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Measured and Predicted Solute Transport in a Tile Drained Field

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ABSTRACT

Most solute transport measurement techniques are tedious and require extensive soil excavation. A field experiment was conducted to evaluate whether surface transport properties determined by a nondestructive time domain reflectometry (TDR) technique could be used to accurately predict tile flux concentrations. A 14 by 14 m field plot selected above a 1.1-m deep tile drain was studied. Low electrical conductivity (EC) water was sprinkled on the plot surface, and after reaching a steady-state condition, a pulse of calcium chloride solution (16.3 cm) with an EC of 23 dS m^{-1} was applied through the same sprinklers. Time domain reflectometry equipment was used to record the change in EC of surface (~ top 2 cm) soil at 45 locations. The EC of the tile drainage flow was measured continuously with an EC probe. The surface convective lognormal transfer (CLT) function parameters, log mean irrigation depth, μ_I, and its standard deviation, σ_I, were found to be 3.44 and 0.94 [ln(cm)], respectively, for a reference depth of 110 cm. These surface parameters were used in a one-dimensional (1-D) CLT model and in a two-dimensional (2-D) model (CLT vertical function combined with exponential horizontal transfer function) to predict the tile flux concentrations. The 1-D CLT model predicted an earlier arrival time of chemicals to the tile drain than observed values. The root mean square error, RMSE, of the 1-D CLT predictions was 0.123, and the coefficient of efficiency, E, was −0.47. The 2-D model predictions of tile flux concentrations were similar to the observed values. The root mean squared errors (RMSE) and E were 0.023 and 0.94, respectively. The findings suggest that in this field soil, the surface solute transport properties determined by TDR could be combined with a 2-D transport model to make reasonable predictions of tile flux concentrations.

The understanding of nutrient and pesticide transport from agricultural lands into ground and surface waters is crucial for improvement of agricultural management practices. Preferential flow has been reported to be one of the major pathways of chemical movement and loss. Contamination risks of surface water receiving drainage from tile-drained fields are intensified with the occurrence of preferential flow. Kohler et al. (2003) conducted a Br leaching experiment in a 1.6-ha portion of a tile-drained arable field and found that peak of Br concentration in drainage effluent coincided with the tile drainage discharge peaks.

During a 2-yr observation period they found that 73% of applied Br leached via preferential flow that was exported away from the site through the tile drains. Everts et al. (1989) detected both nonadsorbing Br and adsorbing rhodamine (WT) dye tracer in a tile drain within the first 4 h after the initiation of irrigation. Kladivko et al. (1991, 1999), Kung et al. (2000a), and Jaynes et al. (2001) found early breakthrough of pesticides as well as Br in tile drains following 1 to 2 cm of water application. To better understand the areas prone to chemical leaching due to preferential flow, knowledge of spatiotemporal variation in solute transport properties of soil is needed. Most of the measurement techniques for quantifying preferential movement of chemicals either involve soil excavation or are limited to lysimeters and undisturbed soil columns. Since it is rarely feasible to conduct large-scale solute transport experiments through the soil profile, there is a need to develop techniques with minimal labor and soil disturbance that allows accurate prediction of subsurface solute leaching.

One possible approach is to combine surface solute transport measurements with a solute transport model to predict the subsurface leaching of chemicals. For instance, Jury et al. (1982) and Butters and Jury (1989) used their shallow soil depth (0.30 m) measurements and a CLT model (Jury, 1982) to predict solute flux concentrations at depths of 1.8 and 3.0 m. Recently, Lee et al. (2002) and Gaur et al. (2003) reported that the soil surface (top 2 cm) solute transport properties measured by TDR combined with a nonequilibrium mobile-immobile (MIM) prediction model (Toride et al., 1993) could be used to predict chemical leaching in 20 cm long undisturbed soil columns and in 30 cm deep soil profiles, respectively. Although, the MIM model has used the measured soil surface solute transport properties to successfully extrapolate surface solute concentrations to 20 or 30 cm, the MIM model may not be able to account for solute spreading at deeper depths. To address the complexity of solute transport in heterogeneous soils, Jury et al. (1982) introduced a stochastic CLT model that is based on the hypothesis that solute travel times diverge at a rate proportional to the square of the distance from the input boundary, as opposed to convective dispersion, CDE, (Parker and van Genuchten, 1984) or MIM models where rate of divergence is linear with distance. Butters and Jury (1989) reported that when model parameters were calibrated with flux concentrations in solution samplers installed at 0.3 m, the CLT provided better predictions of flux concentrations at a depth of 3 m in the field than did the deterministic CDE. For some field conditions, the CLT model may provide better predictions at deeper depths in the field than the CDE or

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Abbreviations: 1-D, one dimensional; 2-D, two-dimensional; CDE, convective-dispersion equation; CLT, convective lognormal transfer; E, efficiency; EC, electrical conductivity; MIM, mobile-immobile; pdf, probability distribution function; RMSE, root mean squared error; TDR, time domain reflectometry.

MIM models. In tile-drained fields, the CLT model has only been curve fitted to the tile drain flux concentrations (Heng et al., 1994; Mageson et al., 1994; Heng and White, 1996), however, no study has been performed to test whether the CLT model can be used to accurately predict tile flux concentrations based on using surface resident concentration measurements. If surface measurements can be used to accurately predict the tile flux concentrations, there will be a convenient method for assessing the effects of crop and tillage management practices on chemical leaching.

