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Solute and Bacterial Transport through Partially-Saturated Intact Soil Blocks

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# Abstract

Steady-state transport of water, chloride and bacteria was measured through intact blocks of Maury and Cecil soils, under partially saturated conditions. Major objectives were to determine if transport occurs uniformly or via preferential flow paths, and if soil physical properties could be used to predict breakthrough. The blocks were instrumented with TDR probes and mounted on a vacuum chamber containing 100 cells that collected effluent. After each experiment the blocks were sampled for soil physical properties. The fluxes showed no spatial autocorrelation and the effluent variance was not statistically different between soils. Less than 3% of the influent bacteria appeared in the effluent. Maximum bacterial breakthrough occurred after 0.25 water-filled pore volumes had been leached, and was greater for Cecil soil than for Maury soil. The chloride breakthrough curves were fitted to the convection dispersion equation. The best predictor of dispersivity was volumetric water content ($R^2 = 0.28$, $P<0.01$), with dispersivity increasing with decreasing water content. Lower water contents lead to more tortuous flow paths and thus, a broadening of the velocity distribution. Soil structural controls on solute dispersion under partially saturated conditions are likely to be indirect, and related to differences in water content at given flux produced by differences in pore-size distribution.

Focus Categories: ST, NPP, AG

Keywords: Solute Transport, Contaminant Transport, Agriculture, Bacteria, Groundwater Quality, Leaching, Unsaturated Flow, Water Quality

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INTRODUCTION

Objectives

Agriculture is often perceived as a major non-point source of surface and groundwater pollution. Predicting transport processes within the unsaturated (vadose) zone underlying agricultural fields is essential to investigating agriculture's role in non-point source pollution. Agricultural chemicals, fertilizers, and manure applied to the land that are not taken up by plants, broken down, volatilized, or lost in runoff must pass through this zone to reach groundwater. However, current models to predict solute and bacterial transport processes within the vadose zone are unsatisfactory because they were developed without regard for soil structure.

Determination of transport properties generally requires empirical measurement of the transport process itself. This study combined measurements of both soil structural properties and transport processes in order to investigate methods of predicting transport properties independently. Solute and bacterial transport was measured through relatively large (approximately 1cubic foot) undisturbed soil blocks under unsaturated conditions. The main objectives of the study were to examine:

- if water, chloride (a non-interacting chemical), and bacteria would elute through partially-saturated, intact soil blocks uniformly or via preferential flow paths;
- if preferential paths were stable with time; and
- if tillage management practices affected either the speed or pattern of water flow through intact, unsaturated soil.

Additional objectives included the quantification of bacterial transport through the soil blocks and description of the displacement of adsorbed and/or entrapped bacteria by successive infiltration events. A secondary objective was to relate soil structural properties within the soil blocks to transport processes using fractal, statistical (pedotransfer functions) and geostatistical analyses. The fractal analyses were not performed because it proved impossible to collect water retention data over a sufficiently wide range of tensions to justify the use of fractal models to parameterize these data.

Previous Studies

Mass transfer of an input solution will be influenced by dispersion and adsorption phenomena. The dispersion of a non-reactive solute can be related to soil structural form and water content (Seyfried and Rao, 1987). The initial conditions are also an important determinant of the degree of dispersion that takes place (Kluitenberg and Horton, 1990). Poletika and Jury (1994) suggest two processes by which solutes and water move through structured soil:

(i) lateral movement and redistribution of the input solution on the soil surface so that some areas of the soil have higher than average solution flow moving through them;

(ii) movement through macropores.

Macroporosity, the fraction of soil volume occupied by large pores, is an important structural property in relation to solute and fecal pathogen transport. Macropores exist because of natural structure, old root channels, and insect and animal burrows. They can be defined as pores with equivalent cylindrical diameters (ECD's) $\geq$ 0.075-1.0 mm (Luxmore *et al.*, 1990). Macropores promote rapid, preferential water, solute and microorganism flow through the vadose zone.

Preferential flow can occur when macropores and/or the surrounding soil matrix are saturated or incompletely saturated. The hydraulics of different forms of preferential flow are discussed by McCoy *et al.* (1994). Flow through macropores can be considered part of a continuum of flow velocities that can be quantified using the classical convection-dispersion equation (Parker and van Genuchten, 1984). Several alternative modeling strategies, including multi-region models, transfer functions and percolation theory, have also been developed to predict macropore flow (McCoy *et al.*, 1994). However, none of these approaches currently permit the prediction of water and contaminant movement in the vadose zone from independently measured soil structural parameters.

Macropore or preferential flow has been documented in several laboratory and field studies. Patterns of localized macropore flow vary with antecedent moisture content, rainfall intensity, soil type and morphology, tillage practice, and earthworm activity (Adreini and Steenhuis, 1990; Edwards *et al.*, 1992; Granovsky *et al.*,1993; Quisenberry *et al.*, 1994; Shipitalo *et al.*, 1990;). A particularly useful approach is to collect percolate from water flowing through undisturbed blocks of soil in a grid of discrete sampling cells that allow for quantitative evaluation of flow variability. Recent studies examining macropore influences on solute transport through intact soil blocks indicate that flow paths are spatially variable (Quisenberry *et al.*, 1994; Wildenschild *et al.*, 1994). Based on leachate collection, only a few cells discharge flow at any time (Wildenschild *et al.*, 1994). For example, Quisenberry *et al.* (1994) found that over half of the total drainage occurred in just 12 to 19 % of collection cells. These observations can be interpreted as evidence of macropore or preferential flow (Quisenberry *et al.*, 1994) and are consistent with the concept of converging preferential flow paths. Dexter (1993) and

Wildenschild *et al.* (1994) have postulated that random, independent flow paths eventually converge with increasing depth into a few preferential flow channels.

There is little documentation to suggest whether preferential flow paths are static or dynamic. For any given leaching event, surface contaminants could elute from constantly changing sites in the soil profile. Flow path stability may be related to soil structural stability. The stability of a soil's structure is the ability of the soil to retain its arrangement of solid and void spaces when exposed to different stresses (Kay, 1990). Flowing water represents an applied stress. Swelling, slaking, and clay dispersion are hydraulically mediated physical processes that are associated with structural breakdown. Changes in structural form due to these processes can cause changes in hydraulic conductivity and the spatial distribution of flow paths over time (Reeve, 1953).

Tillage practices also influence soil structure and transport processes. Drees *et al.* (1994) studied the micro-morphological characteristics of conventional- and no-till soils, and observed differences in the size, shape, and arrangement of both pores and aggregates. Tillage destroys the natural pore structure of surface soils, disrupting macropore continuity, and reducing the extent of bypass flow. Increased infiltration rates in no-till fields have been attributed to the greater number and continuity of macropores in the surface layer compared to moldboard plowed soils (Edwards *et al.*, 1988; Dick *et al.*, 1989). Andreini and Steenhuis (1990) showed that the entire profile of undisturbed soil blocks can be short-circuited by preferential flow in no-till, while solutes must pass through a mixed, unstructured plow layer before bypass flow can occur in conventional-till. Crop residues associated with various conservation tillage practices can also effect the leaching of surface applied chemicals (Green *et al.*, 1995). Reduced macropore flow may promote increased adsorption of agricultural chemicals in soil. This could delay the onset of ground water contamination from individual storms. However, the extent to which tillage management affects contamination and flow once leachate passes through the tilled zone is poorly documented.

RESEARCH PROCEDURES

Field Site and Soil Block Extraction

Six, 32.5 cm square x 32.5 cm deep soil blocks were obtained from a Maury silt loam soil (fine, mixed, mesic Typic Paleudalf) at the University of Kentucky Experiment Station (Spindletop Farm). Four soil blocks were obtained from a Cecil soil (clayey, kaolinitic, thermic Typic Kanhapludult) from South Carolina. Selected soil properties for the Maury and Cecil soils are given in Table 1. Three replicate blocks for the Maury soil and two replicate blocks for the Cecil soil were studied per land use treatment. The land use treatments were:

(a) long-term conventional-till (disk) corn production; and

(b) long-term grass pasture.

The soil blocks were excavated, encased in polyurethane foam, and transported to the laboratory using the methods of Quisenberry et al. (1994). Our excavation methods were similar to those described by Bowman *et al.* (1994), Quisenberry *et al.* (1994), and Shipitalo *et al.* (1990). We encased the soil blocks in plywood on four sides and poured liquid polyurethane foam into the gap between the soil block and the wood casing. We let the polyurethane foam cure overnight, separated the soil blocks from the rest of the soil about 10 cm from the bottom of the casing, and transported the blocks to a temperature-controlled room for storage at 4°C. All blocks were covered in plastic to ensure that the soil would not dry, crack or pull away from the foam interface during storage. For the tilled soil, four-sided metal casings (32.5 cm x 32.5 cm x 17.0 cm height) were hammered into the soil to hold the tilled layer in place before carving the blocks. We excavated the blocks to a depth of 42.5 cm and removed them from the field as previously described.

