Evaluation of a GIS-Linked Model of Salt Loading to Groundwater

D. L. Corwin,* M. L. K. Carrillo, P. J. Vaughan, J. D. Rhoades, and D. G. Cone

ABSTRACT

The ability to assess through prognostication the impact of nonpoint source (NPS) pollutant loads to groundwater, such as salt loading, is a key element in agriculture's sustainability by mitigating deleterious environmental impacts before they occur. The modeling of NPS pollutants in the vadose zone is well suited to the integration of a geographic information system (GIS) because of the spatial nature of NPS pollutants. The GIS-linked, functional model TETrans was evaluated for its ability to predict salt loading to groundwater in a 2396 ha study area of the Broadview Water District located on the westside of central California's San Joaquin Valley. Model input data were obtained from spatially-referenced measurements as opposed to previous NPS pollution modeling effort's reliance upon generalized information from existing spatial databases (e.g., soil surveys) and transfer functions. The simulated temporal and spatial changes in the loading of salts to drainage waters for the study period 1991–1996 were compared to measured data. A comparison of the predicted and measured cumulative salt loads in drainage waters for individual drainage sumps showed acceptable agreement for management applications. An evaluation of the results indicated the practicality and utility of applying a one-dimensional, GIS-linked model of solute transport in the vadose zone to predict and visually display salt loading over thousands of hectares. The display maps provide a visual tool for assessing the potential impact of salinity upon groundwater, thereby providing information to make management decisions for the purpose of minimizing environmental impacts without compromising future agricultural productivity.

SUSTAINABLE AGRICULTURE seeks to attain a delicate balance between maintaining economic stability through increased agricultural productivity, while minimizing both the utilization of finite resources and detrimental environmental impacts (Corwin and Wagenet, 1996). The goal of sustainable agriculture is to meet the needs of the present without compromising the ability to meet the needs of the future. Assessing the environmental impact of NPS pollutants (i.e., pesticides, fertilizers, salts, and trace elements) at local, regional, and global scales is a key component to achieving the goals of sustainable agriculture because assessment provides a means of evaluating change, whether positive or negative, and of evaluating the rate of change. Assessment entails either measuring real-time or prognosticating changes in the environment. A knowledge of both real-time and simulated changes is valuable because real-time measurements reflect activities of the past, whereas predictions with a model provide a glimpse into the future and a means of taking ameliorative action prior to the development of a problem.

Groundwater degradation is regarded as one of the nation's most important environmental quality concerns because roughly half of the nation's drinking water and irrigation water comes from groundwater. Agriculture is acknowledged as the primary contributor of NPS pollutants responsible for groundwater's degradation. Currently, irrigated agriculture is being threatened because of its potential to contribute unsafe levels of organic chemicals (i.e., pesticides), salts and toxic elements (e.g., Se, B, Mo, As) to groundwater supplies. The amelioration of these problems requires a means of minimizing the load flow of solutes to groundwater. The minimization of load flows to groundwater before they manifest requires a means of predicting solute loading. The ability to locate sources of solute loading within irrigated landscapes and model the migration of solutes through the vadose zone to obtain an estimate of solute loading to the groundwater is an essential tool in combating the degradation of our groundwater. Because of the spatial complexity of the heterogeneous soil media, the modeling of NPS pollutants in the vadose zone is well suited to the integration of a one-dimensional, deterministic model of solute transport and a GIS.

Over the past decade the application of GIS to the modeling of NPS pollutants in the vadose zone has burgeoned (Corwin and Loague, 1996). The first applications of GIS for assessing the impact of NPS pollutants in the vadose zone occurred in the late 1980s with the work of Merchant et al. (1987) and Corwin and his colleagues (Corwin and Rhoades, 1988; Corwin et al., 1988, 1989). These early attempts integrated GIS with a crude index model of groundwater pollution hazard assessment (i.e., DRASTIC) and a simple multiple linear regression model of salinity development, respectively. Even though these early attempts were extremely unsophisticated, they introduced a new tool for dealing with the spatial complexities of the vadose zone's heterogeneous soil media. Currently, groundwater vulnerability potential models and some solute transport models of the vadose zone have developed into GIS-linked NPS pollution models that integrate to varying degrees solute transport models, GIS, extended relational database management systems, uncertainty analysis, and geostatistics (see symposium papers in the *Journal of Environmental Quality*, Vol. 25, no. 3). Corwin (1996) and Corwin et al. (1997) provide comprehensive reviews of GIS applications of deterministic and stochastic models to field-, basin- and regional-scale assessments of NPS pollutants in the vadose zone.

Up until now, all applications of GIS-linked models

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Abbreviations: NPS, nonpoint source; GIS, geographic information system; ET, evapotranspiration; EM, electromagnetic induction; REV, representative element volume; CIMIS, California Irrigation Management Information System; ET_o, reference evapotranspiration; ET_c, crop evapotranspiration.

for solute transport in the vadose zone at scales greater than field scale have relied upon generalized data from existing databases such as soil survey data for reasons of expedience, cost, and labor limitations (see Corwin et al., 1997). These data consist largely of ranges of values for soil physical and chemical properties derived from sparse soil samples. Associated with this type of data is a large degree of uncertainty, which makes the reliability of maps produced from modeled results questionable for anything more than a guide for sample collection strategies (Loague et al., 1996). In addition, few GIS-linked models of NPS pollution in the vadose zone have ever been validated or even evaluated at field or larger scales (Corwin et al., 1999). It is the object of this study to (i) rely to the greatest extent possible upon measured model input data to simulate the loading of salt beyond the root zone and (ii) compare the correspondence between measured and simulated results for the functional solute transport model TETrans over several thousand hectares of irrigated, agricultural land. To demonstrate the practicality of the approach, the experimental-design constraint of the study was to use measured input data for the solute transport model that were from readily-available sources (i.e., measurements routinely collected by water districts or government agencies); from the use of rapid, noninvasive measurement techniques; or from best professional judgment, as opposed to data obtained from complicated, costly laboratory analyses or imprecise, qualitative data commonly associated with soil surveys. These self-imposed constraints were intended to make the presented methodology both reliable and practical at basin scales of tens or even hundreds of thousands of hectares.

