was detected in raw wastewater or in water from AIR lysimeters, whereas low levels of

Table 1. Concentration of O_2, CO_2, CH_4, and H_2S in the headspace of aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH) and in the gas inputs to LEACH lysimeters on different sampling dates.†

Sampling date	Treatment	O_2	CO_2	CH_4	H_2S
		mol mol^{-1}		10^{-6} mol mol^{-1}	
12 Mar. 2003	LEACH, input gases	0.160	0.013	30 000	>100
	LEACH, headspace	0.140–0.197	0.007–0.024	750 to >50 000	0–65
	AIR, headspace	0.209	0–0.0003	15–35	0
7 Apr. 2003	LEACH, input gases	<0.021	>0.05	>50 000	>100
	LEACH, headspace	0.085–0.135	>0.05	>50 000	>100
	AIR, headspace	0.209	0.0004–0.0008	15–65	0
14 May 2003	LEACH, input gases	0.040	>0.05	31 250	>100
	LEACH, headspace	0.078	>0.05	>50 000	>100
	AIR, headspace	0.209	0–0.0004	0	0
4 June 2003‡	LEACH, input	0.006	>0.050	20 000	12
	AIR, headspace	0.209	0–0.0012	0–2	0
25 June 2003	LEACH, input	0.093	>0.050	3	0
	AIR, headspace	0.209	0–0.0002	0–5	0

† Headspace values represent range for three replicate lysimeters. Values for LEACH input gases represent a single measurement. No measurements were made on 12 Feb. 2003.

‡ The headspace in LEACH lysimeters was flooded on 4 and 25 June 2003, preventing sampling of headspace gases.

Fe^{2+} (0–3 mg L^{-1}) were observed in LEACH lysimeters on the first four sampling dates (data not shown).

DISCUSSION

Headspace Gases and Dissolved Oxygen

Aeration of soil in AIR lysimeters resulted in levels of O_2 identical to those found in ambient air, and resulted in drainage water saturated with oxygen. By contrast, LEACH lysimeters, which were vented to the leachfield trench, had relatively low headspace O_2 levels and associated low concentrations of O_2 in drainage water. There was little or no measurable O_2 in incoming wastewater, so O_2 dissolved in the drainage water of LEACH lysimeter must have come from gases in the headspace and the gravel below the sand bed. Both of these were vented to the septic system leachfield, and O_2 was present in varying concentrations in the incoming vent gases (Table 1). Walker et al. (1973) found levels of O_2 and CO_2 of 0.196 and 0.007 mol mol^{-1}, respectively, in soil 5 to 10 cm below the biomat of a leachfield from a conventional septic system, suggesting that the soil was fairly well aerated. However, our own field measurements, using the same equipment and methodology as in the present study, suggest that O_2 levels in the atmosphere of the soil below a leachfield can vary widely (0–0.09 mol mol^{-1}), even at depths much greater than the 30 cm used in our study. The presence of high levels of H_2S and CH_4 in the gases venting from the leachfield supports the contention that anaerobic conditions prevailed in the leachfield trench above the biomat on most sampling dates. Methane was also found in the headspace of AIR lysimeters, albeit at concentrations approximately 1000 times lower than in the LEACH treatment (Table 1). The presence of CH_4 in the AIR treatment may be the result of out-gassing of methane dissolved in incoming wastewater and/or the establishment of anaerobic conditions in microsites within the lysimeters, perhaps shortly after dosing with wastewater.

Transformation and Removal of Nitrogen

Aeration had a strong effect on N removal and speciation and it clearly promotes oxidation of ammonium to nitrate. Nitrification results in proton release that, in the poorly buffered quartz sand used in this experiment, results in acidic conditions. The low pH values observed in the AIR lysimeters support this interpretation. The net loss of total N in AIR lysimeters on the first three sampling dates is probably due to denitrification, which is promoted by the presence of high levels of nitrate in the effluent. Conversely, passage of wastewater through LEACH lysimeters only resulted in ammonification of wastewater N, with little or no NO_3^- production or net removal of N. The absence of net N losses in LEACH lysimeters suggests that denitrification did not take place in this treatment, probably limited by very low nitrate levels.