Laboratory Apparatus and Procedures

Effluent Collection Chamber

We trimmed the bottom of each individual soil block flush with the wood casing, placed the block on a collection chamber, and caulked it with silicon to make an airtight and water-proof seal. The top of the collection chamber was a metal grid consisting of 144 cells in a 12 x 12 array

Table 1. Selected Soil Properties for the Untilled Soils

		Physical Property					
Soil	**Depth** (cm)	**Organic Matter** (%)	**pH**	**CEC** (meq/100g)	**Sand** (%)	**Silt** (%)	**Clay** (%)
Maury	0-15	5.6-7.7	5.0-5.2	10.0-15.2	8.1-22.1	60.3-69.4	17.6-22.5
Maury	15-30	2.5-2.6	5.6-5.8	7.1-14.7	6.0-8.2	66.5-68.0	25.3-26.0
Cecil	0-10	1.8-2.1	6.0-6.2	7.4-10.3	51.8-61.7	15.4-14.8	23.5-32.8
Cecil	10-20	1.4-2.1	5.8-6.4	5.9-8.8	51.1-64.8	15.6-16.0	19.6-32.9
Cecil	20-30	0.8-0.9	6.0-6.1	8.8-17.6	22.2-37.9	12.8-15.3	49.3-62.4

that collected water leaching from the block. The collection cells were 3.05 cm square and tapered to a 3 mm diameter drain hole at the bottom. Nylon screens (Nitex 53μm mesh) were placed in the bottom of each cell and the cells were filled with a saturated, 100 μm diameter glass bead phase barrier placed between the soil and the collection chamber. The metal ridges between each cell are intended to cut slightly into the bottom of the soil block and it was assumed that this prevents lateral flow between cells at the bottom of the soil block.

The outermost row of cells collected the outflow from the soil-foam interface. All results here are based on effluent collected from only the 100 innermost collection cells. Therefore, any possible effects of edge flow were minimized.

Plastic trays held 100 plastic centrifuge tubes (50 mL volume) beneath the drain holes of the collection cells to collect soil block drainage and the drainage from the outermost row of cells. The procedures for installing a soil block on this collection chamber are described in greater detail by Quisenberry *et al.* (1994).

A -2.0 kPa vacuum was applied to the lower boundary of each soil block via the collection chamber (Phillips *et al.*, 1995). This vacuum should have drained all pores greater than 0.15 mm in diameter based on the capillary equation (Danielson and Sutherland, 1986).

Rainfall Simulator

Simulated rainfall was applied to the top of each block at a target rate of 1 cm hr^{-1} (1056 mL h^{-1}). The simulator was a square reservoir, 32 x 32 x 5 cm, constructed of acrylic plastic 0.32 cm thick. It was positioned 20 cm above the soil block. One hundred hypodermic needles, 0.25 mm in diameter (25 gauge), were connected to the bottom of the applicator at positions corresponding with the centers of the 100 inner most collection cells of the bottom collection chamber (Quisenberry *et al.*, 1994). Rainfall was simulated by pumping leachate solution (0.003

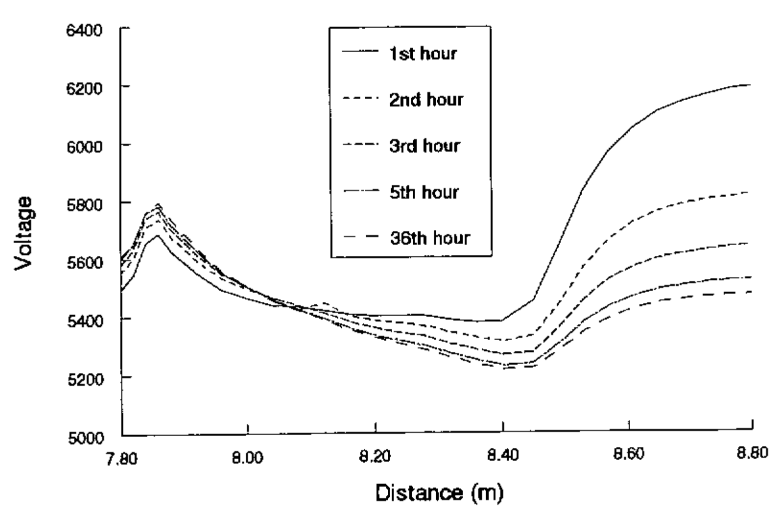
M $CaSO_4$) into the sealed reservoir and through the tip of each hypodermic needle with a peristaltic pump.

Time-Domain Reflectometry

Time-domain reflectometry (TDR) has recently been applied to monitor soil water contents and solute transport in structured soils. Using TDR, a step voltage change is propagated along a transmission line in a dielectric medium. The waveform of the step pulse, reproduced by means of a high-frequency oscilloscope, is then analyzed to estimate the dielectrical properties of the material along the transmission line.

Figure 1 shows typical TDR waveforms measured over time as a solute (chloride) moves into the soil under steady flow conditions. The travel time along the transmission line is proportional to the volumetric water content (θ), while the attenuation of the reflected signal, expressed in terms of impedance, is inversely proportional to the bulk electrical conductivity (BEC).

Figure 1. Typical TDR Waveforms for a Step Increase in Solute Concentration



By measuring the changes in BEC that occur over time in response to a step change in solute concentration, it is possible to obtain the solute BTC (Bowman et al., 1994; Kachanoski et al., 1992). The TDR method is rapid and non-destructive; it has the advantage that water content is measured simultaneously at the same location where the BTC is determined. Furthermore, hardware and software are readily available for automated TDR sampling over time at multiple locations and/or depths.

Each soil block was instrumented with 20-cm long TDR wave guides installed horizontally at the 5, 15 and 25 cm depths. The wave guides were connected to a Tektronix 1502C cable tester via a multiplexer. The cable tester was connected to a laptop computer using an RS-232 serial port. Data were collected automatically and the waveforms were analyzed for volumetric water content and bulk electrical conductivity using software developed by Wraith et al. (1993).

Soil Block Operations

The rainfall simulator was regulated to give a target water flux of 1 cm/hr. However, the measured fluxes varied from block to block. The measured fluxes for each block are given in Table 2. When a steady flow was achieved, the water supply was changed to 0.03 M KCl to produce a step increase in solute concentration. Each experiment lasted 36 hours. Fifteen trays of effluent were collected: 11 trays of 50 mL-tubes collecting effluent from individual drain holes alternated with 4 'bulk' trays (all leachate collected en mass). The bulk trays collected the drainage water at the end of hour's: 12, 20, 24, and 32; two subsamples were taken to determine the chloride concentration. The tube trays were changed at the end of each hour, except trays 5, 7, and 15, which were run for a four-hour period. The trays with tubes were checked regularly for cells with high flow and tubes were replaced once full.

Table 2. Water Fluxes for the Different Soil Blocks

Soil	Treatment	Rep	Flux (cm hr^{-1}) †
Maury	Tillage	1	0.907
Maury	Tillage	2	0.917
Maury	Tillage	3	0.940
Maury	Sod	1	0.715
Maury	Sod	2	0.650
Maury	Sod	3	0.913
Cecil	Tillage	1	0.79
Cecil	Tillage	2	0.76
Cecil	Sod	1	0.91
Cecil	Sod	2	0.76

† Averaged over 36 hours

The 50-mL tubes were weighed to determine leachate volume in each cell. Tubes that contained more than 5 mL of leachate were stored at 4°C and the Cl^- and fecal coliform concentrations were measured within 48 h. Subsamples were also taken from the input solution reservoir every time a tray was changed and were included with the Cl^- and bacterial analyses. The chloride concentration (mg/l) was measured using a Bio-Tech EL-311 microplate autoreader by the automated ferricyanide method (APHA, 1989). The relative chloride concentration (C/Co) in each tube was determined by dividing the concentration of the effluent chloride (C) by the concentration of the influent chloride (Co).

Two *Escherichia coli* strains (ATCC 27662 and 25254) were used as model pathogen species. The strains were labeled with antibiotic resistant markers to permit separate quantification. Each strain was added to the reservoir feeding the rain simulator at rates exceeding 10^6 CFU mL^{-1}. ATCC 27662 (nalidixic acid resistant) was added at the initiation of the experiment and ATCC 25254 (streptomycin resistant) was added 24 hours later to determine whether the newly introduced strain would displace bacteria adsorbed to the soil.

Data for the bacterial breakthrough curves were reported as C/Co, where C is the concentration of fecal coliforms in the effluent and Co represents the geometric mean of the input fecal coliform concentrations for each sampling time. The Co varied from block to block. The measured values of Co for the bacteria are summarized in Table 3. Breakthrough curves were only generated for the first bacterial strain added to the soil block. The sampling interval for the first four observations following each fecal coliform step change was every hour. For subsequent observations, the samples were at 4 hour intervals

Fecal bacteria were enumerated by counting the colony-forming units that developed after filtering known volumes of drainage through sterile 0.45 μm membrane filters followed by incubation for 24 h at 44.5°C on mFC agar plates containing a suitable antibiotic (Streptomycin for ATCC 25254 and Nalidixic acid for ATCC 27662). All blue and dark blue colonies were counted as fecal coliforms (Howell *et. al.*, 1995).