MATERIALS AND METHODS

TETrans: Model Description

The one-dimensional, functional transport model TETrans, introduced by Corwin et al. (1991), was used to predict salt loading beyond the root zone (i.e., >1.5 m) in the Broadview Water District located on the western side of central California's San Joaquin Valley. TETrans was originally developed for application to the transport of NPS pollutants such as salinity and trace elements through the vadose zone. The original philosophical design imperative of TETrans was to develop a solute transport model around input parameters and variables that were known to be routinely measured by agricultural water districts in California. This design imperative was established to minimize the need for parameter estimation methods that can have limited accuracy and to maximize the user-friendliness of the model. Furthermore, to reduce the influence of spatial variability in the input parameters, capacity parameters (e.g., field capacity, minimum water content, etc.) rather than rate parameters (e.g., saturated-unsaturated hydraulic conductivity) were used in the model.

TETrans is a transient-state, mass-balance, layer-equilibrium model that defines nonvolatile solute transport as a sequence of events or processes: (i) infiltration and drainage to field capacity, (ii) instantaneous chemical equilibration of reactive solutes, (iii) uptake of water by plant roots due to transpiration, (iv) evaporative losses from the soil surface, and (v) instantaneous chemical reequilibration. Each process is assumed to occur in sequence as opposed to the collection of simultaneous processes that actually occur. Each sequence of events or processes occurs within each depth increment of a finite collection of discrete depth increments. The physical and chemical processes taken into account include fluid flow, preferential flow, adsorption and evapotranspiration (ET). The model is driven by the amounts of rainfall or irrigation and ET. Output results include average solute concentration profile distributions, and mass-balance information including drainage amount beyond the root zone and total solute loading. A sensitivity analysis of TETrans has shown that field capacity and plant-related inputs (i.e., ET, root penetration depth, and plant water uptake distribution) are the most sensitive parameters (Corwin, 1995). A partial validation of TETrans was previously performed for the transport of chloride and boron in large soil lysimeter columns (Corwin et al., 1991, 1992).

Study Site

An area of 37 contiguous quarter sections comprising 2396 ha within the Broadview Water District located on the westside of central California's San Joaquin Valley was chosen as the experimentation site to test the ability of TETrans to predict the areal distribution of salt loading over a 5-yr study period (May 1991–May 1996). The selection of the Broadview Water District was primarily based upon (i) the availability of model input data, (ii) the district's size, and (iii) the presence of tile drainage. Broadview Water District routinely collects data that are inputs used by TETrans including crop history; irrigation dates, amounts, and salt concentrations; and drainage dates, amounts, and salt concentrations. The district is sufficiently small (only 4000 ha) to permit coverage and study of a large portion of the district. Finally, the water district is nearly completely under tile drainage. Figure 1 shows a map of the study area within the Broadview Water District with delineated quarter section lines and associated quarter section identification numbers. An extensive study period of 5 yr was chosen to dampen the spatial effects of travel-time variations for the movement of salts through the soil profile and into the tile drain system. The GIS ARC/INFO was used to create the spatial coverages and produce all map visualizations.

Sampling Strategy for Handling Soil Property Spatial Variability

A critical aspect of the study was to use a sampling strategy that would reflect the spatial heterogeneity of the physicochemical parameters and variables used in TETrans. To meet this end, the statistical routine developed by Lesch et al. (1992) was used for electromagnetic induction (EM) measurements taken with a Geonics EM-38 to determine soil sample locations. This statistical routine selects sample sites that reflect the spatial heterogeneity exhibited for bulk soil electrical conductivity. The supposition being that the EM measurements are reflective of cumulative transport processes for salinity at a given location and can be used to identify spatial domains of similar transport properties for salt. Because bulk soil electrical conductivity in arid zone soils is primarily a result of salinity, but is also influenced by water content, texture, and bulk density, this supposition relies upon minimal spatial variation in soil properties and upon uniform irrigation applications within a spatial domain defined as similar in its ability to transport salts through the vadose zone.

Within the 37 quarter sections, EM measurements (both EM_h and EM_v, where EM_h is the EM measurement taken with the coil configuration horizontal to the soil surface and EM_v is vertical to the soil surface) were acquired in each quarter