The CLT model basically uses shallow depth measurements and a depth scaling property to predict solute leaching at deeper depths. The depth scaling property of CLT implies a perfect correlation of solute travel times with depth in the vertical direction. In tile-drained fields, however, water flow is not strictly vertical. Water and chemicals may move primarily vertically through the unsaturated zone, and then move in a 2-D manner in the saturated zone along curved streamlines to the tile drain. The travel times in unsaturated and saturated layers of a tile drained field may or may not be correlated. Jury and Roth (1990) suggested that uncorrelated or independent travel times might apply for solute transport to a tile drain where flow through the upper zone of unsaturated soil has different characteristics than that through the saturated zone around the tile drain. Utermann et al. (1990) used an exponential distribution function to simulate solute travel times in the saturated zone of a tile-drained field. Presently, it is not known whether the CLT model alone can accurately simulate tile-drain flux concentrations by using surface resident concentration measurements or if the CLT model needs to be combined with a second model such as the exponential model used by Utermann et al. (1990) to account for travel time differences between the unsaturated and saturated zones. Further research is needed to determine how to apply transfer function models to chemical transport in tile-drained fields.

The objectives of this study were to measure the surface solute transport properties and tile flux concentrations in a strip-cropped field, and to determine whether the surface measurements could be incorporated into a transfer function model to accurately predict tile flux concentrations. A 1-D CLT model and a CLT model combined with an exponential model were used to predict the tile flux concentrations.

THEORY

Jury (1982) introduced stochastic lognormal transfer (CLT) functions that could be used to model solute transport of complex systems in a simple way by characterizing the output flux as a function of the input flux or resident flux. The CLT model assumes that solute moves by convection at different velocities in individual flow tubes without mixing between adjacent tubes. Adopting such a model means that once the probability distribution function (pdf) of solute travel time or path lengths (the transfer functions) between input and output surfaces separated by a depth, l, has been defined, the transport of the solute to other depths can be predicted. The theory is based on the linearity of the solute transport process. Solute fluxes such as tile drain fluxes in the field leaving the soil profile are obtained by convoluting the solute input function with the transfer functions. Solute transport from the surface to the tile drain was modeled in two ways: 1-D flow from surface points of entry to the tile depth (or water table), and 2-D flow that includes travel time from the surface to the water table and from points of entry at the water table to the tile drain. The procedure of determining the flux and resident concentrations for both modeling approaches is described below.

For a rectangular-pulse chemical input at the surface, with input concentration (C_{in}) as C_o for a duration of t_o, followed by zero concentration solution input, that is

$$C_{in} = \begin{cases} C_o & 0 < t < t_o \\ 0 & t > t_o \end{cases} \quad [1]$$

the resulting output solute flux and resident concentrations can be described by:

$$C^k_{out}(z, t) = \begin{cases} C^k(z,t) & 0 < t < t_o \\ C^k(z,t) - C^k(z, t-t_o) & t > t_o \end{cases} \quad [2]$$

where, $C(z, t)$ is the solute concentration at depth z and time t. The superscript k can be replaced by f for flux concentration or r for resident concentration.

The pdf for 1-D solute transport can be described by a CLT function model. The cumulative pdfs or the resulting flux, $C^f(z,t)$ and resident, $C^r(z,t)$ concentrations can be described by (Ellsworth et al., 1996):

$$C^f(z, t) = \frac{C_0}{2}\left\{1 + erf\left[\frac{\ln\left(\frac{tl^*}{z^*}\right) - \mu_l}{\sqrt{2}\sigma_l}\right]\right\} \quad [3]$$

$$C^r(z, t) = \frac{C_0 J_w}{2l^*}\exp\left(\mu_l + \frac{\sigma_l^2}{2}\right)\left(1 + erf\left\{\frac{\ln\left(\frac{tl^*}{z^*}\right) - \mu_l - \sigma}{\sqrt{2}\sigma_l}\right\}\right) \quad [4]$$

where, the wetted depths $z^* = z\theta$, $l^* = l\theta$. θ is the volumetric water content, and l is the reference depth. The constant μ_l is the logarithmic mean of solute travel time to l, and σ_l^2 is the corresponding variance. J_w is input or drainage flux density. The resulting flux and resident concentrations for a rectangular pulse input can be determined by putting Eq. [3] and [4] in Eq. [2]. The input flux J_w, and CLT parameters, μ_l and σ_l for reference depth, l, can be determined by fitting Eq. [2] to the observations. The travel time transfer function parameters (μ_l and σ_l) can be further transformed into cumulative irrigation depth function parameters (μ_I and σ_I) by the following relationships:

$$\begin{aligned} \mu_I &= \mu_l + \ln(J_w) \\ \sigma_I^2 &= \sigma_l^2 \end{aligned} \quad [5]$$

These transformed cumulative irrigation/drainage functions can be used in Eq. [3] and [4] after replacing

t with cumulative irrigation, I, and setting J_w equal to 1 and unitless.

One-Dimensional Model

Jury (1982) and Jury et al. (1986) reported that the solute travel time distribution at the reference depth, l, may be projected to depths greater than l to model the solute transport in deeper soil. The CLT uses a lognormal solute travel time pdf, $f_l(I)$, and projects solute movement beyond the depth of calibration by assuming that the solute travel times through deeper soil "layers" are perfectly correlated. This provides the following travel time pdf from the surface to depth z (i.e., the solute concentration at depth z):

$$f_z(I) = \frac{l^*}{z^*} f_l(Il/z) \qquad [6]$$

Subsequently, the reference depth for our transfer function parameter estimates was adopted as being equal to the tile depth. The solute transfer parameters μ_I and σ_I, were determined by fitting Eq. [2] and [4] to the measured surface resident concentrations that were later used with Eq. [2] and [3] to predict the tile flux concentrations.