Table 3. Influent Concentrations of Fecal Coliforms for the Different Soil Blocks

Soil	**Treatment**	**Rep**	**Bacteria** CFU/ml
Maury	Tillage	1	147,263
Maury	Tillage	2	711,267
Maury	Tillage	3	1,422,612
Maury	Sod	1	3,747,633
Maury	Sod	2	70,402
Maury	Sod	3	2,719,522
Cecil	Tillage	1	8,053,160
Cecil	Tillage	2	1,089,644
Cecil	Sod	1	9,741,041
Cecil	Sod	2	2,432,801

Parameterization of Chloride Breakthrough Curves

The objectives of this procedure were to measure BTC's for a non-reactive solute (chloride) in large undisturbed soil blocks using TDR, to parameterize these BTC's using the convection-dispersion equation (CDE), and to investigate the effects of land use, soil properties, and depth on the resulting parameters.

Once the BTC data were obtained, they were parameterized using the CDE. For the transport of chloride, the principal parameters of the CDE are the dispersion coefficient (D) and the average pore water velocity (v). The D characterizes the spread of solute velocities. These parameters are often combined to give the dispersivity (α), defined as $\alpha=D/v$. The dispersivity is a property of the porous medium and its state – especially its water content for unsaturated systems.

In our case, the flux (q) of water through the soil blocks was known from effluent volume measurements and the volumetric water content (θ) was known from the TDR measurements. The average pore water velocity is the flux divided by the water content ($v = q/\theta$). Thus, the dispersion coefficient is the only unknown parameter in the CDE.

The CXTFIT solute transport model (Parker and van Genuchten, 1984; Toride et. al., 1995) was used to estimate the solute dispersion characteristics of the soil blocks. CXTFIT has an inverse capability that can estimate the dispersion coefficient (and other transport parameters when necessary) from the observed results of a solute transport experiment. The program uses a non-linear least squares fitting procedure to optimize the fit between the observed data and the convection dispersion equation.

A principal limitation to the use of CXTFIT for the analysis of the soil block data is that the soils are known to be layered (Table 1). CXTFIT does not consider multi-layered soils. Other models can simulate solute transport in multi-layered systems, and could be used inversely. However, it is likely that non-unique parameter estimates would be obtained from such a procedure. Because of this layering, the estimates of dispersivity for the 15 and 25 cm depths in soil blocks may be biased by the transport properties of the overlying horizons.

For the analysis of the data collected during this project, concentrations as a function of time and depth determined from TDR measurements were used as input to the CXTFIT model. Additional analyses were carried out on the soil block effluent. The solute dispersion coefficient was the only transport parameter estimated. Coefficients of determination (R^2) for the fits were generally greater than 0.9.

For the Maury soil blocks, soil BEC as measured by the TDR probes was normalized with respect to the maximum BEC. This is appropriate when the infiltrating solution completely displaces all solution previously held in the porous medium, at least to the depth of the TDR probe.

For the Cecil soil, breakthrough was incomplete and raw BEC values were used in the model. The maximum BEC and the dispersion coefficient were both estimated in this case. Numerous CXTFIT runs with different estimated maximum BEC values were made. The best estimate of the maximum BEC was selected from the run yielding the largest R^2 value for the fit.

Soil Water Retention Measurements

After each experiment, the soil blocks were drained overnight, removed from the collection chamber and triplicate soil cores were taken at the 5, 15, and 25-cm depth. These cores were used to determine the bulk density and the soil water retention (drying) curve (Klute, 1986). Tempe cells were used to collect the retention data between tensions of 0 and 3 kPa, and a pressure plate apparatus was used to collect the corresponding data between 3 and 1500 kPa (Klute, 1986). Total porosity was calculated from the bulk density data assuming a particle density of 2.65 g/cm^3. The soil water retention curves were parameterized using the empirical model developed by Campbell (1974). The estimated parameters in this model are ψ_a, the air entry value, and b, the pore-size distribution index. Coefficients of determination (R^2) for the fits ranged from 0.77 to 0.99.

Saturated Hydraulic Conductivity Measurements

The cores were also used to determine saturated hydraulic conductivity (K_{sat}). These measurements were carried out using the constant head method, as described in Klute and Dirksen (1986).

RESULTS AND DISCUSSION

Soil Physical Properties

Bulk Density

The soil bulk density data are summarized in Table 4. Analysis of variance of these data revealed significant land use, depth and land use by depth interaction effects for the Maury soil.

Table 4. Soil Bulk Densities for the Different Soil Blocks as a Function of Depth

Soil	Treatment	Rep	Bulk Density (g/cm^3) † (depth, cm) 5	15	25
Maury	Tillage	1	1.04	1.30	1.27
Maury	Tillage	2	1.00	1.35	1.44
Maury	Tillage	3	1.01	1.27	1.34
Maury	Sod	1	1.17	1.38	1.28
Maury	Sod	2	1.20	1.34	1.41
Maury	Sod	3	1.15	1.32	1.32
Cecil	Tillage	1	1.34	1.58	1.33
Cecil	Tillage	2	1.43	1.68	1.67
Cecil	Sod	1	1.19	1.63	1.20
Cecil	Sod	2	1.57	1.60	1.47

† Means of 3 subsamples

The ANOVA R^2 value was 0.75 and the coefficient of variation was 6.0%. This analysis showed that the surface bulk density in the tilled soil was significantly ($P<0.05$) lower than that for the sod-covered treatment, with little or no differences between the treatments at the other depths. This result is probably attributable to soil loosening caused by the tillage operations that were performed before sampling.

In contrast, a similar analysis of variance for the Cecil soil revealed no effect of land use treatment, only an effect of depth, with the bulk density at 15 cm significantly ($P<0.05$) higher than at either 5 or 25 cm. The R^2 value for this analysis was 0.39, and the coefficient of variation was 10.8%.

Soil Water Retention

The parameters from fitting the Campbell (1974) equation to the soil water retention data, ψ_a and b, are summarized in Table 5. The estimates of ψ_a were highly variable, and analysis of variance

Table 5. Water Retention Parameters for the Different Blocks as a Function of Depth

Soil	Treatment	Rep	ψ (kPa)†			b†		
			(depth, cm)					
			5	15	25	5	15	25
Maury	Tillage	1	0.06	0.21	0.19	8.66	12.44	11.46
Maury	Tillage	2	0.06	0.54	1.66	8.92	11.86	14.02
Maury	Tillage	3	0.11	0.32	1.60	8.79	12.40	12.70
Maury	Sod	1	0.16	0.71	0.26	14.52	13.74	10.23
Maury	Sod	2	0.72	0.53	0.88	12.04	12.59	15.5
Maury	Sod	3	0.12	0.39	0.37	14.22	13.69	12.31
Cecil	Tillage	1	0.47	0.39	0.41	5.45	10.01	15.75
Cecil	Tillage	2	0.37	0.17	2.13	5.13	8.25	18.08
Cecil	Sod	1	0.08	0.77	0.28	7.01	13.10	12.36
Cecil	Sod	2	0.09	0.08	0.29	9.42	7.76	17.47

† Means of 3 subsamples

was only able to explain 27% of the total variation. The only significant effects were depth and the land use by depth interaction; both effects were independent of soil type. These effects indicate that the air entry value greatly increased with increasing depth for the tilled blocks, but increased only slightly with increasing depth for the sod covered blocks (Table 5). This trend may be due to loosening of the surface soil and/or subsurface compaction (i.e. a plow pan) in the tilled blocks as compared to the sod covered blocks.

In contrast to ψ_a , the b parameter was much more predictable. The b parameter is an indicator of pore size distribution; for large values of b, small pores dominate the total porosity, and vice versa. Soil, land use, depth, soil by depth interaction and the land use by depth interaction were all significant factors influencing b at $P<0.05$. The R^2 for this analysis of variance was 0.60 and the coefficient of variation was 21%. For the Maury soil, the b value increased with increasing depth for the tilled soil, but was relatively constant with depth for the sod treatment (Table 5). For the Cecil soil, b increased with increasing depth in both treatments. However, this increase was more pronounced in the tilled blocks as compared to the sod blocks (Table 5).

Saturated Hydraulic Conductivity

Saturated hydraulic conductivity (K_{sat}) data were collected at each depth for sod block # 3 and tilled block #3 of the Maury soil and for the four Cecil soil blocks (Table 6). Because of the large number of missing values it was not possible to conduct an analysis of variance on these data. However, the data for the Maury soil indicate that the tilled block had higher surface K_{sat} values than the sod covered soil. The land use effect was much less pronounced for the Cecil soil, which generally appeared to have decreasing K_{sat} values with increasing depth regardless of management history. This can probably be attributed to the marked increase in clay content with depth for this soil (Table 1).