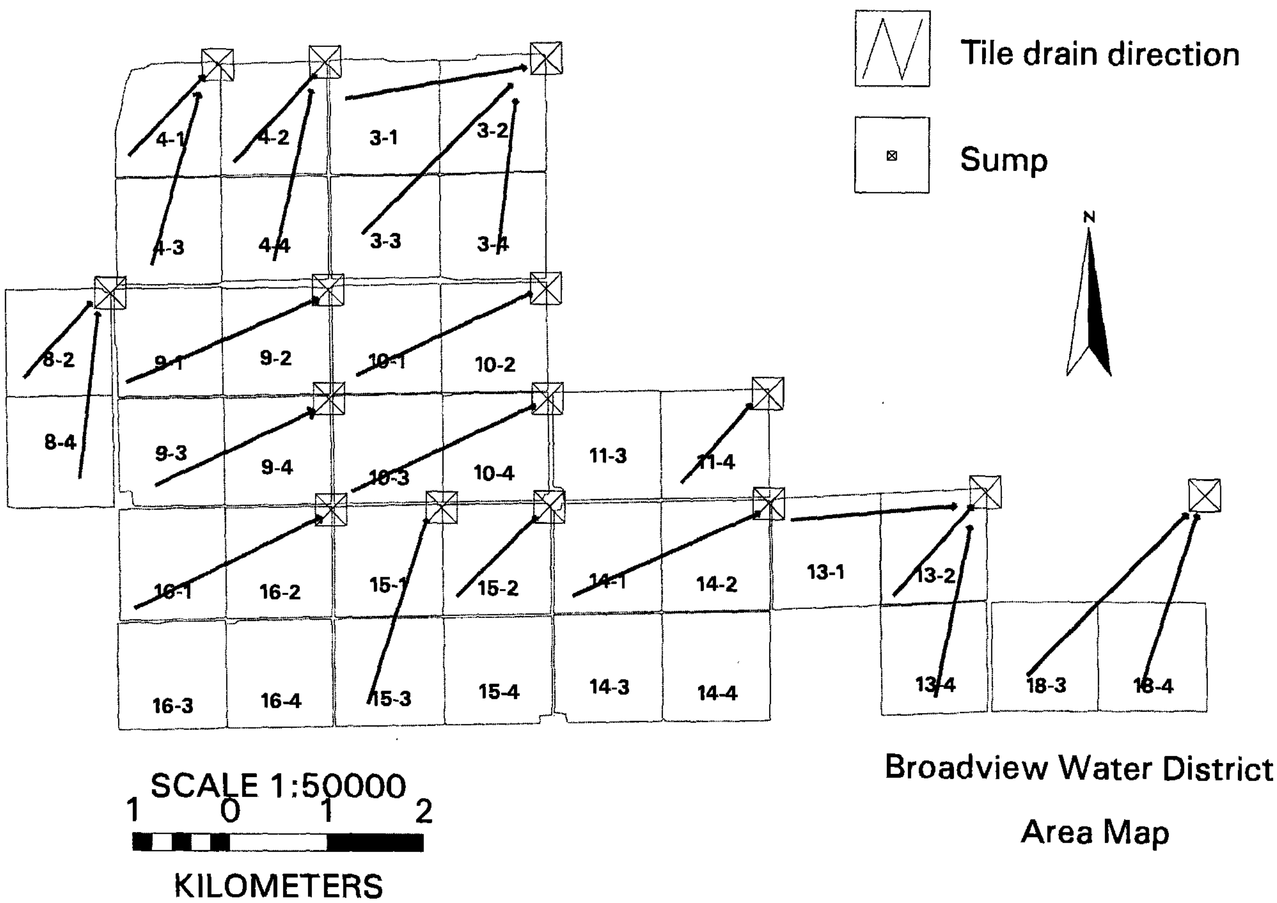

Fig. 1. Area and tile drainage map of the study site within the Broadview Water District. Map shows the quarter section boundary lines, the associated quarter section identification numbers, and the tile drainage pattern for each quarter section. The arrows indicate the area from which the underlying tile drains empty into corresponding drainage sumps.

section on a centric, systematic 8-by-8 grid generating 64 survey locations per quarter section (i.e., 2368 total locations for all 2396 ha). The geometric mean EM levels were defined as sqrt(EM_h*EM_v). The profile ratios were defined as EM_h/ EM_v. In the horizontal coil configuration, the response of the EM-38 is mainly to bulk electrical conductivity in the top 0.5 m of the soil profile. Sensitivity declines continuously with depth. In the vertical orientation, the response of the EM-38 is zero at the soil surface, peaks at roughly 0.4 m, then declines with increasing depth. Due to the response of the EM-38 in different coil orientations, the profile ratio provides an indication of the bulk electrical conductivity profile. Profile ratios equal to 1 indicate a uniform profile, profile ratios <1 indicate an increasing profile with depth, and profile ratios >1 indicate an inverted profile (i.e., conductivity decreases with depth).

To minimize soil sampling requirements to a realistic number of locations that could be handled with limited manpower resources, soil cores at 0.3 m increments to a depth of 1.2 m were taken at between 8 to 12 of the 64 locations within each quarter section. From the 2396 sites, a total of 315 locations were selected for soil-core sampling. Figure 2 shows the location of the soil-core sample sites in relation to the quarter section boundary lines. Thiessen polygons for each quarter section were created from the soil-core sample sites (see Fig. 2). Each Thiessen polygon was assumed to be analogous to a "representative element volume" (REV), representing a spatial domain of solute transport properties where the variability of the properties is least (Bouma, 1990; Mayer et al., 1999). The selection of the 315 soil sampling sites was based on the observed EM field pattern using the technique of Lesch et al. (1992). In essence, the first four sample locations of each quarter section were selected so that one location satisfied each of the following four criteria: (i) high mean EM and high profile ratio, (ii) high mean EM and low profile ratio, (iii) low mean EM and high profile ratio, and (iv) low mean EM and low profile ratio. The next four sample locations were chosen randomly within the quarter section. High and low values for the means and ratios were relative, that is, the highs and lows were identified in each field or quarter section on a field-by-field basis.

The depth of penetration of the EM-38 measurement is approximately 1 to 1.5 m depending upon the coil configuration (i.e., horizontal or vertical). A shallow water table can have a significant influence upon the EM measurement. However, 85% of the study area was tile drained with all drainage tiles installed below 1.5 m. In general, the tile drain depth varied between 1.5 to 2.4 m. In those areas where no tile drains were present, the watertable was well below 1.5 m.

Model Input Data

A complete data set of spatially-referenced input parameters and variables including irrigation data (i.e., irrigation dates, corresponding irrigation amounts and salt concentrations), crop data (i.e., crop ET amount between irrigation events; maximum root penetration depth of each crop; plant water uptake distribution of each crop; and planting date, harvesting date, and days to maturation of each crop), soil

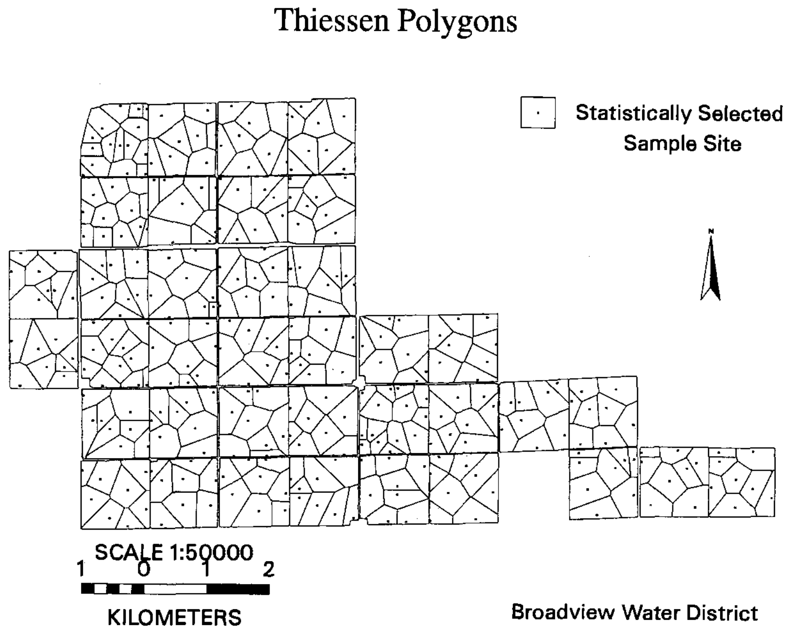

Fig. 2. Map showing the location of the 315 soil-core sample sites and the associated Thiessen polygons derived from the sample sites.

property data (i.e., thickness and bulk density of each soil horizon or layer) and initial conditions (i.e., initial water content and initial soil solution salt concentration for each soil layer) was assembled. Irrigation and drainage data were routinely measured by the water district. Reference evapotranspiration (ET_o) was calculated using the California Irrigation Management Information System (CIMIS) for the vicinity of the Broadview Water District. The reference ET_o was converted to crop evapotranspiration (ET_c) with a crop coefficient, K_c: $ET_c = ET_o \times K_c$. Figure 3 shows the ET_c for each of the 5 yr of the study.