Denitrification may occur in anaerobic microsites even when oxygen concentrations in the bulk medium are high (Conrad, 1996; Sexstone et al., 1985). In addition, denitrification can take place under aerobic conditions. For example, Robertson and Kuenen (1991) have shown that in batch culture experiments, *Thiosphaera pantotropha*, a known denitrifier, can remove O_2 and nitrate simultaneously, and the presence of both electron acceptors resulted in more rapid growth on acetate than when either O_2 or nitrate was present. Studies of nitrogen removal in buried sand filters (of similar age and under similar loading and temperature conditions to ours) have also suggested that denitrification, rather than microbial immobilization, is responsible for total

Table 2. Average extent of removal for total N, total P, five-day biological oxygen demand (BOD_5), and fecal coliforms in aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH) at different loading rates.

Parameter	Treatment	Removal rate at a loading rate of 12 cm d^{-1}	4 cm d^{-1}
		%	
Total N	LEACH	1.3	0.8
	AIR	23.6	2.9
Total P	LEACH	(8.2)†	(1.0)
	AIR	(13.4)	(10.6)
BOD_5	LEACH	29.6	60.9
	AIR	99.5	99.3
Fecal coliforms	LEACH	98.0	98.9
	AIR	99.1	99.9

† Values in parentheses indicate average increases.

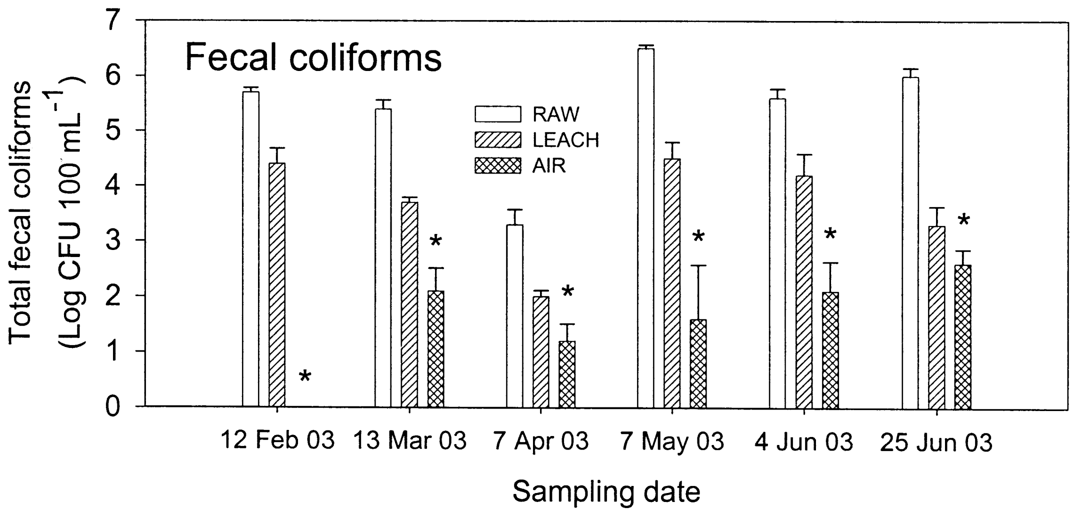

Fig. 3. Time course of fecal coliform bacteria in raw wastewater input and in drainage water from aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH). The wastewater loading rate was changed from 12 to 4 cm d^{-1} on 21 Apr. 2003. Values are means (n = 3). Bars represent one standard deviation. Significant differences between AIR and LEACH treatments (P < 0.05) for a particular sampling date are indicated with an asterisk (*).

N losses (e.g., Gold et al., 1992). Measurement of the gaseous products of denitrification (e.g., NO, N_2O) should help in elucidating the role this process has in N removal in leachfield soils.

The fact that net N removal in AIR lysimeters was limited to those sampling dates when wastewater was applied at the higher rate (12 cm d^{-1}) suggests that the length of time the soil remains saturated with wastewater has a strong influence on whether N is removed by denitrification in the leachfield. The additional organic C associated with higher wastewater loading may also have enhanced denitrification by providing additional electron donors and/or promoted localized anaerobic conditions. In addition, transient increases in soil water content of a greater magnitude at the higher dosing rate may also have interfered with soil aeration. Denitrification in septic system leachfields is thought to be limited by the availability of organic C in wastewater (Sikora et al., 1976). Plósz et al. (2003) observed a similar effect of organic substrate input on denitrification rates in an anoxic reactor exposed to oxygen, which they attributed to high dissolved oxygen consumption, which in turn supported denitrifying conditions. Gaseous losses of N may also be due to the activities of nitrifying bacteria, which produce NO and N_2O under anaerobic or microaerophilic conditions (Groffman, 1991). Such conditions may be established temporally in areas of the lysimeter after dosing with wastewater.