Table 6. Saturated Hydraulic Conductivities for the Different Soil Blocks

Maury Soil	K_{Sat} (m s^{-1})	Cecil Soil	K_{Sat} (m s^{-1})
Sod #1		Sod #1	
5 cm	X	5 cm	0.21×10^{-2}
15 cm	X	15 cm	0.50×10^{-3}
25 cm	X	25 cm	0.13×10^{-2}
Sod #2		Sod #2	
5 cm	X	5 cm	0.34×10^{-2}
15 cm	X	15 cm	0.24×10^{-2}
25 cm	X	25 cm	X
Sod #3		Till #1	
5 cm	0.27×10^{-3}	5 cm	0.13×10^{-2}
15 cm	0.88×10^{-3}	15 cm	0.25×10^{-2}
25 cm	0.80×10^{-4}	25 cm	0.16×10^{-2}
Till #1		Till #2	
5 cm	X	5 cm	0.48×10^{-2}
15 cm	X	15 cm	0.29×10^{-2}
25 cm	X	25 cm	0.36×10^{-3}
Till #2			
5 cm	X		
15 cm	X		
25 cm	X		
Till #3			
5 cm	0.76×10^{-2}		
15 cm	0.28×10^{-2}		
25 cm	0.80×10^{-3}		

X=not measured

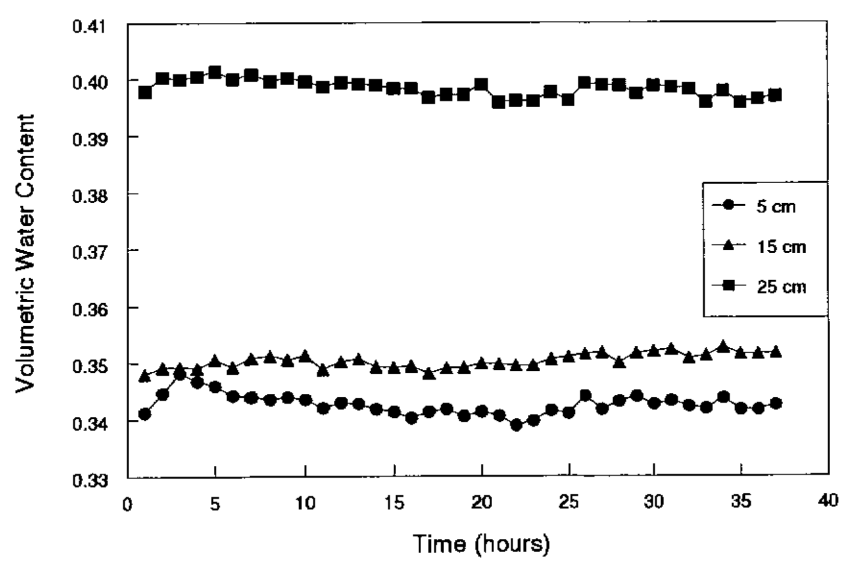
Volumetric Water Contents

Volumetric water contents (θ) within the soil blocks, as measured by the TDR system, were extremely stable over the course of the 36-hour experimental period (Figure 2). Mean values ranged from 0.27 to 0.51 for the Maury blocks and from 0.27 to 0.50 for the Cecil blocks (Table 7. Analysis of variance indicated significant depth, land use, and depth by land use interaction

Table 7. Volumetric Water Contents for the Different Blocks as a Function of Depth

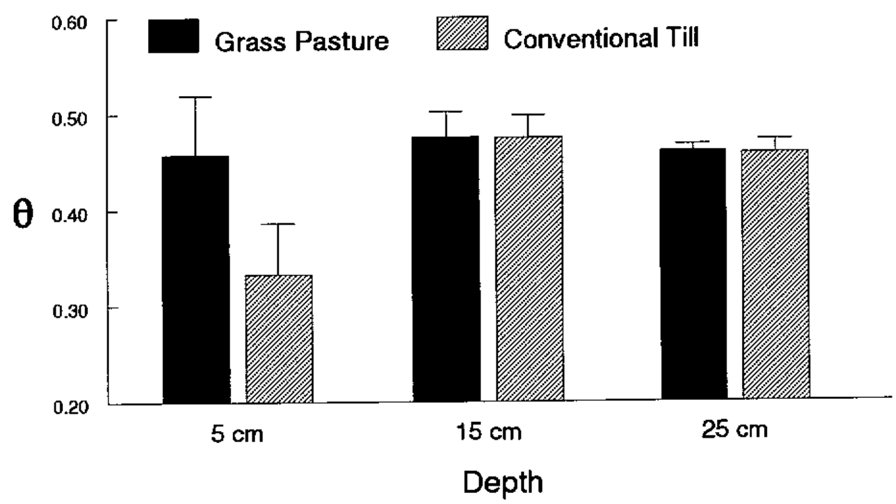
Soil	Treatment	Rep	Volumetric Water Content ($m^3 m^{-3}$) (depth, cm)		
			5	15	25
Maury	Tillage	1	0.27	0.48	0.47
Maury	Tillage	2	0.35	0.50	0.46
Maury	Tillage	3	0.34	0.45	0.44
Maury	Sod	1	0.47	0.45	0.45
Maury	Sod	2	0.39	0.48	0.46
Maury	Sod	3	0.51	0.50	0.46
Cecil	Tillage	1	0.31	0.35	0.39
Cecil	Tillage	2	0.27	0.25	0.27
Cecil	Sod	1	0.34	0.35	0.40
Cecil	Sod	2	0.27	0.27	0.33

Figure 2. Volumetric Water Content as a Function of Time for Cecil Sod Block #1



effects for the Maury soil, with the mean θ of the conventionally-tilled soil 1/4 less than the mean of the sod-covered soil at the 5 cm depth. There were no differences in the mean θ at the other depths. These results are illustrated in Figure 3, and indicate a difference in the water

Figure 3. Mean Volumetric Water Contents for the Different Treatments: Maury Soil



retention properties of the two land use treatments close to the soil surface.

Overall, the Cecil soil blocks had lower volumetric water contents than the Maury soil blocks (Table 7). Generally lower water fluxes in these blocks (Table 2) and differences in water retention characteristics (Table 5) are likely to be responsible for this observation. Differences in θ between the sod and tilled treatments were less pronounced for the Cecil soil (Table 7). For this soil, the main effect was the higher mean value of θ for the 25 cm depth as compared to the 5 and 15 cm depths (Figure 2). This corresponds with the increase in clay content with depth observed for this soil (Table 1).

Water Fluxes

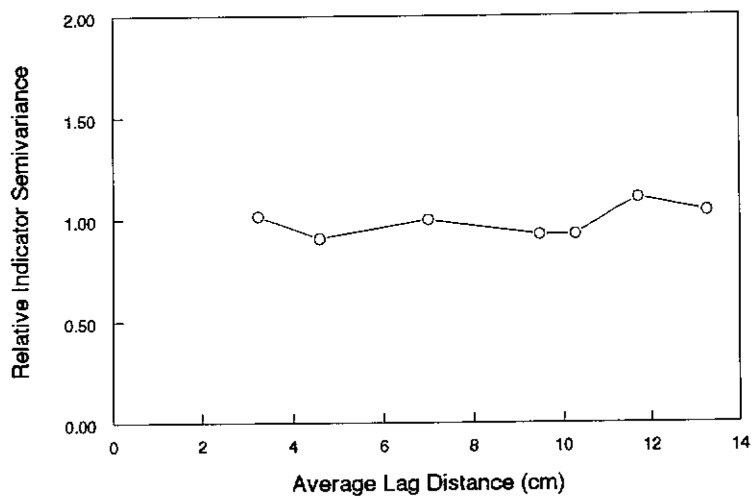
The 'whole block' water fluxes (cm/hr) were determined by adding the total drainage for the 11 'tube' trays and 4 'bulk' trays (cm^3/hr) and dividing by the total area of the block (1056.25 cm^2). The 36-hour average fluxes for the different blocks are given in Table 2.

The collected leachate in each cell was also converted into a flux by dividing the flow rate for each cell by the area of that cell (9.61 cm^2) for each of the 11 tube trays. Since the target flux was the same for each block, it was hypothesized that any differences in flow due to differences in pore-size distribution and continuity would be manifested in the variance of the individual fluxes. However, the actual fluxes deviated from the target fluxes, and these differences may have also influenced the magnitude of the variance.

A geostatistical analysis was performed by computing indicator variograms (1=flowing, 0=non-flowing) for the individual fluxes for each block. However, the results indicated little or no spatial structure in the data (Figure 4). Therefore, the individual flux data were analyzed using conventional statistical procedures.

An analysis of variance was completed on the mean and variance of the individual fluxes between the treatments on only those cells that were involved in the collection of drainage water. The results indicated that there were no significant differences ($P < 0.05$) between the two treatments with respect to the percent of cells collecting leachate, the mean flux, or the variance of the fluxes. Descriptive statistics resulting from this analysis are given in Table 8. For the three Maury sod blocks, an average of 33% of cells collected leachate with a mean flux of

Figure 4. A Typical Indicator Variogram for the Spatial Distribution of Effluent Flux



2.67cm hr^{-1} and a variance of 2.77 cm^2 hr^{-2}. The Maury tilled blocks averaged 38% of cells collecting leachate, with a mean flux of 3.48 cm hr^{-1} and a variance of 5.52 cm^2/hr^2. These results indicate that the sod blocks had a slightly more concentrated flow pattern than the tilled blocks. However, the Maury tilled blocks exhibited much greater spatial variability in the magnitudes of the fluxes for those cells that were flowing.

For the Cecil tilled blocks, the mean percentage of cells collecting drainage water was 40% and the mean flux was 2.23 cm hr^{-1} (Table 8). For the sod blocks, the mean percentage of cells was 50% with a slightly lower mean flux of 1.96 cm hr^{-1}. The means of the variances for the two treatments were identical at 3.55 cm^2 hr^{-2}.