Two-Dimensional Model

The travel time in terms of cumulative drainage, I, from the surface points of entry to the tile can be divided into two sections: (i) the drainage I_u required for solute to move from the surface to the water table; (ii) the drainage I_s required for solute to move through the saturated zone to the tile drain. The cumulative distribution function of the solute concentration at a depth z is determined by:

$$C^f(z,I) = \int_0^\infty \int_0^\infty C_{in}(I - I_u - I_s) f(z,I_u,I_s) dI_u dI_s \qquad [7]$$

Where, $f(z, I_u, I_s)$ is the joint pdf of I_u, and I_s; the subscripts u and s represent unsaturated and saturated media, respectively. If the transport paths through the vertical profile and water table path are uncorrelated, then the joint pdf may be written as the product of the pdf's through the individual zones and can be solved by convolution integral (Jury and Roth, 1990):

$$f(z,I_u,I_s) = f_{us}(z,I = I_u + I_s) = \int_0^I f_u(z,I_u) f_s\,(I - I_u) dI_u \qquad [8]$$

As a result, for a rectangular pulse input during $0 < t < t_o$ (or $0 < I < I_o$), Eq. [7] can be rewritten as:

$$C^f(z,I) = \int_0^\infty C_{in}(I - I') f_{us}(z,I') dI' \qquad [9]$$

In Eq. [8], the solute transfer function, $f_u(z, I_u)$, from the surface to the water table was considered to follow the CLT pdf (Jury et al., 1986):

$$f_u(z,I_u) = \frac{1}{\sqrt{2\pi}\sigma_I I_u} \exp\left\{ -\frac{\left(\ln\left(\frac{I_u l^*}{z^*}\right) - \mu_I\right)^2}{2\sigma_I^2} \right\} \qquad [10]$$

For commonly used drain spacings, the chemical transport in the saturated zone to the tile, $f_s\,(I_s)$ can be represented by a γ distribution function (Utermann et al., 1990):

$$f_s(I_s) = \frac{\beta^{1+\alpha} I_s^\alpha \exp[-\beta I_s]}{\Gamma(\alpha + 1)} \qquad [11]$$

Where, α is a γ or factorial function parameter, β is a scale factor [$= \eta/(S\theta)$], and η is the ratio of tile spacing, S, to the depth of impermeable layer, D, below the tile drain. Utermann et al. (1990) determined the γ function parameters by using the estimated travel time by Jury (1975) and found that the best γ fits were obtained for $\alpha = -0.05$ for $\eta = 2.5$ (deep impermeable layer) and $\alpha = 0$ for $\eta = 10$ (shallow impermeable layer). For typical drainage systems, the ratio of tile spacing to impermeable layer depth is more likely to be in the range of $\eta = 10$. Subsequently, for values of η near 10, Eq. [11] can be simplified to the following exponential model:

$$f_s(I_s) = \beta \exp[-\beta I_s] \qquad [12]$$

The pdfs in Eq. [12] (or [11]) and [10] were combined in Eq. [8] to determine the joint pdf for solute travel from the surface to the tile drain. This joint pdf was further applied in Eq. [9] to determine the solute concentration in the tile during tracer input. The 2-D prediction of tile flux concentrations as a result of a rectangular pulse input were obtained by putting Eq. [9] in Eq. [2].

MATERIALS AND METHODS

Site Description

The study was performed at the Iowa State University Agronomy and Agricultural Engineering Research Center near Ames, IA during the fall season of 2002 on a field plot with a chisel-plow, strip-cropping system. The study was focused on three crop strips consisting of soybean (*Glycine max* L. *Merr*), corn (*Zea mays* L.), and oat (*Avena* L.) (Fig. 1). The crops were harvested before conducting the experiment. A plot 14 by 14 m was selected above a tile drain. The tile drain was situated in parallel drainage system with tile spacing of 36.6 m at an average depth of 110 cm.

The soil at this site is predominantly Clarion loam in the Clarion-Nicollet-Webster soil association (USDA-SCS, 1981). The glacial till derived soil is poorly drained and moderately permeable with a slope of 2 to 3%.

Measurements

A portable irrigation system with four Gilmour oscillating sprinklers (Model# 9836z) was used to apply the water and solutions at a rate of approximately 0.2 to 0.3 cm h^{-1}. An on-off switch was used to maintain the desired rate. During tracer application, the on and off times were set to 30 and 30 s, respectively, that resulted in an irrigation rate of 0.3 cm h^{-1}. To minimize ponding in the interrows, the off time was increased to 45 s during water application, and the sprinklers attained an irrigation rate of 0.2 cm h^{-1}. Tipping bucket rain gages coupled with a datalogger were used for monitoring the sprinkler irrigation rate at a total of nine locations. Simultaneously, the drainage rate and the EC of the tile drainage water were recorded continuously. The drainage rate was determined by measuring the amount of water pumped by a flow meter. The EC of drainage

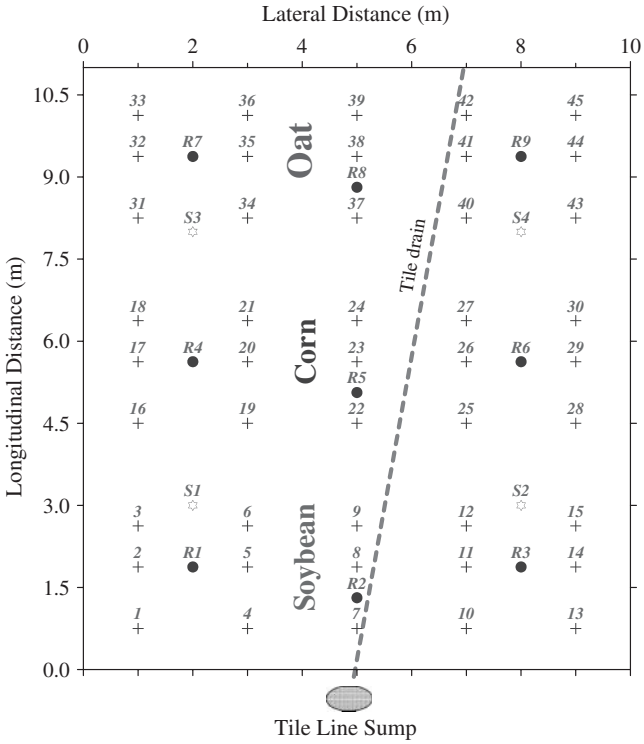

Fig. 1. Field experiment layout (S# and R# refer to sprinkler and rain gauge locations. The numbers refer to TDR probe locations for surface measurements).

water was recorded by installing an EC probe (CS547A, Campbell Scientific Inc.) at the tile outlet. In addition to an EC probe, an ISCO-3700 sampler was used to collect water samples hourly, which were later analyzed in the laboratory for EC.