The initial conditions of water content and total salt concentration in the soil solution were established from the soil core samples taken from April to May of 1991. Figure 4 shows the spatial distribution of salinity at 0.3 m increments to a depth of 1.2 m at the start of the study. Field capacity, wilting point, and bulk density were also determined from the soil core samples.

The boundary condition at the soil surface was established by the irrigation schedule for each quarter section and the occurrence of precipitation. Irrigations generally occurred over a 2 to 3 d period and there were generally four to seven such periods during the summer growing season. Applied irrigation amounts were corrected for tail-water losses (runoff). Measured precipitation amounts were corrected for evaporative losses due to low-salinity precipitation water that would not infiltrate into the soil and remained ponded on the soil surface. The TETrans calculation requires that irrigation or precipitation be characterized as specific events in which an amount of water is applied instantaneously; therefore, the actual input data representing the boundary conditions consisted of depth of water applied, the date applied, and the salt concentration of the irrigation water. Figure 5 displays the total irrigation + precipitation amount for each growing season of the 5-yr study period. Total dissolved solids (TDS) for each irrigation water was estimated from the electrical conductivity. Chemical analyses of the irrigation water including electrical conductivity were performed by the Soil Testing Laboratory at Colorado State University. Sampling was conducted at approximately 1-mo intervals. Figure 6 shows a graph of the varying salt concentration of the applied irrigation water from 1991 to 1996. The salinity of irrigation water varied considerably over the study period.

Most input parameters and variables for TETrans were measured values. The parameters and variables associated with the plant (i.e., ET, maximum plant root penetration depth, and plant water uptake distribution) were the exception. These parameters and variables were either estimated using the best professional judgment of farmers and water district personnel or generically derived from the literature.

TETrans has one adjustable parameter, that is, the mobility coefficient γ. The mobility coefficient accounts for preferential flow and is defined as the fraction of the resident water that is subject to displacement; therefore, 1-γ represents the fraction of soil water that is bypassed due to preferential flow. The mobility coefficient is analogous to the fraction of applied water theoretically and experimentally shown by Wierenga (1977) to be responsible for solute movement under transient water flow. Bypass is dependent upon the upper boundary condition; consequently, the ponding of irrigation water on the surface will result in a different degree of bypass than lightly sprinkling, even though the same amount of water may have been applied. For this reason, the mobility coefficient should be determined under actual field irrigation conditions. Therefore, the mobility coefficient was determined through a field calibration. The deviation of the measured chloride concentration in the soil solution from predicted chloride concentration assuming complete piston-type displacement was used as the measure of the mobility coefficient (see Corwin et al., 1991 for a detailed discussion of the mobility coefficient and its determination). Because of the near ideal water balance measured for the combined quarter sections 10-1 and 10-2, this area was used to estimate the depth-varying mobility coefficients for the entire study area.

Model Evaluation

To evaluate the predictive quality of TETrans, simulated salt loading beyond the root zone was compared to measured salt leaving tile drain sumps over the 5 yr of the study. Figure 1 provides a map of the quarter sections and locations of the tile sumps. The arrows indicate the area from which the