Correlations were performed for each block between the spatial distribution of fluxes in tray 1 (hour 1) and the distribution in tray 15 (hour 36). The comparison of these two trays provides evidence of structural (flow pattern) stability from the start of the experiment to completion.

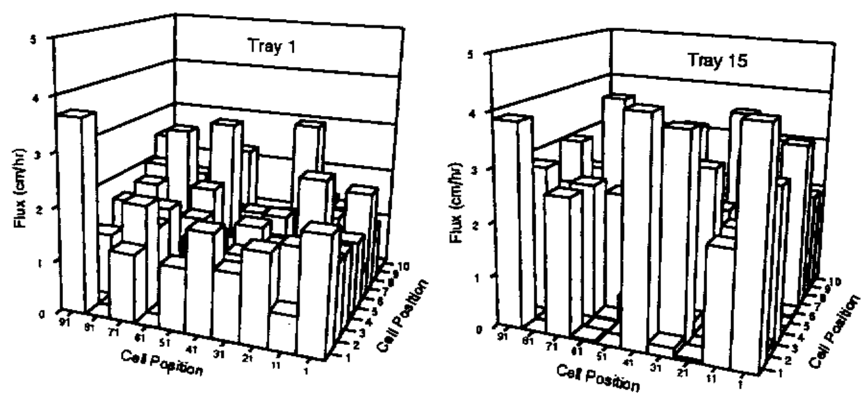
Table 8. Descriptive Statistics for the Individual Cell Fluxes

Soil	Treatment	Mean	Mean Variance	Mean Cell Count	Mean R†
		(cm hr^{-1})	(cm^2 hr^2)		
Maury	Sod	2.67	2.77	33	0.73
Maury	Tilled	3.48	5.52	38	0.65
Cecil	Sod	1.96	3.55	50	0.61
Cecil	Tilled	2.23	3.55	40	0.44

† Correlation between tray 1 and tray 15

Although the mean correlation coefficients (r values) were not significantly different at $P<0.05$ between the two Maury treatments, the tilled blocks had a lower mean r value than the sod blocks (Table 8). The higher r value for the sod blocks indicates a more stable flow pattern from start to finish than for the tilled blocks. Tilled block #3 is an extreme example of the variability in the water conducting cells that occurred throughout the 36 hour period. For tray 1 the mean flow rate was 1.25 cm hr^{-1} with 75% of the cells collecting leachate. Tray 15 had a mean flow rate of 1.40 cm hr^{-1} with 71% cells collecting leachate. Even though these values are similar, the r- value between the two trays was only 0.36, indicating a fluctuating flow rate for the individual cells (Figure 5). These fluctuations may be related to soil structural stability. A correlation analysis was also completed between trays 1 and 15 for each block of the Cecil soil (Table 8). The lower mean r values for this soil, relative to the Maury soil, may indicate a lower aggregate stability for Cecil as compared to Maury soil. Although the correlation coefficients for the two land use treatments were not significantly different ($P<0.05$), as with the Maury soil, the Cecil tilled blocks had a lower mean r-value, 0.44, than the Cecil sod blocks at 0.61. The higher mean r-value for the sod block shows a more steady and even flow rate from start to finish, while the lower mean r- value for the tilled blocks indicates a more temporally variable flow pattern through the system.

Figure 5. Temporal Changes in the Spatial Distribution of Fluxes (Maury Till Block #3)



Tillage affects soil aggregation; in the absence of a protective cover, like sod, aggregates are vulnerable to disruption by flowing water (Hillel, 1980). As aggregates collapse and flow lines change, the fluxes for some cells might change over time while the overall flow rate for the block remains fairly constant from tray 1 and tray 15. The sod-covered soil probably had an accumulation of organic matter, so that aggregates in this treatment should be more stable than those in the tilled soil. Greater aggregate stability would be expected to result in more stable flow patterns over time, as indicted by the higher r-values for the sod versus tilled blocks (Table 8).

Effluent Breakthrough Curves

The effluent breakthrough curves are generated from measurements of the soil block effluent over time. The effluent breakthrough curves represent the flow-weighted average chloride concentrations. These curves have some advantages and limitations relative to the breakthrough curves obtained using TDR methods.

One principal advantage is that the effluent concentrations are normalized with respect to a well-known input solution concentration. In contrast, the TDR bulk electrical conductivity data are typically normalized with respect to their own late-time values. This means that complete breakthrough is important to the success of the standard TDR method but not as critical to the effluent method. However, TDR data can generally be collected more frequently than effluent concentration data and by automated methods.

Figure 6 shows the mean Cl- breakthrough curves for the Maury soil blocks. Both axes represent dimensionless numbers. Vo is the water-filled pore volume and V/Vo is the number of water-filled pore volumes flushed through the soil. All of the Maury soil block breakthrough curves were sigmoidally shaped.

In contrast to the Maury breakthrough curves, the Cecil breakthrough curves are not sigmoidally shaped (Figure 7). All four Cecil soil blocks showed detectable Cl^- after the first hour . The data represented in Figures 6 and 7 were fitted to the convection-dispersion equation with CXTFIT (Table 9). The dimensionless column Peclet number, defined as $P = L/\alpha$, where L is the length of the block from surface to base, is also listed. Clearly, the dispersivities are significantly larger (and the Peclet number correspondingly smaller) for the Cecil soil blocks as compared to the Maury blocks. An analysis of variance suggested that soil management was not a significant source of variability among the block results.

Figure 6. Effluent Chloride Breakthrough Curves for the Maury Soil

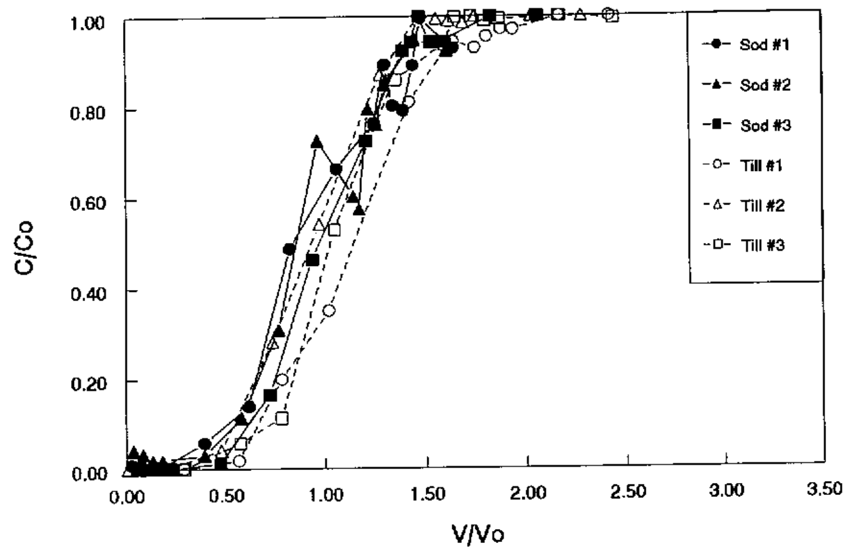
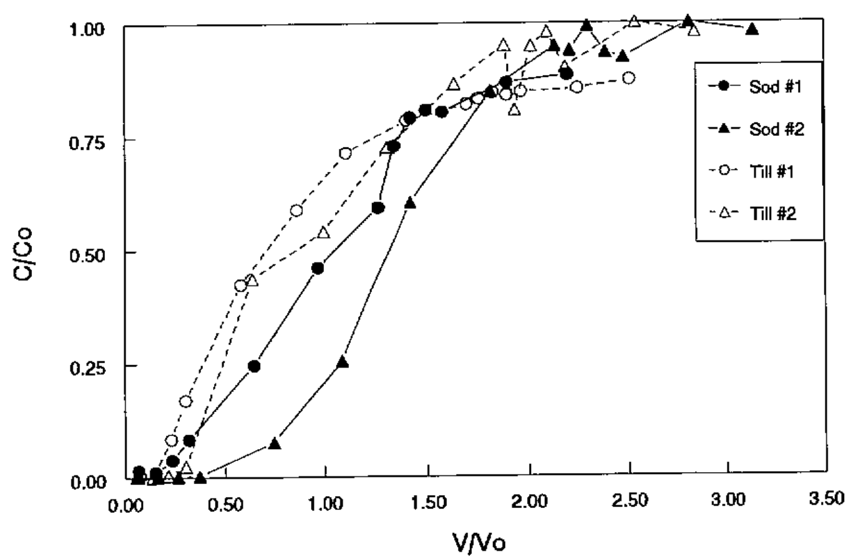


Figure 7. Effluent Breakthrough Curves for the Cecil Soil



As the dispersivity increases (and the Peclet number decreases), the step input breakthrough curves undergo a transition away from the typical sigmoidal shape (van Genuchten and Wierenga, 1986). High dispersivity corresponds to a broad range of flow velocities and can be indicative of preferential flow. The soil block results presented in Table 9 suggest that the Cecil soil was much more susceptible to preferential flow than the Maury soil under the conditions of these experiments.