The field plot was pre-irrigated with well water having an EC of 0.68 dS m^{-1} for 240 h until a steady-state water condition was attained. After reaching the steady-state condition, 17.9 cm of calcium chloride ($CaCl_2$) solution (14.4 g L^{-1}) with an EC of 23 dS m^{-1} was applied through the sprinkler system. The solute was applied for 78 h at an average rate of 0.23 cm h^{-1} and was followed by an additional well water application for 92 h at an average rate of 0.21 cm h^{-1}.

Time domain reflectometry probes were used to determine the surface chemical transport properties by measuring the bulk EC in the surface soil (~ top 2 cm). To ascertain the distribution of solute transport properties near the surface, a total of 45 TDR probes were installed within the study area which included soybean, corn, and oat cropping (Fig. 1).

The TDR equipment consisted of two-rod probes (0.38-cm diameter, 10-cm long, and 2-cm spacing), a cable tester (model 1502B, Tektronix Corp., Redmond, OR), and a computer program to store and analyze the data. The 45 probes were connected to the cable tester via a multiplexer setup. The TDR probes were inserted at about an 11° angle from the surface to an approximate depth of 2 cm. Surface insertion of angled probes has some advantages over vertical or horizontal insertion of probes. Compared with vertical probes, angled probes have a substantially greater soil/probe contact in a thin layer of soil. Increased soil/probe contact can improve the quality of the probe measurements in the soil layer. Compared with horizontal insertion of probes, surface insertion of probes minimizes soil disturbance because surface insertion removes the need for digging access trenches. Thus, surface insertion avoids disrupting the hydrology and chemical transport properties of the soil layer being measured. Surface insertion of angled probes also has some disadvantages over vertical or horizontal insertion of probes. Compared with vertical probes, angled probes can have a substantially greater fraction of the probe length near the surface-air interface. The near presence of air may influence probe readings. Compared with horizontal probes, inserted probes have more variation in depth of soil sampled. Horizontal probes can be placed at a known, single depth, while angled probes cover a range of depths. It is directly valid to assign the horizontal probe measurements to a single depth for analysis, but when the angled probe measurement is assigned to the middle depth of the probe one makes the implicit assumption that the soil EC varies linearly with depth in the soil layer sampled. The linear EC assumption may not always accurately represent soil conditions. In this study we chose to use angled, surface insertion of our TDR probes. In doing so, we avoided surface soil disturbance, but we were forced to make an assumption that each probe readings represented soil EC at the middle depth of the probe.

During the pulse input for the steady state leaching experiment, relative solute concentrations $R(t)$ can be represented as (Lee et al., 2000):

$$R(t) = \frac{C(t)-C_i}{C_0-C_i} = \frac{EC(t) - EC_i}{EC_0 - EC_i} \qquad [13]$$

where C_i is background solute concentration, C_o is input solute concentration, EC_i is TDR-measured EC for C_i, and EC_o is TDR bulk EC corresponding to C_o. Under steady-state conditions, because of the linear relationship between EC and C, one can determine normalized resident concentrations, $R(t)$, of breakthrough curves (BTCs) by using Eq. [13]. In this study, $EC(t)$ as a function of time was determined with the aid of the Win TDR99 (Or et al., 1998) computer program. It was assumed that each TDR probe measured the average bulk soil EC of the soil surrounding the probe. The actual depth of each probe was measured at the end of experiment. The probes had an average depth of 2.25 cm. No attempt to account for variable surface/air influences on the TDR readings was included, and EC measurements of each probe were assigned to the middle depth of the soil layer sampled. The average depth sampled was about 1.1 cm.

One day after irrigation ceased, soil profile cores were collected from all 45 TDR locations. A hydraulic sampling device was used to collect 3.81-cm diam., 120-long cores in zero-contaminated clear butyrate plastic tubes. Each sample was obtained in a single tube entering from the surface to a depth of 120 cm. The soil cores were later analyzed for chloride content.

Data Analysis

Examples of surface layer relative resident concentrations (determined by TDR) versus cumulative time during the pulse application are shown in Fig. 2. The observed resident concentration during the chemical application did not display an asymptotic shape during chemical application (during $0 < t < t_o$). Instead the TDR measurements showed diurnal fluctuations. These fluctuations can partly be attributed to interruptions in tracer input caused by sprinkler malfunctioning during the first two nights of tracer application. Other causes were diurnal fluctuation of surface temperature (Heimovaara et al., 1995) and variation in the input concentration due to evaporation from the sprinkler water droplets and surface soil water. The evidence of evaporation effects was independently supported by surface soil samples that were collected before switching from chemical to water application ($t = t_o$) at the time of peak TDR EC readings. Due to variation in TDR probe depths, irrigation and evaporation rate, the relative chemical concentrations of surface soil samples collected at TDR loca-