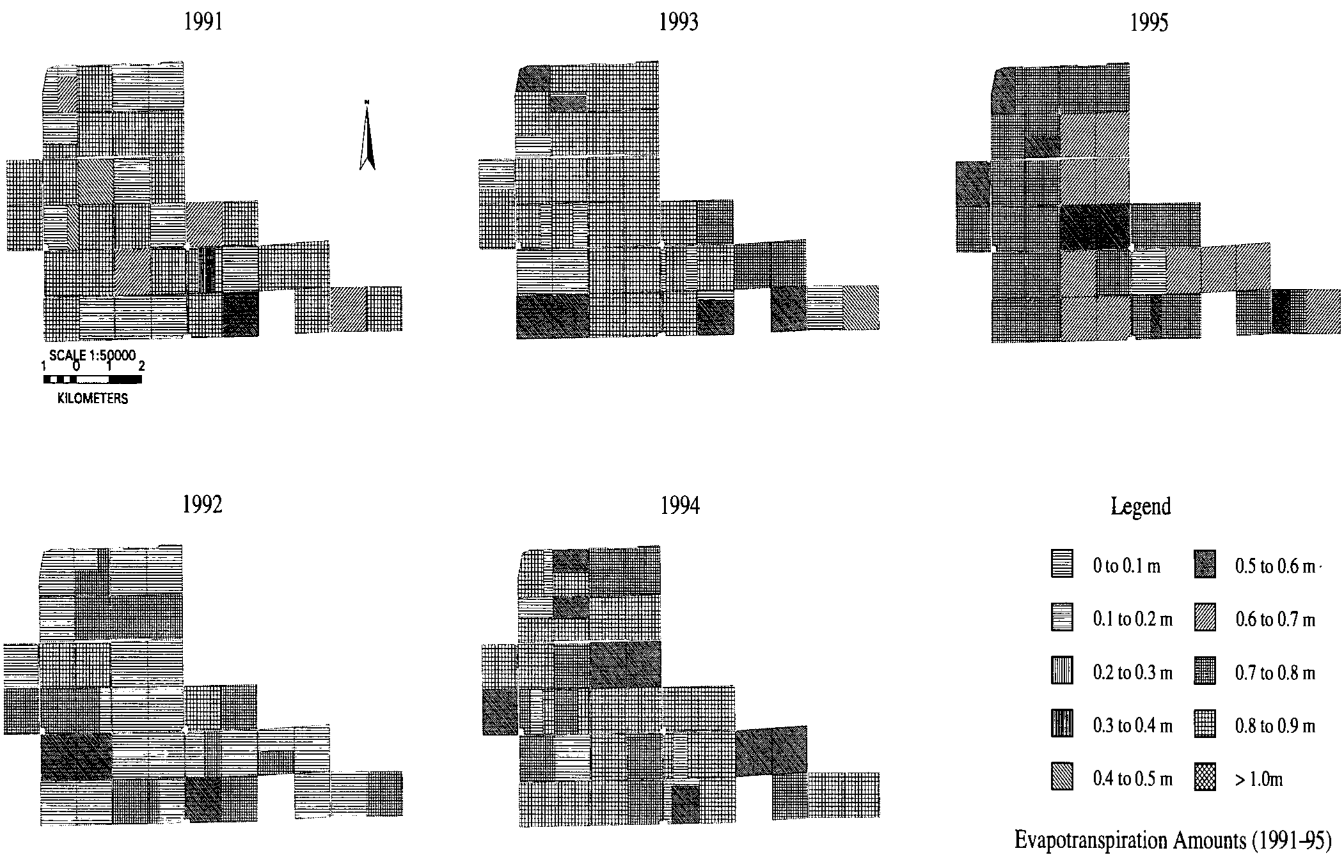

Fig. 3. Maps of evapotranspiration for each growing season from 1991 through 1995.

underlying tile drains empty into corresponding drainage sumps. Quarter sections with no arrow traversing them indicate areas with no drainage tiles. For example, the drainage tiles for quarter sections 10-3 and 10-4 drain into the sump in the northeast corner of quarter section 10-4. Therefore, to compare simulated salt loads to measured salt loads for the combined land area of quarter sections 10-3 and 10-4, the predicted salt leaving the root zone (i.e., leaving 1.5 m) from 1991 through 1996 was totaled for all Thiessen polygons comprising quarters sections 10-3 and 10-4, and compared to the measured cumulative salt drained into the sump in the northeast corner of 10-4 at the end of the 5-yr period.

DISCUSSION OF RESULTS

Water Balance

Salt transported by the lateral flow of water would confound and complicate the evaluation of a one-dimensional model such as TETrans by jeopardizing the comparison of simulated and measured salt loads. To ascertain whether or not the sump drainage was affected by the lateral movement of salt from adjacent vicinities, a water balance was performed for each drainage sump and its associated area of drained soil. The water-balance analysis combined with temporal observation well data indicated that there was a lateral flow of water and salt from outside the Broadview Water District moving from the southwest to the northeast. Only the centrally-located sections (i.e., sections 3, 4, 9, and 10) were unaffected by the lateral flow and maintained a water balance. Ostensibly, the centrally-located sections were being buffered by the surrounding sections of land whose drainage systems were efficient enough to intercept and remove the laterally flowing water. The laterally moving water was presumably from the adjacent water districts to the south and west whose lands did not have tile drainage systems. For this reason only the simulated salt loads for the following quarter sections were determined and subsequently compared to measured salt amounts from drainage sumps: 3-1, 3-2, 3-3, 3-4, 4-1, 4-2, 4-3, 4-4, 9-1, 9-2, 9-3, 9-4, 10-1, 10-2, 10-3, and 10-4. The best water balance was found to occur for the combined water draining from quarter sections 10-1 and 10-2 (subsequently referred to as drainage management unit 10-1/10-2) into the sump located in the northeast corner of 10-2 (see Fig. 1). All tile drains within the centrally-located sections of sections 3, 4, 9, and 10 were located between 1.5 to 2.4 m from the soil surface.

Mobility Coefficients

The mobility coefficient has been shown to be temporally and spatially variable (Corwin et al., 1991). Limited reliable data resulted in the determination of only a single set of depth-variable mobility coefficients that were optimized for the drainage management unit 10-1/10-2. The set of mobility coefficients (γ) was estimated to be 0.53, 0.69, 0.82, and 0.86 for depths of 0 to 0.3, 0.3