TDR Breakthrough Curves

Calculation of the TDR breakthrough curves for the Maury soil blocks was done by normalizing the bulk electrical conductivity (BEC) values from the TDR probes relative to the smallest and largest values, i.e.

$$C/Co = \frac{C - C_{initial}}{C_{max} - C_{initial}}$$

where C is the BEC measured at any time, $C_{initial}$ is the BEC prior to application of the KCl solution, and C_{max} is the maximum BEC value, which is typically attained when the KCl solution breakthrough is complete.

The Cecil blocks did not reach complete breakthrough in some instances and, as discussed in Chapter 2, a different procedure, in which the maximum BEC was estimated, was used. This procedure was applied to all Cecil TDR data.

Because the TDR probes simultaneously give the BEC and the volumetric water content, it is possible to estimate the average pore water velocity v in the vicinity of a probe when the water flux through the column is known. Since there is a breakthrough curve and a velocity for each

Table 9. Dispersivities and Column Peclet Numbers from Effluent Breakthrough Curves

Soil	Treatment	Rep	Dispersivity (cm)	Column Peclet Number
Maury	Tillage	1	1.74	19
Maury	Tillage	2	0.90	36
Maury	Tillage	3	1.11	29
Maury	Sod	1	1.93	17
Maury	Sod	2	2.26	14
Maury	Sod	3	1.11	29
Cecil	Tillage	1	9.53	3
Cecil	Tillage	2	5.73	6
Cecil	Sod	1	7.95	4
Cecil	Sod	2	3.07	11

layer represented by a TDR probe, separate estimates of dispersivity in each layer are possible. However, as discussed in Chapter 2, CXTFIT assumes a uniform vertical soil profile. The known layering of the soils violates this assumption. This violation is likely to be more problematic for the analysis of these layer-specific TDR data than it is for the preceding effluent data evaluation. A multi-layer model could have been used inversely to evaluate these data, but it is possible that non-unique parameter estimates would have be obtained from such a procedure.

Figure 8 shows typical normalized breakthrough curves measured with the TDR probes as a function of depth for the Maury and Cecil soils. Near the source of the high-conductivity solution, the BEC measured at the 5-cm TDR probe is the first to increase after the step increase in applied water conductivity. Subsequent increases are observed at the 15- and 25-cm deep probes. The curves for both soils take on a more sigmoid shape as the solute moves farther from the source and deeper into the soil block.

Figure 8. Typical Normalized TDR Breakthrough Curves for Maury Soil (Sod Block #1)

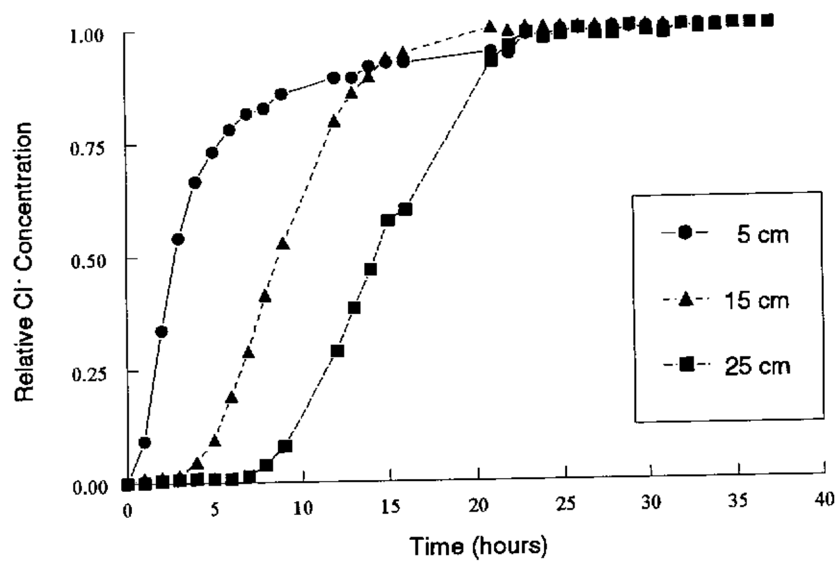


Table 10 presents the results of the CXTFIT parameter estimation procedure for the TDR data. Dispersivities ranged from just above 1 to nearly 90 cm. The highest value was more than four times larger than the next highest value and was excluded from all subsequent analyses as an outlier. Confidence in this estimate (Cecil tilled block #1, 15 cm depth) was low because of the high dispersivity in the uppermost 5 cm. (Table 10) This condition may have violated CXTFIT's non-layered assumption to such an extent that the resulting estimate of dispersivity for the 15 cm depth was unreasonable.

The mean dispersivity was higher for the Cecil blocks (Table 10). The Maury soil blocks generally showed a decrease in dispersivity with increasing depth. No consistent trend with depth was apparent from the Cecil block results. The Peclet numbers should increase with depth in a

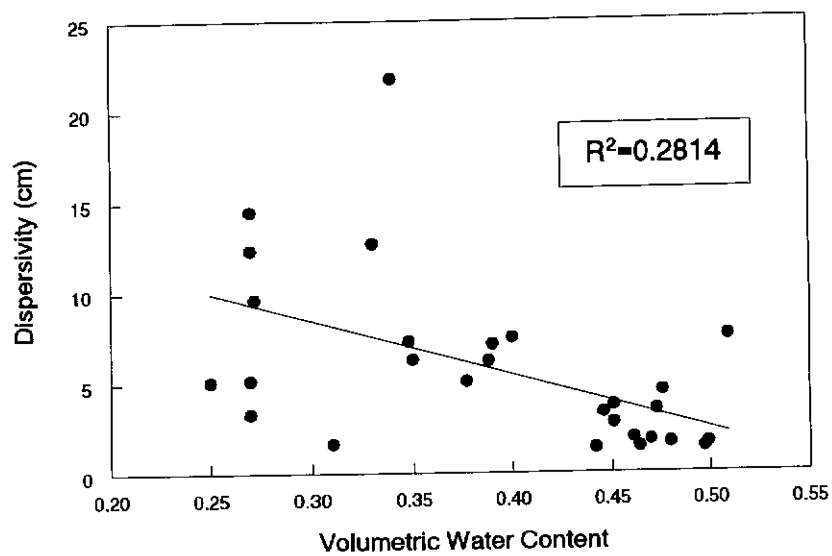
Table 10. Dispersivities and Column Peclet Numbers from TDR Probe Data

Soil	Treatment	Block	Depth (cm)	Dispersivity (cm)	Column Peclet Number
Maury	Sod	1	5	3.50	1.43
			15	3.36	4.47
			25	2.77	9.04
	Sod	2	5	6.23	0.80
			15	1.66	9.04
			25	1.43	17.50
	Sod	3	5	7.56	0.66
			15	1.66	9.05
			25	1.43	17.44
Maury	Tilled	1	5	9.63	0.52
			15	4.54	3.31
			25	1.80	13.91
	Tilled	2	5	7.31	0.68
			15	1.39	10.76
			25	1.93	12.97
	Tilled	3	5	5.10	0.98
			15	3.74	4.01
			25	1.37	18.23
Cecil	Sod	1	5	1.65	3.03
			15	6.30	2.38
			25	7.14	3.50
	Sod	2	5	14.49	0.35
			15	5.09	2.95
			25	5.18	4.83
Cecil	Tilled	1	5	21.80	0.23
			15	86.40	0.17
			25	7.50	3.33
	Tilled	2	5	3.32	1.50
			15	12.35	1.21
			25	12.71	1.97

uniform soil. That was consistently the case for the Maury blocks, suggesting that they were relatively uniform with depth. In contrast, there was no Peclet number trend with depth for the Cecil blocks. This result suggests that the effects of layering on solute transport in this soil were appreciable.

Regression analysis revealed a significant ($R^2 = 0.28$, $P<0.01$) effect of volumetric water content on dispersivity, with dispersivity decreasing with increasing water content (Figure 9). Lower

Figure 9. Effect of Volumetric Water Content on Dispersivity for TDR Data



water contents lead to more tortuous flow paths and thus, a broadening of the velocity distribution.

Dispersivity was also significantly ($R^2 = 0.26$, $P<0.01$) correlated with the soil water retention parameter b (Table 5). Dispersivity increases as the b parameter decreases (i.e. as the pore size distribution shifts to a predominance of larger pores). This change in pore size distribution implies the possibility of a broader range of solute velocities and therefore a larger dispersivity. However, the correlation between water content and the b parameter may be largely responsible for the relationship between b and dispersivity.

We conclude that any structural effects on solute dispersion in unsaturated soils subjected to initial and boundary conditions similar to those in our experiments will be indirect, and can be related to the differences in water content at given flow rate produced by differences in pore-size distribution. It is also possible that lower water fluxes in the Cecil soil contributed to lower water contents and higher dispersivities.

The effects of management on solute dispersion (and hence on preferential flow) were not appreciable under the conditions of these experiments. At a steady rainfall rate of 1 cm/hr, unsaturated conditions were maintained throughout the blocks and no runoff was generated. Under saturated conditions, flow in macropores would be expected to significantly increase solute dispersion. Under field conditions, flow of surface runoff water into macropores could also greatly enhance solute dispersion even when the soil remains unsaturated.