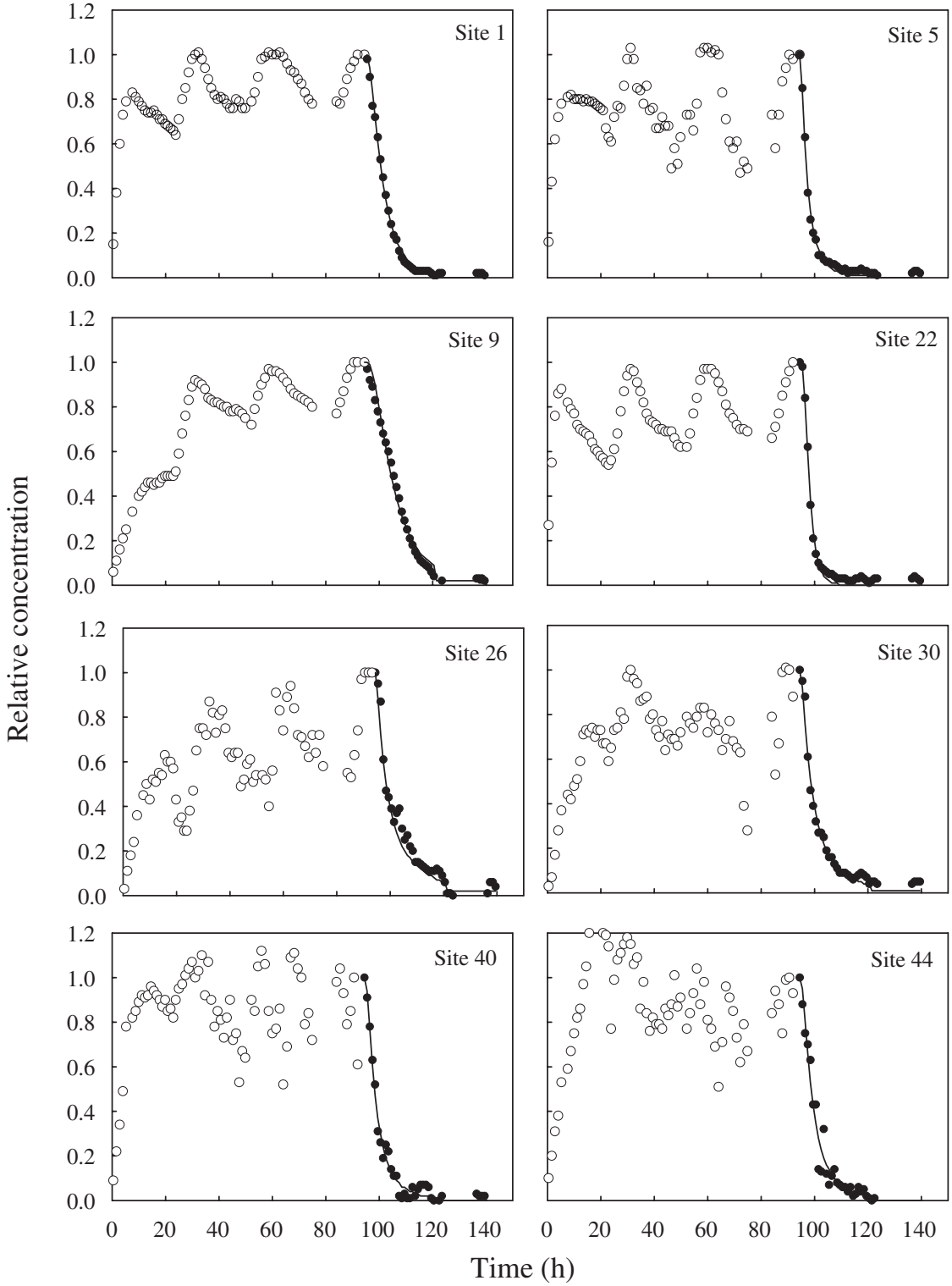

Fig. 2. Examples of measured and fitted surface resident concentrations versus time. The empty circles refer to observed concentrations during tracer application, solid circles refer to observed concentrations during water application, and the solid lines refer to the fitted concentrations.

tions ranged from 0.96 to 1.45 with an average value of 1.13. Because the initial water application following the chemical application ($t > t_o$) took place during an evening, the majority of the resident concentration curve step down occurred during the night when evaporation was at a minimum. Thus, to reduce the effects of sprinkler malfunctioning and diurnal fluctuations of temperature and evaporation, only the falling portion ($t > t_o$) of the concentration curves was selected for analysis. We assumed that after 16 pore volumes (I_o) of tracer solution input before water application, the chemical was uniformly distributed in the surface soil around the TDR probes. Subsequently, the TDR EC readings were normalized

with respect to the EC ($C_0 = EC_o$) recorded in the soil at each location when $t = t_o$. The CLT model was fitted to these measured relative resident (EC) concentrations to determine the CLT model parameters, J_w, μ_I, and σ_I. The CLT parameters for each probe location were estimated by using the mid-point depth of each corresponding probe. The estimated transfer parameters were transformed to irrigation transfer function parameters μ_I and σ_I (Eq. [5]) that were used to predict the tile flux concentrations. The tile flux concentrations were predicted by using 1-D and 2-D models by putting Eq. [3] and [9] in Eq. [2], respectively. In the 2-D model, the shallow barrier condition ($\eta = 10$) was used based on the information available on soil properties in the experimental plot. Subsequently, the constant β in Eq. [12] was taken equal to 0.063 with equivalent spacing, S, of 400 cm and satiated water content, θ, of 0.40. The S was determined by averaging the distances of farthest measurement locations from the tile drain. The average value of volumetric θ was determined by measured gravimetric water content at the 80- to 110-cm depth and assuming a bulk density of 1.47 g cm^{-3} (Kanwar et al., 1989).

The goodness of fit was evaluated by RMSE and coefficient of efficiency, E (Nash and Sutcliffe, 1970). E is one minus the ratio of mean square error to the observed data variance. E can range from minus infinity to unity. The confidence interval of fitted CLT parameters was estimated with the method suggested by Draper and Smith (1966) for nonlinear estimations.

To predict the tile flux concentrations from surface measurements, two forms of estimated transfer function parameters were used: local-scale parameters and field-scale parameters. The local-scale transfer function parameters were determined by averaging the parameters obtained at all 45 TDR locations. The field-scale transfer function parameters were obtained by fitting Eq. [5] to area-averaged resident concentrations and by assuming the depth of measurement to be equal to the average mid-point depth of all 45 probes. The area-averaged concentrations were determined by averaging the chemical concentrations over all 45 locations for each time increment. The area-averaged concentrations represent the solute transport properties of the tile-drained field by considering the soil to be a well-mixed system (Jury and Roth, 1990; Scotter et al., 1991).