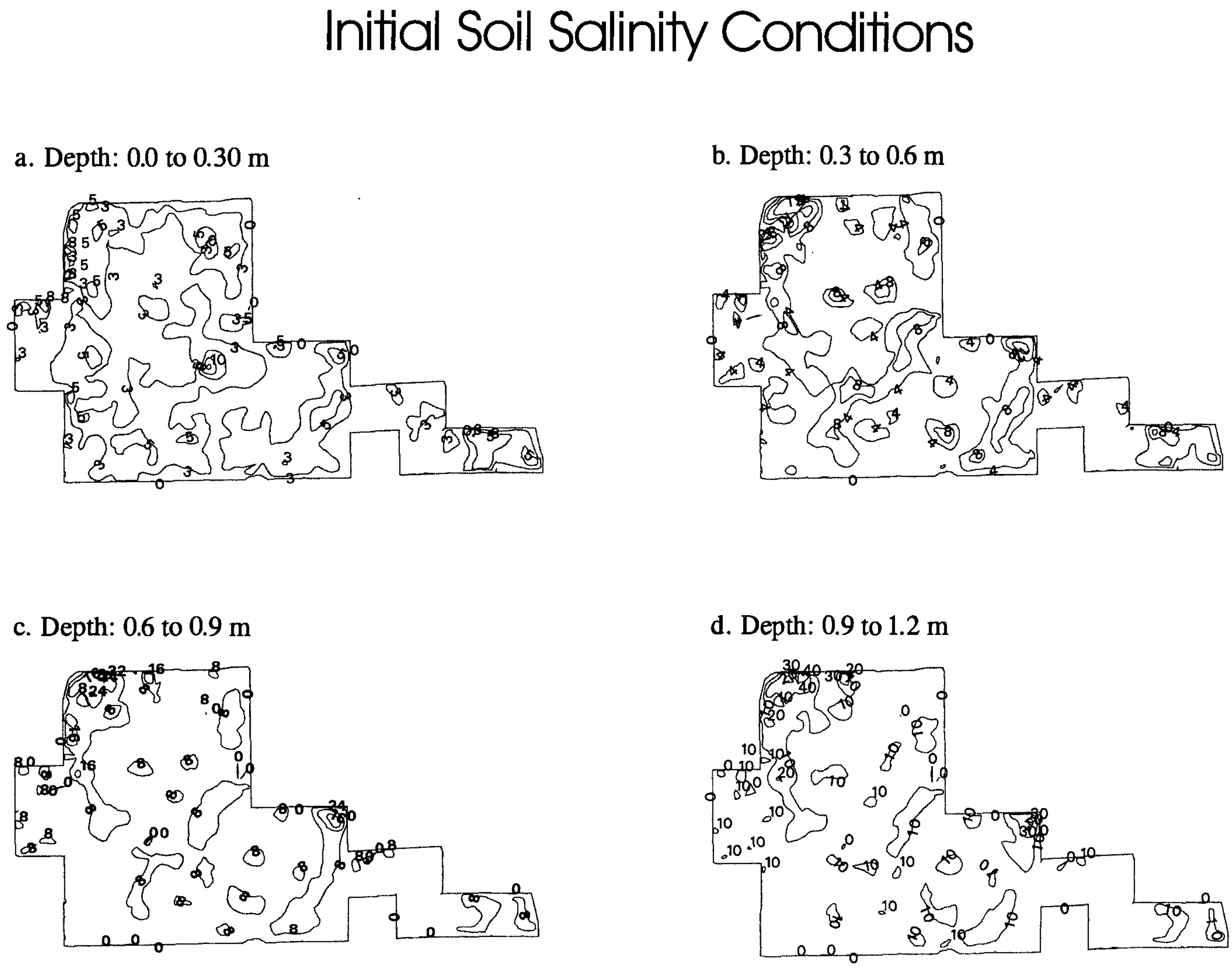

Fig. 4. Maps showing the spatial distribution of soil salinity initial conditions at depth increments of (*a*) 0 to 0.3 m, (*b*) 0.3 to 0.6 m, (*c*) 0.6 to 0.9 m, and (*d*) 0.9 to 1.2 m.

to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m, respectively (bypass coefficients = $1 - \gamma$ = 0.47, 0.31, 0.18, and 0.14).

The single set of mobility coefficients was used for all simulations in sections 3, 4, 9, and 10. The decision to use a single set of mobility coefficients was based upon practicality and observed criteria. The issue of practicality meant that the mobility coefficient must be determined from data that was either being collected by the water district or easily measured with some rapid, noninvasive technique. The issue of meeting observed criteria required that the mobility coefficients fit previously observed patterns of association and behavior (Corwin et al., 1991). For example, soils with no significant horizontal textural discontinuities typically have shown mobility coefficients that decrease with depth (Corwin et al., 1991). Furthermore, mobility coefficients that have been measured under controlled conditions have typically fallen between 0.3 and 0.85 (Corwin et al., 1991). In other words, for field-measured mobility coefficients to be considered reliable, they must make physical sense by following previously observed patterns of association and behavior.

Knowing the depth-varying mobility coefficients for each Thiessen polygon in the study would be ideal, but not practical due to the excessive cost and labor required. Next to knowing the mobility coefficients for each Thiessen polygon, a set of depth-varying mobility coefficients either for each soil type or for each drainage management unit would be preferable. Both were attempted. Determining the depth-varying mobility coefficients for each soil type required that a drainage management unit consist of a single soil type. Only drainage management units 4-1/4-3 and 4-2/4-4 fit this criteria, but just for the Lillis clay soil type (a very-fine, montmorillonitic, thermic Entic Chromoxerert) (see Fig. 1 and 7). Determining the depth-varying mobility coefficients for each drainage management unit generally resulted in poor results. Only the drainage management unit of 10-1/10-2 produced a set of optimized mobility coefficients that decreased with depth and existed over a range of values typically associated with a clay loam. Optimized fits to all other drainage management units resulted in mobility coefficients that made no physical sense. The 10-1/10-2 drainage management unit repre-

Fig. 5. Maps displaying the spatial distribution of irrigation + precipitation amounts for each growing season from 1991 through 1995.

sented the best possible site for the determination of the mobility coefficient(s) because it had the best mass balance. Furthermore, soil cores that were taken at 16 locations within quarter section 10-2 only at the end of the study period were used in the optimization fit for establishing the variation in the mobility coefficient with depth. This additional soil-core data proved useful for determining the variation with depth of the mobility coefficients, but required considerable additional field effort that under most situations of limited resources would be considered marginally practical. Though not ideal, there was sufficient data to only determine a single set of mobility coefficients with reliability. The fact that a single set of mobility coefficients was determined for just the 10-1/10-2 drainage management unit and then applied to sections 3, 4, 9, and 10 would unquestionably influence the simulated results.

Model Evaluation

Figure 8 is a map showing the spatial distribution of simulated salt loading for sections 3, 4, 9, and 10. Table 1 shows a comparison of measured and simulated results. A linear regression of the measured and simulated data resulted in a slope of 1.75 and a y-intercept of -13.56. Nevertheless, the simulated results are highly correlated to the measured results ($r = 0.99$) and are acceptable in quality for management-oriented applications.

In general, there is a tendency for the model to underestimate salt loads at the high end while overestimating at the low end (see Table 1). The greatest source of error is likely associated with the most sensitive input parameters and variables having the greatest uncertainty associated with their determination. This would be the plant parameters and variables. The ET, for instance, was estimated by using meteorologic data from

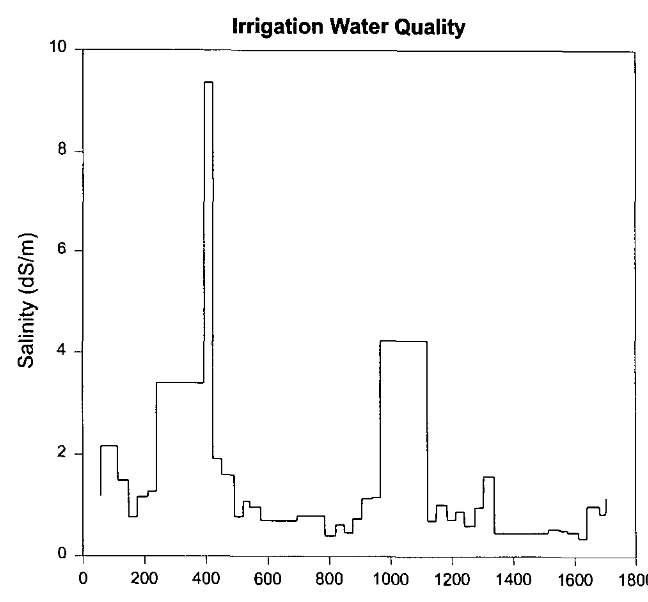

Fig. 6. A graph of the varying salinity of the applied irrigation water for the five growing seasons from 1991 through 1995.