Fecal Coliforms

The added fecal coliforms behaved similarly in the Cecil and Maury soils. Within four hours the maximum C/C_0 was obtained in both soils (Figures 10 and 11). Thereafter, the value for C/C_0 remained constant or declined slightly for the duration of the experiment. This pattern is consistent with the results of Smith *et al.* (1985) for the transport of *E. coli* via preferential flow through macropores. In terms of pore volumes, the maximum C/C_0 was reached in both soils after approximately 0.25 pore volumes had leached (Figures 10 and 11).

In the Cecil soil, tilled soil blocks had slightly higher C/C_0 values than sod-covered soil blocks. The opposite was true in two of the three Maury soil blocks; sod-covered soils had a higher C/C_0 than the tilled soils. The higher the C/C_0, the greater the fecal coliform movement through soil. Previous research by McMurry *et al.* (1998) in surface applied poultry litter suggested that sod-covered Maury soil promoted greater fecal coliform movement due to greater continuity of soil pores. The Cecil soil clearly did not follow such a pattern in these experiments.

Figure 10. Bacterial Breakthrough Curves for the Maury Soil

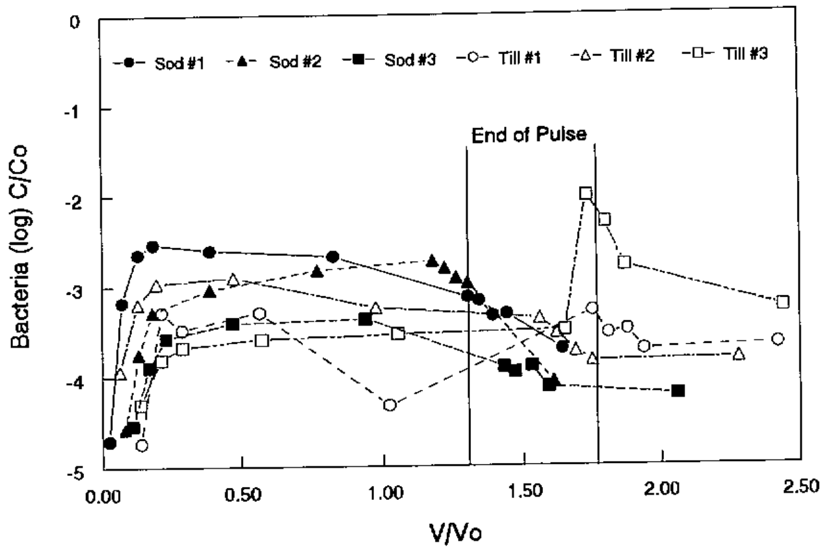
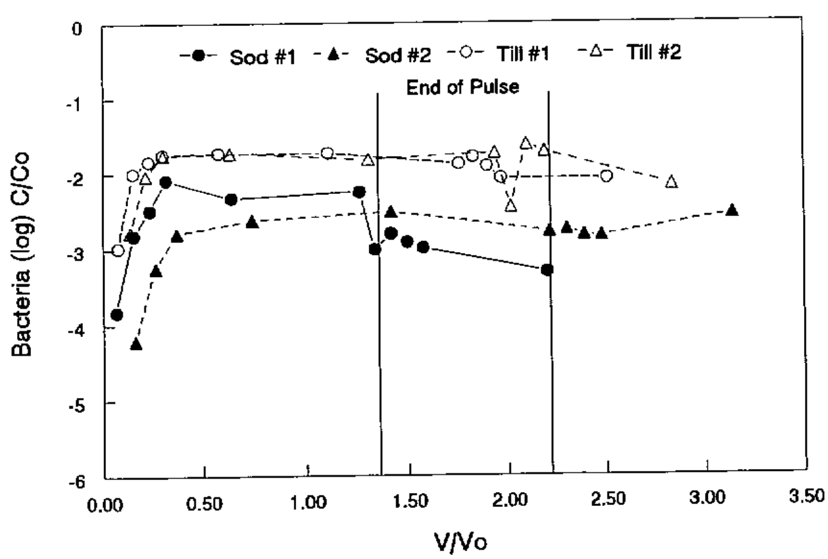


Figure 11. Bacterial Breakthrough Curves for the Cecil Soil



The magnitude of the C/C_0 in Cecil soil was greater than in Maury soil. In tilled treatments effluent fecal coliform concentrations in Cecil soil ranged from 0.6 to 3.4% of influent concentrations. In Maury soil the corresponding range was 0.03 to 0.12%. In sod-covered soils, the influent fecal coliform concentrations in Cecil soil ranged from 0.2 to 0.6%. In Maury soil, effluent concentrations in two blocks averaged 0.25% of influent concentrations (a third was at 2.4%). It appears that the capacity of fecal coliforms to move through Cecil soil is greater than in Maury soil based on these data, which is consisted with the higher dispersivity increase for the Cecil soil. It should be noted, however, that the concentration of fecal coliforms applied to Cecil soil blocks routinely exceeded that applied to Maury soil blocks, so these observations may be an artifact of slightly different experimental conditions (Table 3).
No more than 3% of the influent fecal coliform concentration was recovered in the effluent. Passage through the soil blocks filtered out, or retarded, the majority of fecal coliforms despite the apparent continuity of pores from surface to discharge point. Nevertheless, the breakthrough curves demonstrate that within 2 hours of initiating the bacterial application in the Maury soil and one to two hours in the Cecil soil, fecal coliforms were eluted. Concentrations ranged from 200 to 842,800. In all but two cases the effluent concentration of fecal bacteria exceeded minimum standards for recreational water contact of 2000 CFU/100 mL and greatly exceeded drinking water standards. Fecal coliform concentrations eluting from both soils could exceed millions of CFU per 100 mL.

We could not detect a consistent effect of adding ATCC 25254 (the second fecal coliform strain) on the displacement of ATCC 27662 (the first fecal coliform strain added to the soil blocks). If adsorption processes were responsible for bacterial retention, such a displacement might have resulted in an increase in the C/C_0 value of ATCC 27662 after switching to ATCC 25254 as the sole fecal coliform strain in the input reservoir. A small increase in C/C_0 occurred in the Maury soil for two out of the three tilled soil blocks. However, none of the Cecil soil blocks demonstrated this phenomenon.

It is worth noting that in Cecil soil, the appearance of ATCC 25254 in leachate occurred within one hour of its addition. This was a more rapid rate of appearance than we observed when ATCC 27662 was added to initiate the experiments. The same was true of ATCC 25254 added to the tilled Maury soil. However, there was no effect on the speed with which ATCC 25254 appeared in the sod-covered Maury soil. Because there was a different effect with the sod-covered and tilled Maury soils, there may yet be an unidentified dynamic between the retention of multiple strains passing through some soils depending on their tillage management.

As expected, the distribution of fecal coliforms eluting through the soil blocks was not uniform, rather, it occurred at spatially discrete locations at the bottom of the soil blocks. These locations were frequently, but not always exactly, the locations through which the second fecal coliform added to the soil blocks flowed. There were numerous cells through which no leachate flowed, and consequently, through which no fecal coliforms eluted.

Sources of Error

Rainfall Simulator Variability

Several experiments were run to determine the spatial variability associated with the water droplets coming from the rainfall simulator. These experiments were performed by putting 100 collection tubes directly underneath the simulator, and measuring the flow rate in each tube at a target flow rate of 1 cm hr^{-1}. The average coefficient of variation for the individual flow rates was 38.2%.

Because of the spatial variation in water droplets from the simulator, there was some concern that the variability in effluent fluxes measured at the base of the soil blocks was the result of the imposed rainfall pattern as opposed to the distribution and connectivity of flow paths within the block. To test this idea, a block of Cecil soil was run, first with the rainfall simulator kept static and then with it rotated 90° every 15 min.

The mean flow rates were similar, with the rotation at 0.78 cm hr^{-1} and the non-rotate at 0.76 cm hr^{-1}. The coefficient of variation for the non-rotate (177%) was slightly lower than that (197%) for the rotate. However, both of these values were much greater than the coefficient of variation for the simulator itself. Furthermore, the correlation coefficient between the effluent fluxes for the rotation of the rainfall simulator and the non-rotation was 0.75. This relatively high r-value indicates that the spatial pattern of effluent was relatively independent of the spatial pattern of the rainfall. This was confirmed by correlating the spatial distribution of flow from the simulator with the spatial distribution of effluent fluxes from the static phase of the experiment; the resulting r-value (0.02) was not significant at $P<0.05$. From the above experiments, we conclude that the spatial distribution of effluent was mainly the result of flow processes occurring within the block and/or at the block/phase barrier interface.

Phase-barrier Contact

It is possible, because of the unevenness of the bases of the trimmed soil blocks and because cells in the base plate were filled flush with glass beads, that interfacial contact was incomplete for some of the runs. The cells in the base plate have sharp edges designed to penetrate the soil matrix. However, when they are filled flush with glass beads penetration is impeded and contact reduced. This could cause water to move laterally at the base of the soil block before draining into adjacent cells.

To test this possibility, a block of Cecil soil was prepared as usual, flow was initiated, and the spatial distribution of effluent fluxes was measured over time. After 6 hr of steady flow the block was removed from the base and cleaned of all existing beads. The cells were then repacked with the same size beads as before. However, an additional 2-3 cm layer of beads was applied to the bottom of the block. The block was placed on the top of the collection chamber and was run in the same way as before, with measurements of the spatial distribution of fluxes taken for another 6 hr.