RESULTS AND DISCUSSION

On average, the chemical tracer was applied at a net irrigation rate of 0.21 cm h^{-1} for 78 h totaling a net water (irrigation-evaporation) depth of 16.3 cm followed by 17.4 cm (0.19 cm h^{-1}) of net water application in 92 h. A total of 33.7 cm of irrigation was applied after the start of the tracer application. Based on the rain gauge measurements, the uniformity of irrigation was about 74%.

Near-surface Time Domain Reflectometry Measured Concentrations

During the water application following the application of tracer, the decrease in EC was rapid. After the application of 7 cm of water, the resident EC approached the initial background resident EC (Fig. 2). The fitted logarithmic mean irrigation depths, μ_I, that is, the amount of irrigation required to move the center of mass of solutes to the reference tile depth, and its standard deviations, σ_I, for all 45 locations are summarized in Table 1. The coefficients of efficiency, E, ranged from 0.91 to 0.99. For the reference depth of 110 cm, μ_I ranged from 3.08 to 3.74 [ln(cm)] with an average value of 3.44 ± 0.18 [ln(cm)] (Table 1). Out of 45 measurement locations, 87% showed μ_I in the range of 3.2 to 3.8 [ln(cm)]. On average, σ_I was found to be 0.94 ± 0.19 [ln(cm)]. Seventy percent of the 45 locations showed σ_I in the range of 0.7 to 1.1. The values of σ_I suggest large heterogeneity in solute flow that can be attributed partly to variation in irrigation rate uniformity and partly to surface soil heterogeneity.

Table 1. Convective lognormal transfer function parameters (N = 45 and Reference depth, l = 110 cm).

Parameters	Mean travel irrigation depth, μ_I ln(cm)	Standard deviation, σ_I ln(cm)
Mean	3.44	0.94
Median	3.46	0.94
Min.	3.08	0.56
Max.	3.74	1.28
Std. Dev.	0.18	0.19
C.I. (95%)	0.05	0.06

The measurement locations included soybean, corn, and oat crops. A one-way Anova test indicated that CLT parameters were not affected significantly by crops with p-value of 0.40 for μ_I and 0.45 for σ_I. Therefore, the average of all 45 locations was used for CLT parameter analysis. The surface transfer function parameters measured at 45 locations were integrated in two ways to predict the tile flux concentrations: by averaging the local-scale transfer parameters for all measurement locations (local-scale parameters) and transfer function parameters determined by area-averaged concentrations for all 45 locations (field-scale parameters). As shown in Table 2, the field-scale surface transfer function parameters, μ_I and σ_I, were found to be 3.36 and 1.04, respectively. Overall, the average local-scale transfer function parameters were not significantly different from the field-scale surface parameters. Similarly, Gupte et al. (1996) found small differences between field-scale and local-scale parameters. In contrast, Jacques et al. (1997) found greater differences between the local-scale and field-scale transfer function parameters particularly near the surface in unsaturated soil. They found larger μ_I at the local-scale than at the field-scale.

To compare these results with results from previous studies, the estimated transfer function parameters were transformed for a reference depth of 30 cm. The resulting average local-scale μ_I and σ_I were 2.14 and 0.94, respectively. These values were found to be within the range of values reported in past studies. For example, Vanderborght et al. (1996) installed horizontal TDR probes at different depths in a lysimeter having a 1 m deep undisturbed layered sandy soil monolith and found that μ_I and σ_I in the surface layer (7.5 cm depth) were 2.80 and 1.01, respectively. Furthermore, Gasser et al. (2002) conducted a leaching study in a potato field with sandy soil and determined transfer function parameters for Br resident concentrations in the soil profile of 105 cm around the lysimeter. They reported lower μ_I of 1.28 and larger σ_I of 1.07 at 7.5 cm observed depth. In contrast, Ellsworth et al. (1996) reported μ_I of 2.16 and lower σ_I of 0.22 for a fine sandy loam soil at 13.5 cm wetted depth.

Table 2. The root mean square error, RMSE, and coefficient of efficiency, E, from 1-D and 2-D model predictions of tile flux concentrations.

Method	Approach	Mean irrigation depth, μ_I [ln(cm)]†	Standard deviation, σ_I [ln(cm)]	Root mean square error	E
Tile flux concentrations	Curve fitted	3.76 (± 0.004)	0.60 (± 0.004)	0.0004	0.99
Local-scale	1-D model	3.44	0.94	0.123	−0.47
	2-D model			0.023	0.94
Field-Scale	1-D model	3.36	1.04	0.146	−1.07
	2-D model			0.032	0.89

† Reference depth, l = 110 cm.

Observed Tile Flux Concentration

As shown in the tile flow hydrograph (Fig. 3), the tile started flowing soon after irrigation commenced. The tile flow began after 6 cm of irrigation (23-h period). It took about 4 d to get steady flow in the tile drain. The drain flow fluctuated following natural rainfall of 3.3 cm on 4 and 5 Oct. 2002. After irrigating for 240 h with well water, the tracer ($CaCl_2$) application was started on 8 Oct. 2002. The irrigation rate was reduced from 0.30 to 0.23 cm h^{-1} and 0.21 cm h^{-1} during tracer and water applications, respectively to minimize ponding on the surface. After the stoppage of irrigation, the drain flow rates dropped rapidly and 1 d later, the drain was flowing at 10% of its steady state flow. From the tracer pulse net irrigation application of 33.7 cm, 68% (22.0 cm) was recovered in tile flow (Table 3). The percentage of missing inflow (~32%) was constant during water and chemical application. It is envisaged that the fraction (32%) of the total inflow not captured by the tile drain, moved away by lateral flow and deep percolation (Table 3). The sudden drop in the drain flow soon after the stoppage of irrigation indicated that the water table was near the tile depth and that the drainable porosity was small.