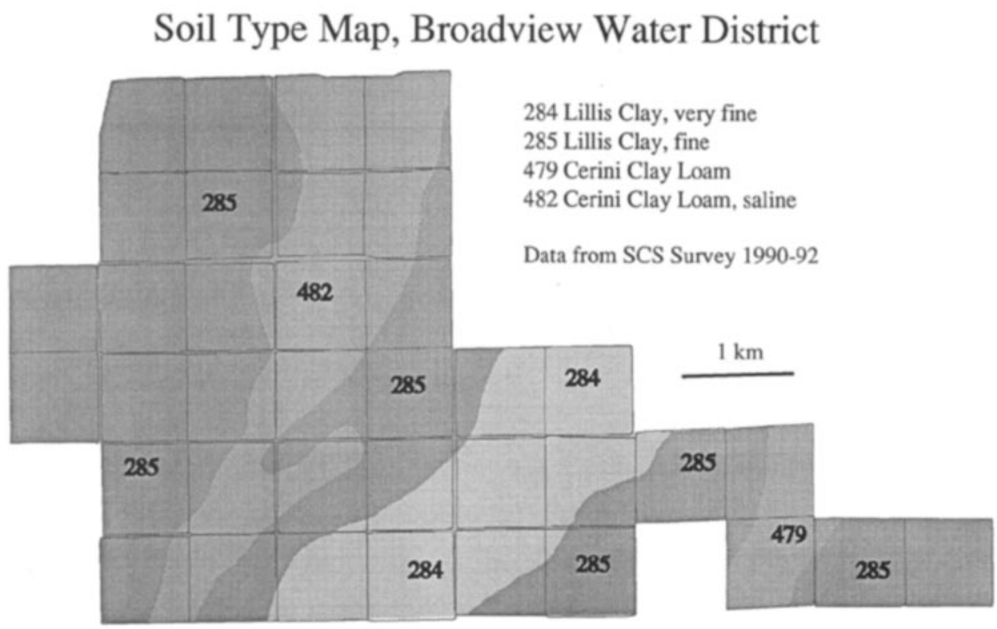

Fig. 7. Soil type map of the study area.

the vicinity, but not on-site. Actually, several miles separated the weather station from the study area. Aside from ET estimates that were questionable, the spatial variation of ET across each cropped field was not considered. Furthermore, the maximum plant root penetration depth was an estimated parameter based on field experience and the plant water uptake distributions were generic functions based on the type of crop.

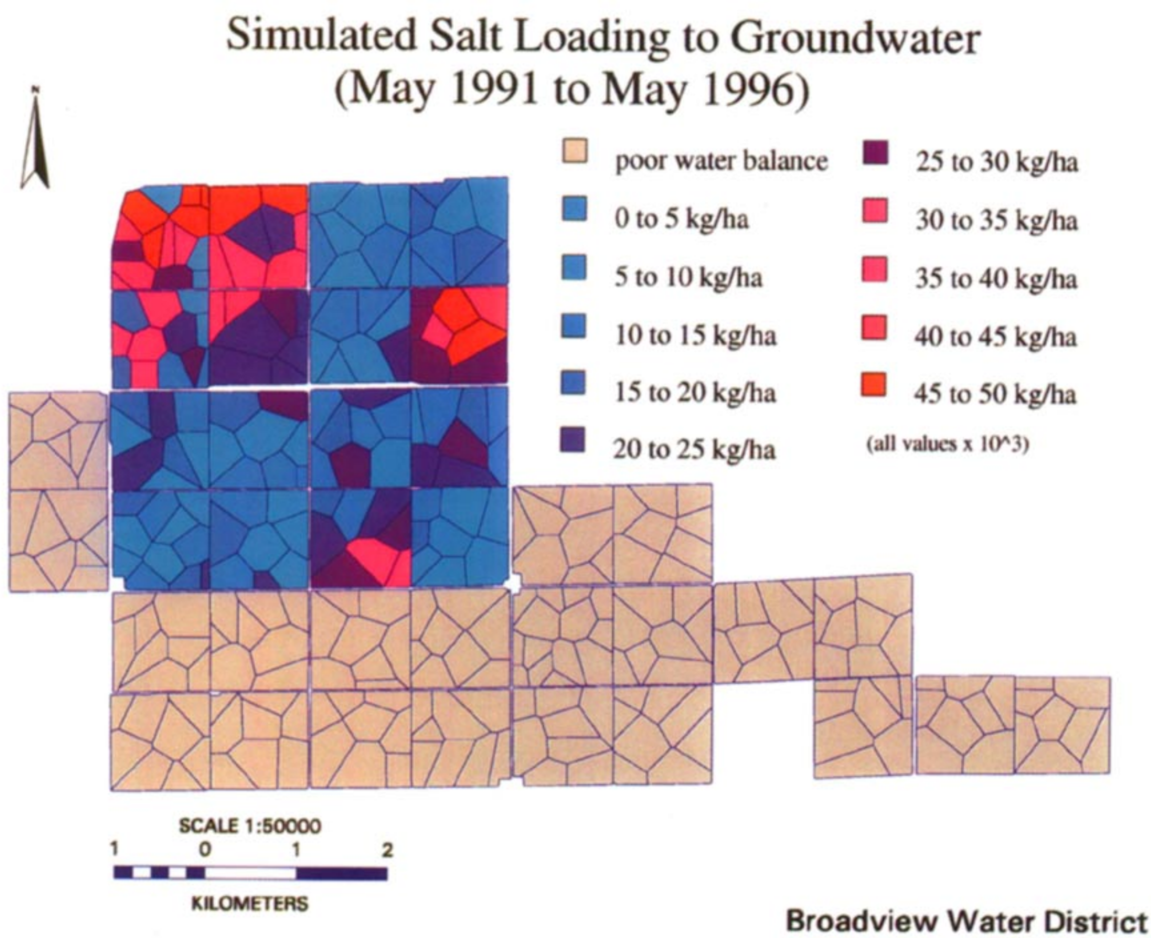

Fig. 8. Spatial distribution of salt loading simulated with TETrans for sections 3, 4, 9, and 10.