The additional beads appeared to improve the contact between the soil block and the base plate. Whereas 57 cells registered zero flow in the first run, only 14 cells had no flow in the second run. Moreover, the correlation between the spatial distributions of fluxes for the two runs was only r=0.15 (NS). Thus, poor contact may have been a factor contributing to the spatial variation in effluent fluxes observed in some of the experiments described in this report. Additional research is needed on methods to improve soil phase barrier contact in future block studies of this nature. Although poor contact may have confused our spatial variability results to some extent, it is unlikely that this factor had any significant impact on the mean effluent breakthrough curves or the TDR results. This hypothesis is supported by the observation that volumetric water contents measured at the 5, 15 and 25 cm depths during both the 'normal' and improved contact block runs were with ±1% of each other.

SUMMARY AND CONCLUSIONS

Agricultural chemicals, fertilizers, and manure that are not taken up by plants, broken down, volatilized, or lost in runoff must pass through the unsaturated (vadose) zone to reach ground water. Macropores can promote rapid, preferential flow of water, solutes and microorganisms through this zone. Flow through macropores can be considered part of a continuum of flow velocities that can be quantified using the classical convection-dispersion equation. Several alternative modeling strategies, including multi-region, transfer function and percolation models, have also been developed to improve predictions of macropore flow . However, none of these approaches currently permit prediction of water and contaminant movement in the vadose zone from independently measured soil structural parameters.

This study measured steady state solute and bacterial transport through relatively large intact soil blocks under partially saturated conditions. The main objectives were to examine:

- if water, chloride (a non-interacting chemical), and bacteria would elute through partially-saturated, intact soil blocks uniformly or via preferential flow paths;
- if these preferential paths were stable with time;

- if tillage management practices affected either the speed or pattern of water flow through the soil blocks;
- if statistical and/or geostatistical analyses could be used to predict transport processes from soil structural properties; and
- if displacement of adsorbed and/or entrapped bacteria occurs in response to the introduction of a second competing strain.

Six, 32.5 cm square x 32.5 cm deep soil blocks were obtained from a Maury silt loam soil (fine, mixed, mesic Typic Paleudalf) at the University of Kentucky Experiment Station (Spindletop Farm). Four soil blocks were obtained from a Cecil soil (clayey, kaolinitic, thermic Typic Kanhapludult) from South Carolina. Two land use treatments were sampled for each soil type: conventional-till (disk) corn production and grass pasture. Three replicate blocks per land use treatment were studied for the Maury soil and two replicate blocks for the Cecil soil

The soil blocks were excavated, encased in polyurethane foam, and transported to the laboratory. We trimmed the bottom of each individual soil block flush with its wood casing, placed the block on a collection chamber, and instrumented it with TDR probes at the 5, 15 and 25 cm depths. The collection chamber contained 100 cells arranged in a 12 x 12 array that collected effluent leaching from the block. The cells were filled with saturated glass beads, which acted as an interfacial phase barrier. A -2.0 kPa vacuum was applied to the lower boundary of each soil block via the collection chamber. Simulated rainfall was applied to the top of each block at a target rate of 1 cm hr^{-1} via 100 hundred hypodermic needles, 0.25 mm in diameter, arranged in a grid pattern corresponding with the centers of the cells in the bottom collection chamber.

Volumetric water contents and bulk electrical conductivities within the soil blocks were measured automatically using TDR. When steady flow was achieved, the concentrations of chloride and bacteria in the water supply were increased in a stepwise fashion, and the resulting breakthrough curves measured over time. Two *Escherichia coli* strains (ATCC 27662 and 25254) were applied sequentially. The effluent (chloride) and TDR breakthrough curves were fitted to the convection dispersion equation (CDE) using CXTFIT. The parameters of the CDE are the dispersion coefficient (D) and the average pore water velocity (v). These parameters were combined to give the dispersivity (α), defined as α=D/v. The goodness of fit for these analyses (R^2 generally > 0.9) indicates the CDE is applicable to the flow conditions in these experiments. At the end of each experiment the soil blocks were sampled destructively and the following soil structural properties were determined: bulk density, air entry value, pore-size distribution index (b), and saturated hydraulic conductivity.

A geostatistical analysis was performed by computing indicator variograms for the individual fluxes measured on each block. However, the results indicated little or no spatial structure in the data. Therefore, the individual flux data were analyzed using conventional statistical procedures. Since the target flux was the same for each block it was hypothesized that any differences in flow due to differences in pore-size distribution and continuity would be manifested in the variance of the fluxes. However, the actual fluxes deviated from the target fluxes, and these differences may have also influenced the magnitude of the variances. Analysis of variance indicated no significant effect of soil type, land use, or their interaction on the variance of the fluxes. The absence of any significant trends for the variance of the fluxes may be related to imperfect contact between the soil blocks and the collection chamber. For Maury soil, an average of 33 cells flowed for the sod blocks and 38 for the tilled blocks, while for Cecil soil, 50 and 40 cells flowed for the sod and tilled blocks, respectively. It is likely that more cells would have flowed if contact between the soil block and collection chamber had been completely uniform.

Correlations were performed for each block between the spatial distribution of fluxes in tray 1 (hour 1) and the distribution in tray 15 (hour 36). The comparison of these two trays provides evidence of structural (flow pattern) stability from the start of the experiment to completion. The higher mean r-values obtained for the sod blocks (0.73 and 0.61 for Maury and Cecil, respectively) as compared to the tilled blocks (0.65 and 0.44, respectively) indicates a more stable flow pattern for this land use. The sod-covered soil probably had an accumulation of organic matter, so that aggregates and pores in this treatment were more stable than those in the tilled soil. Greater structural stability would be expected to result in more stable flow patterns over time, as indicted by the higher r-values for the sod versus tilled blocks.

We could not detect a consistent effect of adding ATCC 25254 (the second fecal coliform strain) on the displacement of ATCC 27662 (the first fecal coliform strain). Therefore, our analyses focussed on the first strain only. The breakthrough curves for this strain were similar in shape for both soils. A maximum normalized concentration (C/C_0) was observed in the effluent after approximately 0.25 water filled pore volumes had been leached, and this concentration remained constant over the course of each experiment. The magnitude of C/C_0 for the Cecil soil was greater than for the Maury soil. The higher the value of C/C_0, the greater the fecal coliform transport through soil. Thus, it appears that the capacity of fecal coliforms to move through Cecil soil is greater than in Maury soil. It should be noted, however, that the concentration of fecal coliforms applied to Cecil soil blocks routinely exceeded that applied to Maury soil blocks, so this observation may be an artifact of slightly different experimental conditions.

No more than 3% of the influent fecal coliform concentration was recovered in the effluent. Passage through the soil blocks filtered out, or retained, the majority of fecal coliforms despite the apparent continuity of pores from surface to discharge point. Nevertheless, the breakthrough curves demonstrate that within 2 hours of initiating rainfall in the Maury soil and within one to two hours in the Cecil soil, fecal coliforms were eluted. In all but two cases the effluent concentrations of fecal bacteria exceeded minimum standards for recreational water contact of 2000 CFU/100 mL and always greatly exceeded drinking water standards.

Dispersivities computed from the effluent chloride breakthrough curves were much larger for the Cecil soil (3.07-9.53 cm) than for the Maury soil (0.90-2.26 cm). Large values of dispersivity correspond to a broad range of flow velocities, and a non-symmetric step input breakthrough curve that can be indicative of preferential flow. Thus, under the conditions of these experiments, the Cecil soil appeared to be more susceptible to preferential flow than the Maury soil. Analysis of variance suggested that soil management was not a significant source of variability among the soil blocks in terms of effluent dispersivity.

Dispersivities computed from the TDR data ranged from just above 1 to nearly 90 cm. The highest value was more than four times larger than the next highest value and was excluded from all subsequent statistical analyses as an outlier. The Maury soil blocks generally showed a decrease in dispersivity with increasing depth, while no consistent trend with depth was apparent for the Cecil soil blocks. There was no evidence of any interaction between depth and land use. Regression analysis revealed a significant ($R^2 = 0.28$, $P<0.01$) effect of volumetric water content on dispersivity, with dispersivity decreasing with increasing water content. Lower water contents lead to more tortuous flow paths and thus, a broadening of the velocity distribution. Dispersivity was also significantly ($R^2 = 0.26$, $P<0.01$) correlated with the b parameter, with dispersivity increasing as the pore size distribution shifted to a predominance of larger pores. Such a change in pore size distribution implies the possibility of a broader range of solute velocities and therefore a larger dispersivity. However, the existence of a highly significant correlation between water content and the b parameter may be indirectly responsible for the relationship between b and dispersivity.

We conclude that any structural effects on solute dispersion in unsaturated soils subjected to initial and boundary conditions similar to those in our experiments are likely to be indirect, and can be related to differences in water content at a given flow rate produced by differences in pore-size distribution. Future studies concerned with establishing a direct relationship between solute dispersion and soil structural properties should be conducted under saturated conditions.

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