The presence of nitrate in drainage water from LEACH lysimeters, albeit at low levels, indicates that nitrification is taking place in the soil, since there is no detectable nitrate in incoming wastewater. However, the low concentration of nitrate suggests that nitrification is inhibited to some extent under these conditions. Levels of dissolved oxygen in LEACH water were between 0 and 4 mg L^{-1} (Fig. 2) and O_2 was present in the leachfield gases (Table 1), although generally at low concentrations. The high concentration of methane may have inhibited ammonia oxidation. Methane is a competitive inhibitor of ammonia oxidation (Bedard and Knowles, 1989) and was present at concentrations exceeding $20\,000 \times 10^{-6}$ mol mol^{-1} on most sampling dates (Table 1). By contrast, extensive nitrification appeared to take place in AIR lysimeters, as indicated by levels of NO_3^- in drainage water that accounted for up to two-thirds of the total N inputs from septic tank effluent (Fig. 4). High levels of dissolved O_2 and NH_4^+ and methane levels that were three orders of magnitude lower than in LEACH lysimeters probably created conditions conducive for nitrification in AIR lysimeters. Our results suggest that the role of methane as an inhibitor of ammonia oxidation in leachfield soils warrants further study.

Removal of Biological Oxygen Demand

Greater BOD_5 removal in AIR than in LEACH lysimeters indicates that microbial decomposition of organic carbon is restricted under LEACH conditions. The enhanced availability of O_2 as a terminal electron acceptor in AIR lysimeters probably results in a shift toward aerobic respiration, which is more energetically efficient than fermentative and anaerobic respiration pathways (Fuhrmann, 1998). This may explain the absence of biomat development in AIR lysimeters, since a greater proportion of organic C inputs would be oxidized to CO_2. Removal of BOD_5 in AIR lysimeters was not affected by loading rate, whereas removal in LEACH lysimeters loaded at 4 cm d^{-1} was double that at 12 cm d^{-1}. The effect of loading rate on BOD_5 removal in the LEACH treatment may reflect the limited capacity of the LEACH soil to provide the microbial community with the types and levels of electron acceptors necessary to metabolize organic compounds efficiently.