The tile flux breakthrough curve (Fig. 4) shows the change in relative EC with time. The change in tile flux EC was noticeable after 24 h when the irrigation depth was 5.6 cm compared with an approximate profile pore volume of 45 cm. Such evidence of preferential flow in tile-drained fields has been documented in earlier studies as well (Everts et al., 1989; Kladivko et al., 1991, 1999; Kung et al., 2000a, 2000b; and Jaynes et al., 2001). A relative flux concentration of 0.31 was observed in the tile effluent following an irrigation of 33.7 cm (171 h after the start of pulse application). Overall, the tile flow captured 14% of the total Cl applied during the study period. Of the total applied chloride, 59% was recovered in soil cores collected at the end of the experiment (Table 3). The total chloride recovery in tile and soil profile (73%) was similar to the water recovery in tile (68%), thus indicating that 27 to 32% of the total water was lost via lateral flow and deep percolation.

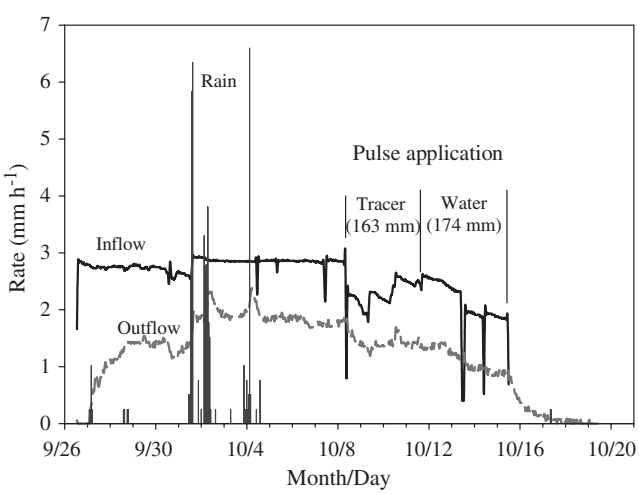

Fig. 3. Irrigation and hydrograph of tile flow.

Modeling Tile Flux Concentration

The CLT model was fitted to the observed tile flux concentrations. The model fitted the data well with an E of 0.99 (Fig. 4). The transfer function parameters, μ_I and σ_I, for the tile flux concentration BTC, were determined with respect to applied irrigation rate to compare with the surface transport function parameters. As a result, the values of μ_I and σ_I were found to be 3.76 (3.756 ± 0.004) and 0.60 (0.599 ± 0.004) for a reference depth, l = 110 cm, respectively. For comparison, Mageson et al. (1994) reported a comparatively low value of μ_I (2.88) and large value of σ_I (1.15) for 0.45-m deep mole pipe drain and 2-m spacing located in a silt loam soil under unsaturated conditions.

The value of μ_I (3.76) for tile flux concentration was larger than for the surface resident concentrations (3.44). The tile measurements indicated that a larger amount of water is required to move a solute to a certain depth than was estimated by the surface measurements. One of the obvious reasons for a larger μ_I from the tile BTC is that the actual travel distance from measurement locations is larger than the tile depth assumed during surface parameter estimates. Measurement locations positioned at relatively large distance from the tile drain would require more irrigation to move solute to the drain than would the measurement locations positioned just above the tile drain. Ellsworth et al. (1996), Persson and Berndtsson (1999); Gasser et al. (2002) also reported larger mean irrigation depth, μ_I, for flux concentrations than those estimated from the resident concentrations.

Table 3. Water and salt balance during the pulse input to the field plot.

Parameters	Water	Tracer
	cm	kg
Input	33.7	460
Tile outflow	22.9(68%)	65(14%)
Soil profile (120 cm)	0†	271(59%)
Balance	10.8(32%)	124(27%)

† No change in water storage under steady state condition.

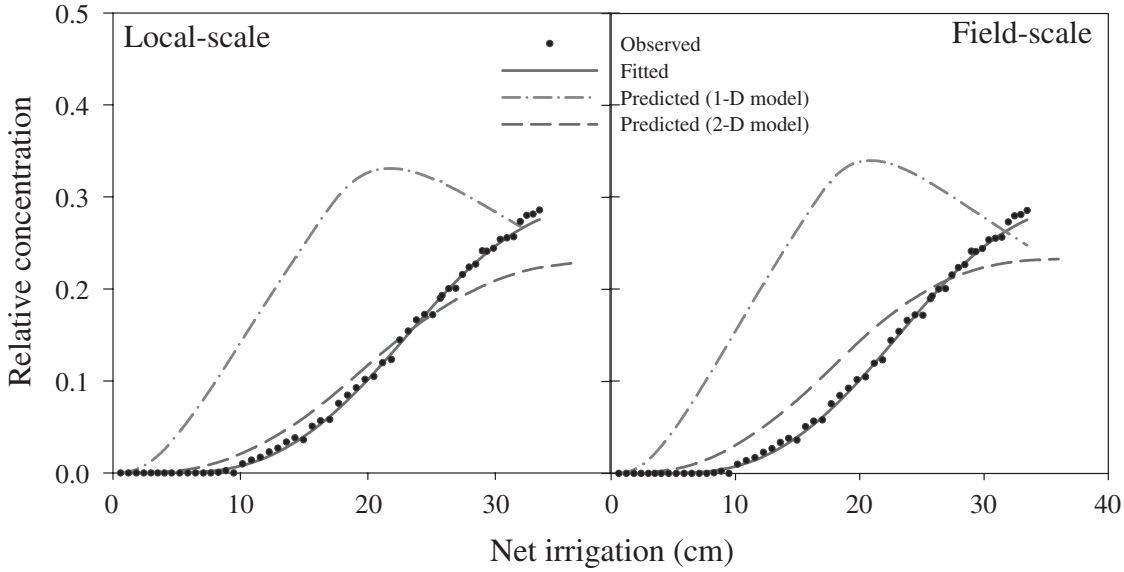

Fig. 4. Measured and predicted tile flux concentrations (0-cm irrigation depth refers to the start of tracer application that was followed by water application).