Table 1. Comparison of measured and simulated salt loading amounts in the Broadview Water District (May 1991–May 1996).

Quarter section(s)	Measured†	Simulated‡
	kg × 10³/ha	
3-1, 3-2, 3-3 & 3-4	14.33	16.97
4-1 & 4-3	39.22	31.84
4-2 & 4-4	46.23	33.00
9-1 & 9-2	11.48	13.22
9-3 & 9-4	2.1	10.45
10-1 & 10-2	16.53	16.56
10-3 & 10-4	16.05	15.91

† Measured at drainage sump.
‡ Area-weighted averaged of between 8 to 16 simulated Thiessen polygons within each quarter section.

As intimated earlier, the use of a single set of mobility coefficients is a cause for error in the model simulations. The tendency of the model to underestimate the salt loads at the high end and overestimate at the low end may not be as much a problem with the model as it is in the use of a set of mobility coefficients that is derived from a medium salt-loading area. It is plausible that the farther from the calibration point, the more likely the model is to over or underestimate the salt load. A cause for the strong correlation between simulated and measured results is that water flow appears to be the predominant transport process taking place at the scale of interest in this particular study area. As indicated by field observations and profile measurements, water appears to initially infiltrate through cracks in the soil surface that extend to depths of 0.15 to 0.3 m, followed by a lateral flow of water through the peds, and subsequently vertical piston-type displacement through the lower depths of soil (i.e., 0.3–1.2 m). TETrans appears to be able to handle the hydraulics of this particular system adequately.

SUMMARY AND CONCLUSIONS

The justification for the application of a one-dimensional model to a three-dimensional problem is a controversial issue. In this instance, the use of a one-dimensional model is based upon the concept of the REV. Dullien (1991) considered the identification of REVs to be an intuitive process related to experimental data. There is a theoretical basis justifying the use of spatially-averaged quantities over theoretical REVs (see Bear, 1972; Bachmat and Bear, 1986). In this study an alternative definition of REV along the line proposed by Bouma (1990) is used. As used herein, the REV is identified as a noninteracting, spatial domain where the lower limit of measurement scale at which variability between measurements is least (Bouma, 1990; Mayer et al., 1999). In addition, the identification of REVs is guided by practical considerations. A means of identifying REVs is proposed that uses rapid, easily-applied, noninvasive techniques of EM, which presupposes that REVs for salt leaching are associated with measurements of bulk soil electrical conductivity. Because the bulk soil electrical conductivity is influenced by the properties of soil salinity (the dominant factor in the arid zone soils of the southwestern USA), water content, texture and bulk density, spatial domains of similar EM measurements (EM_h and EM_v) are assumed to have minimal spatial variations in these and other soil properties influencing the transport of salt. An awareness of this assumption is critical to understand the limitation of using just EM measurements to establish the REVs.

This study demonstrates that TETrans linked with a GIS can be an effective and efficient technique for the evaluation of solute loading from fields or management units comprising different combinations of soils, crops, and irrigation scheduling. The major advance of the study rests on the use of a rapid assessment field reconnaissance technique (i.e., electromagnetic induction) to streamline the number and selection of field soil sampling locations while still honoring the spatial patterns within the study area. As Mayer et al. (1998) stated and may certainly hold in this study, "even if the concept of a theoretical REV does not correspond to reality, a related model may be useful if it provides reliable answers to our problems."

Most importantly, the study design is based on a comparison of measured landscape-scale field results to simulated results from a combined vadose-zone/GIS model. The agreement of measured and simulated results serves as a partial validation of TETrans as a model for solute loading to the groundwater of NPS pollutants at a scale of thousands of hectares when used in conjunction with the spatial statistical technique of Lesch et al. (1992) and coupled to a GIS. Heretofore, most GIS-linked NPS pollution models of the vadose zone have relied upon existing soil survey type databases for their basic input. Furthermore, none of the GIS-linked models have been critically evaluated for the accuracy of their simulated results.

Aside from serving as a partial validation, the results indicate the practicality and utility of applying a one-dimensional GIS-linked solute transport model of the vadose zone to predict and visually display salt loading to groundwater over hundreds or thousands of hectares. The generated maps of areal distributions of salt loading provide a useful tool for identifying those areas of greatest concern within a field or management unit so that precision farming techniques can be used to minimize solute loads and diminish environmental impacts.

Future work needs to proceed in four areas. The first is the creation of associated uncertainty maps using Monte Carlo techniques or first-order uncertainty analysis to establish the reliability of the simulations. This is currently in progress. A second area that is also in progress is to improve upon the spatial accuracy of delineating REVs or rather spatial domains with a similar propensity to transport a solute through the vadose zone. Defining the boundaries between REVs, as Dullien (1991) suggested, has been an intuitive process. The use of Thiessen polygons to represent REVs is the simplest approach to a significantly more complex problem. A means of delineating spatial domains of "homogeneous transport" needs to be developed beyond the practical, yet oversimplified, statistical routine employed herein. Improved delineation requires a rapid, cost-effective approach for defining spatial domains with a minimum variation in the transport properties. Corwin et al. (1998) have suggested the combined use of high-density EM grid samples (i.e., grid density of up to 40 sites/ha) to establish the general domains and fuzzy logic to handle transitional boundaries between these domains, with the additional use of geostatistics and the overlaying capabilities of GIS. The third area is the development of an efficient, practical means of quantifying and parameterizing preferential flow. Even though the mobility coefficient in TETrans is an adjustable parameter that characterizes preferential flow in a simple form, the spatial characterization of the mobility coefficient is probably too difficult for areas of tens of thousands of hectares or more without considerable labor and cost. Finally, the presented application of a one-dimensional model to a three-dimensional problem is cradled in an assumption of minimal interaction between REVs. Because assumptions constitute inherent limitations in a model, an understanding of the interactions between REVs and the ability to measure these interactions is needed to be able to develop the conceptual model algorithms that will account for lateral water flow and solute transport.

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