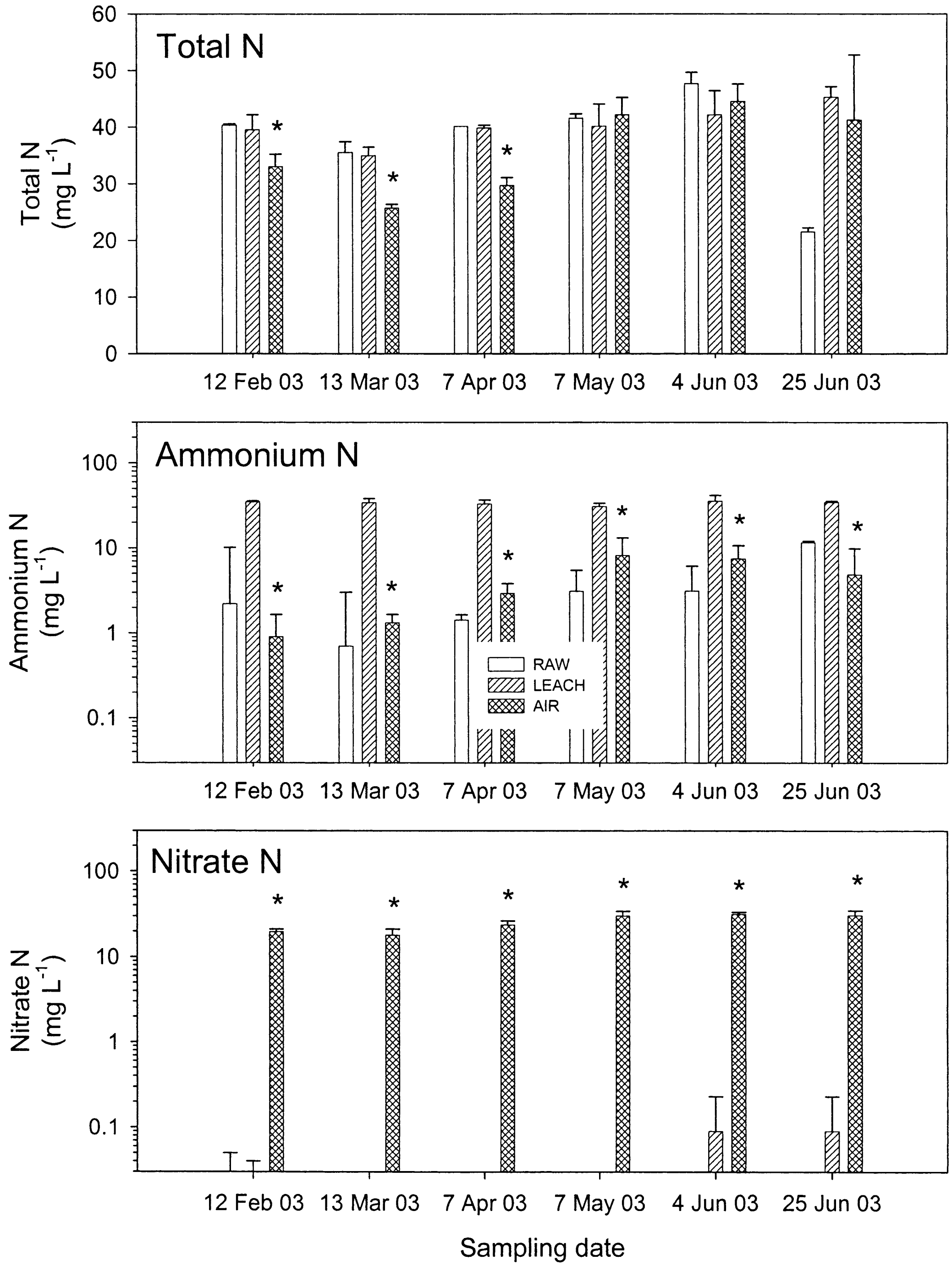

Fig. 4. Time course of total N, ammonium N, and nitrate N in raw wastewater input and in drainage water from aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH). The wastewater loading rate was changed from 12 to 4 cm d^{-1} on 21 Apr. 2003. Values are means (n = 3). Bars represent one standard deviation. Significant differences between AIR and LEACH treatments ($P < 0.05$) for a particular sampling date are indicated with an asterisk (*).

Sulfate Production

Higher levels of sulfate in AIR than in LEACH lysimeters are indicative of enhanced microbial oxidation of reduced S compounds found in septic tank effluent. For example, a number of species within the genus *Thiobacillus* derive energy from the oxidation of H_2S and $S_2O_3^{2-}$ (formed from bacterial reduction of organic sulfur compounds) within the pH values observed in our experiments (Germida et al., 1992). Furthermore, some of these bacteria are also capable of denitrification (Kanter et al., 1998; Robertson and Kuenen, 1991), possibly contributing to the loss of N in AIR lysimeters.

Total Phosphorus and Phosphate

The lack of PO_4^{3-} or total P removal in AIR or LEACH lysimeters is not surprising. The main mecha-