The σ_I determined from the surface (0.94) was significantly larger than the σ_I (0.60) from tile data. The large surface σ_I suggests that the surface had a relatively large solute flow heterogeneity. Relatively large surface values of σ_I were in agreement with previous studies (Jury et al., 1982; Dyson and White, 1989; Vanderborght et al., 1996; Persson and Berndtsson, 1999; and Gasser et al., 2002). For all of the previously reported studies, σ_I was larger at shallower depths than at deeper depths. For most of the previous studies, the shallowest depths ranged from 7.5 to 30 cm as opposed to the surface 2 cm used in this study.

The predicted tile flux concentrations determined by using the near-surface transfer function parameters in the 1-D and 2-D models are shown in Fig. 4. The 1-D model predictions show an earlier peak and greater peak concentrations than the observed tile flux concentrations with a RMSE of 0.123 and E of −0.47 (Table 2). Jury et al. (1982) and Butters and Jury (1989) used transfer function parameters calibrated for a 30-cm depth. They found that the CLT model was overpredicting the flux concentration at greater depths. Their overpredictions were partly attributed to the losses of the chemical past the solution samplers, which diminishes the water and chemical recovery. The early predicted chemical break-through in our case can be attributed to the assumption of 1-D flow. The 1-D approach assumes that the solute travel distance is equal to the tile depth only (110 cm) and ignores the additional solute travel distances along the curved streamlines from the water table to the tile drain.

The 2-D model uses an exponential distribution function to include the travel time distribution from the points of entry at the water table to the tile drain. The predicted tile flux concentrations by using local-scale 1-D CLT parameters and the 2-D model were similar to the observed tile flux values with a RMSE of 0.023 and E of 0.94 (Table 2 and Fig. 4). The predictions made by local-scale transfer function parameters provided numerically better estimates than the field scale parameters (RMSE = 0.032, E = 0.89). Similar predicted and observed concentrations indicated that the delay of water within the soil profile before reaching the tile drain was well accounted for by the 2-D model. The tile flux concentrations during the chemical application period were overpredicted, and they were underpredicted during the water application period. A possible explanation for this discrepancy between measured and predicted values could be that the equivalent area contributing to tile flow was smaller than the area assumed in the model. Another reason could be that the decrease in bulk density with depth led to a travel time increase with depth, which was not accounted for by the exponential travel time distribution model. The difference between average surface and subsurface bulk densities was included during predictions by considering only average surface and subsurface water contents. A third reason for fast response of predictions was possibly due to the large surface σ_I values. The surface σ_I values indicated comparatively large profile heterogeneity and hence resulted in prediction of greater preferential flow than the actual tile flow indicated. A fourth reason could be a density effect of the input tracer. In a 0.9-m sandy soil column study, Wood et al. (2004) observed a threefold increase in dispersivity and a significant decrease in mean breakthrough pore volume when the solution density increased from 1 to13 g L^{-1}. Due to instability in gravitational flow, it becomes progressively difficult to curve fit models that do not explicitly account for density effects. Despite some discrepancies, the model performed well particularly for the early breakthrough or rising limb period. The model was successful at predicting the solute concentration for a drainage pore volume of 23 cm, which represents a relatively large drainage for a subhumid region.

The findings suggest that a CLT model coupled with an exponential distribution model (2-D model) that makes use of surface solute transport properties is able

to describe solute leaching to a tile drain. The CLT model used in this measurement technique is applicable under heterogeneous soil conditions. The 2-D modeling technique performed well under our soil conditions in which measurements were made long after the tillage operations. The intact root channels from the recent crop may have caused the soil to be comparatively homogeneous with depth. In essence, this model should perform well for relatively uniform, undisturbed soil such as no-till soil and/or soil that has not been tilled for several months. The technique may have limitations for use on recently tilled soil or layered soil conditions where differences in surface and subsurface pore volumes and lateral flow may dominate. For example, Butters and Jury (1989) were successful in accurately predicting the flux concentrations down to a depth of 3 m, but due to a significant change in soil texture at 3 m, the flux concentrations at 4.5 m were underpredicted.

Since it is often impractical to conduct large-scale experiments through the entire vadose zone, the surface TDR measurement technique coupled with a 2-D transport model should offer a means for realistic estimation of subsurface leaching in tile-drained fields. Drains are installed above impervious layers; therefore, in general, it is unlikely that significant horizonation leading to dominant lateral flow will be encountered at depths shallower than the tile drain.

CONCLUSIONS

There is a need for a field solute transport measurement technique that requires minimal labor and soil disturbance. Many of the existing techniques for subsurface leaching measurements lead to extensive excavation or soil disturbance. One approach to predict subsurface leaching can be the coupling of surface measurements with a transport model. A field experiment was conducted in central Iowa to test whether surface TDR measurements can be used in such a way for accurate prediction of subsurface tile flux concentrations. The field experiment was conducted in a tile-drained 14- by 14-m field plot. Water with relatively low EC was applied to the plot by a portable sprinkler system until a steady state water condition was attained. After reaching the steady-state condition, a pulse of high EC (23 dS m^{-1}) water was applied to the plot by a sprinkler system. A 1-D convective lognormal transfer (CLT) model was fitted to the observed surface concentrations to determine the surface transfer function parameters. The surface CLT parameters, μ_I and σ_I, were found to be 3.44 and 0.94, respectively, for a reference depth of 110 cm. These surface parameters were later used by 1-D CLT and 2-D models to predict the tile flux concentrations. The 1-D model predicted earlier breakthrough of tile flux concentrations than the observed concentrations with a RMSE of 0.123 and E of −0.47. The 1-D model only included vertical solute travel to the tile depth while ignoring any lateral travel distances from the water table to tile drain. The 2-D model included vertical solute transport in the unsaturated zone and lateral travel below the water table to the tile drain. The 2-D model predictions of tile flux concentrations were similar to the observed values (RMSE = 0.023, E = 0.94). The TDR is a promising tool for determining surface solute transport properties. These findings suggest that a surface TDR measurement technique coupled with a 2-D model can offer realistic estimates of subsurface leaching in tile drained fields.

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