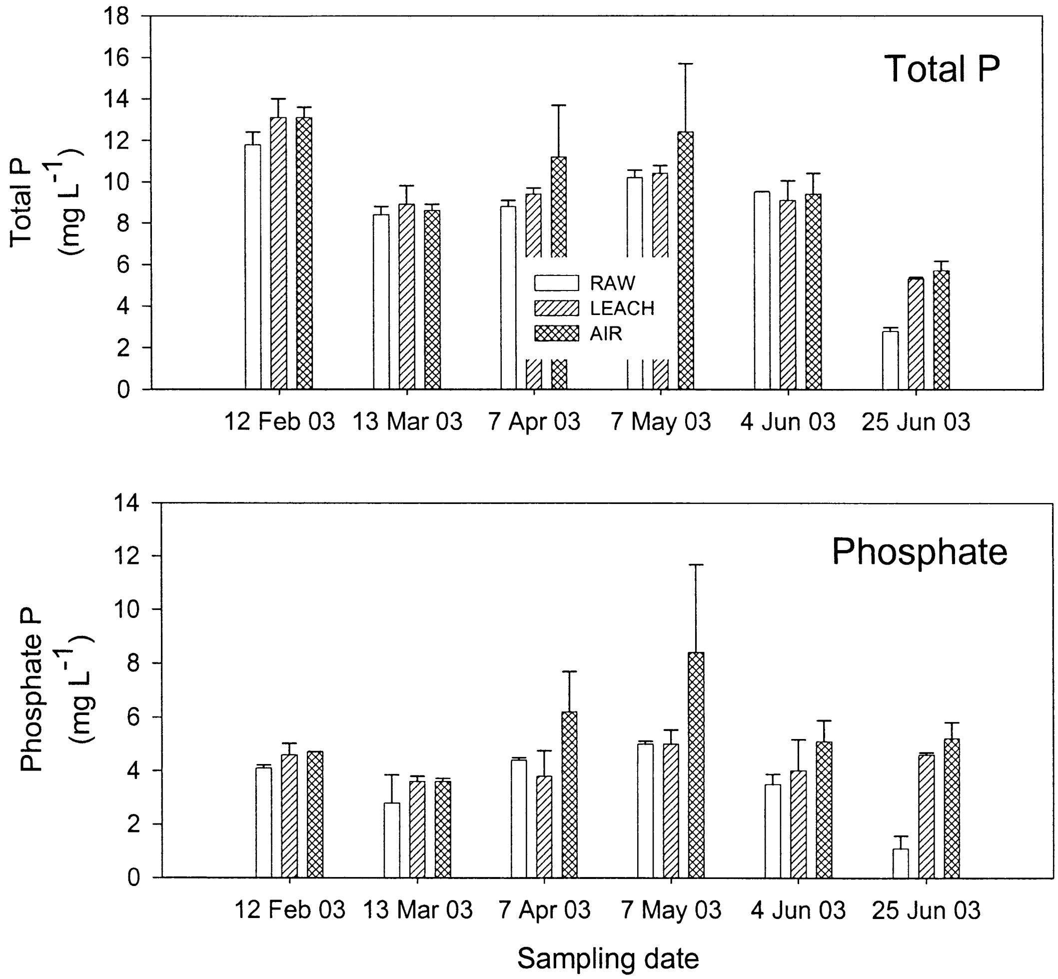

Fig. 5. Time course of total P and phosphate P in raw wastewater input and in drainage water from aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH). The wastewater loading rate was changed from 12 to 4 cm d^{-1} on 21 Apr. 2003. Values are means (n = 3). Bars represent one standard deviation. Significant differences between AIR and LEACH treatments (P < 0.05) for a particular sampling date are indicated with an asterisk (*).

nisms for removal of organic and inorganic phosphate in leachfield soil involve sorption and binding of the PO_4^{3-} to iron and aluminum oxides and oxyhydroxides (Robertson, 2003; Zanini et al., 1998). The sand used in the present study was manufactured from monocrystalline industrial quartz and did not contain appreciable amounts of these oxides. Oxides could have formed on the surface of sand particles originating from reduced forms of iron in incoming water. However, we did not detect Fe^{2+} in the wastewater input stream. Even if such mineral coatings did form, they did not appear to have an effect on P removal. The increase in total P levels in drainage water from both LEACH and AIR lysimeters relative to raw wastewater was unexpected (Table 2), although it has been reported by others (e.g., Gold et al., 1992). A large number of microorganisms has been shown to accumulate excess phosphate in the form of polyphosphate (Kulaev and Vagabov, 1983). Shifts in redox status and availability of carbon substrates can induce release of phosphate from polyphosphates (Kornberg, 1995). This process has been implicated in the release of phosphate from enhanced biological phosphorus removal processes in wastewater treatment plants (Carucci et al., 1999) and may account for our results.

Removal of Fecal Coliform Bacteria and *E. coli*

Fecal coliform bacteria removal was similar in both treatments, with a removal rate of 98 to nearly 100%, depending on loading rate. Drainage water from AIR lysimeters had levels of fecal coliform bacteria that were one to four orders of magnitude lower than LEACH lysimeters. Removal of fecal indicator bacteria in soil absorption fields is attributed to a number of factors, including straining, temperature, soil moisture, pH, organic matter, type of bacteria, and antagonistic microflora (Bitton and Harvey, 1992; Hagedorn et al., 1981). The more acidic conditions of AIR lysimeters may have contributed to the greater extent of total coliform removal, as suggested by others (Gold et al., 1992; Reddy et al., 1981; Reneau et al., 1989). We did not observe formation of a biomat in the AIR lysimeters, suggesting it may not be necessary for effective removal of pathogens in well-oxygenated soil absorption fields. Concerns with reduced treatment efficiency in systems that do

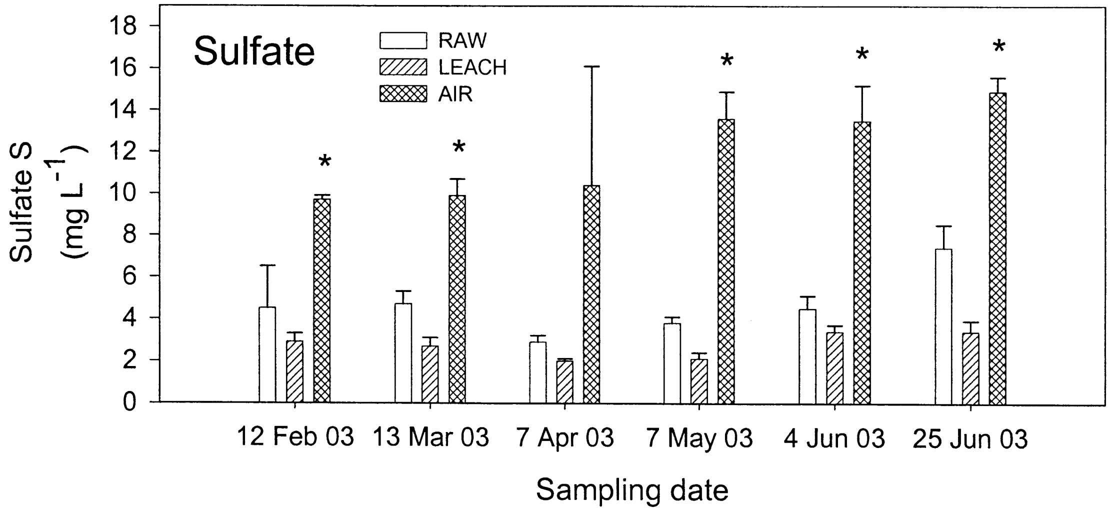

Fig. 6. Time course of sulfate in raw wastewater input and in drainage water from aerated lysimeters (AIR) and lysimeters vented to the leachfield (LEACH). The wastewater loading rate was changed from 12 to 4 cm d^{-1} on 21 Apr. 2003. Values are means ($n = 3$). Bars represent one standard deviation. Significant differences between AIR and LEACH treatments ($P < 0.05$) for a particular sampling date are indicated with an asterisk (*).

not develop a biomat (e.g., Postma et al., 1992; Tyler and Converse, 1994) may not be warranted in the present case, since aeration appears to enhance removal of fecal coliform bacteria.

Comparison with Previous Studies

Water quality parameters in LEACH lysimeters appear to differ from previous studies. For example, we observed consistently low dissolved O_2 levels in drainage water from LEACH lysimeters (Fig. 2), whereas it is generally thought that the soil beneath leachfields is relatively well oxygenated (USEPA, 2002) as a result of unsaturated flow below the biomat. We also observed that ammonium dominated the inorganic N pool (Fig. 3), whereas a number of studies have found that nitrate is the main form of inorganic N in the soil beneath a leachfield (e.g., Anderson et al., 1994; Bunnell et al., 1999), suggesting that nitrification is an active process in this zone. In addition, significant (30–40%) net loss of total N has been reported in leachfield soils (USEPA, 2002), whereas we observed no net N loss in LEACH lysimeters. There are a number of possible explanations for these discrepancies. The unstructured nature of the sand used in the present study could account for some of the differences. Soils generally exhibit some level of aggregation, not present in the sand used in our lysimeters, that affects the spatial distribution of pore sizes, water, and air, and leads to the formation of anaerobic microsites where denitrification may take place (Sexstone et al., 1985). In addition, determinations of nitrate concentration are often made at depths considerably greater than the 30 cm in this study (e.g., Stewart and Reneau, 1988). In other instances nitrate was measured in ground water receiving the leachfield water (e.g., Arravena et al., 1993). These situations are not comparable with our experimental setup, since they allow for the possibility of contact with oxic soil or water outside the area immediately below the leachfield. Nitrification could be taking place in oxygen-rich areas of the soil profile or in ground water. Nitrate could also have other sources, such as fertilizers or decomposing vegetation. Finally, laboratory experiments simulating leachfield conditions may not take into account the composition of the atmosphere above the leachfield (e.g., Van Cuyk et al., 2001), which our data indicate differs significantly from ambient air.

CONCLUSIONS

We found that aeration has a strong effect on the speciation of nitrogen, and enhances significantly the removal of nitrogen, BOD_5, and fecal coliforms and *E. coli* in leachfield lysimeters. Furthermore, this enhancement took place in the absence of a conventional biomat. These effects have implications for the functioning of conventional septic systems. Water managers and regulatory agencies are increasingly concerned with the effects of effluent from septic system leachfields on ground and surface water, especially in the case of failed or improperly constructed fields (USEPA, 2002). The high costs and unpredictable outcome of leachfield replacement makes this an economically unattractive alternative for most homeowners. Aeration may be an effective alternative to both prevent failure and rejuvenate septic system leachfields.

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