

Research Report 128

Impact of Irrigation on the Dynamics of Nitrate Movement in a Shallow Sand Aquifer ■

by

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**ILLINOIS STATE WATER SURVEY
DEPARTMENT OF NATURAL RESOURCES**

1999

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Reference: Kelly, Walton R., and Chittaranjan Ray. Impact of Irrigation on the Dynamics of Nitrate Movement in a Shallow Sand Aquifer. Illinois State Water Survey, Champaign, Research Report 128, 1999.

Indexing Terms: ground-water quality, nitrate, tracers, bromide, irrigation, ground-water flow modeling, solute transport modeling, Illinois, redox conditions.

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2204 GRIFFITH DRIVE
CHAMPAIGN, ILLINOIS 61820-7495

1999

Editors: Eva Kingston and Agnes Dillon
Graphic Designer: Linda Hascall

This report was printed with soybean ink on recycled and recyclable papers

Printed by authority of the State of Illinois (12—99—100)

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ACRONYMS AND ABBREVIATIONS

Ag	silver	Mg	magnesium
Al	aluminum	MLS	multi-level sampler
As	arsenic	Mn	manganese
B	boron	Mo	molybdenum
Ba	barium	N	nitrogen
Be	beryllium	N ₂	nitrogen gas
Br ⁻	bromide	Na	sodium
Ca	calcium	NaBr	sodium bromide
CaCO ₃	calcite	NaNO ₃	sodium nitrate
CaMg(CO ₃) ₂	dolomite	NH ₄ ⁺	ammonium
Cd	cadmium	Ni	nickel
Cl ⁻	chloride	NO ₂ ⁻	nitrite
Co	cobalt	NO ₃ ⁻	nitrate
CO ₂	carbon dioxide	NO ₃ ⁻ -N	nitrate as nitrogen
Cr	chromium	<i>o</i> -PO ₄ ³⁻	ortho-phosphate
Cu	copper	P	phosphorous
DIC	dissolved inorganic carbon	Pb	lead
DO	dissolved oxygen	PVC	polyvinyl chloride
DOC	dissolved organic carbon	S	sulfur
Eh	electrode potential	Sb	antimony
F ⁻	fluoride	Se	selenium
Fe	iron	Si	silicon
³ H	tritium	Sn	tin
HCO ₃ ⁻	bicarbonate	SO ₄ ²⁻	sulfate
Hg	mercury	Sr	strontium
HNO ₃	nitric acid	TDS	total dissolved solids
H ₂ SO ₄	sulfuric acid	Ti	titanium
K	potassium	Tl	thallium
Li	lithium	V	vanadium
MCL	maximum contaminant level	Zn	zinc

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ABSTRACT

A field-scale project in Mason County, Illinois, was performed to monitor the movement of nitrate in ground water beneath an irrigated field. Chemical tracers were used to assess the migration of solutes both laterally and vertically under the influence of an irrigation well and to determine the amount of recycling at a site due to irrigation pumpage and the amount of off-site transport of nitrate due to regional ground-water flow.

Water samples from the sand aquifer at the site reveal considerable spatial and temporal heterogeneity in aqueous chemistry. Recharge is rapid in this system, and it is probable that the water chemistry of the recharge water also is variable spatially and temporally; it is especially influenced by agricultural practices. Nitrate (NO_3^-) concentrations are elevated in a zone between approximately 15 and 30 feet (ft) beneath the surface, although this zone was not persistent laterally or with time. The maximum nitrate concentrations in this zone were slightly greater than 20 milligrams per liter (mg/L) as nitrogen, well above the drinking water standard of 10 mg/L. Nitrate was generally absent below 30 ft in the aquifer, probably due to denitrification reactions. The tritium data suggest that vertical movement of solutes in the aquifer is rapid, and that there has been enough time to transport solutes from the surface or soil zone to depths in excess of 100 ft. Because drinking-water wells generally are screened well below the zone of elevated nitrate concentrations in this area, it appears that fertilizer applications do not have a negative effect on drinking-water quality for most homeowners.

From the results of tracer tests, the effects of irrigation pumping on solute transport are measurable but not substantial. Tracer movement both horizontally and vertically was slight under pumping conditions, less than 10 ft horizontally and between 1 and 2 ft vertically about 100 ft from the irrigation well after three days of pumping. The vast majority of nitrate applied in this area is not being recycled through the irrigation wells.

INTRODUCTION

Purpose and Scope

The detection of pesticides and nitrate (NO_3^-) in rural water supplies has become a national concern in recent years. Nitrate is probably the most widespread contaminant in ground water in both the United States and the world (Hallberg and Keeney, 1993) and has numerous sources, including chemical fertilizers, mineralization of soil organic nitrogen, livestock operations, septic systems, and atmospheric deposition. Elevated levels in drinking water have been linked to methemoglobinemia, a potentially fatal condition in infants that affects transport of oxygen in the bloodstream (Kross et al., 1992). Infants ingest nitrate from formula prepared with contaminated well water. The standard procedure for dealing with high levels of nitrate in ground-water supplies has been to provide bottled water to homes with infants. Several researchers have reported circumstantial evidence linking high concentrations of nitrate with various types of cancer, although there is considerable skepticism concerning these results (Assembly of Life Sciences, 1981). The U.S. Environmental Protection Agency (USEPA) has set a maximum contaminant level (MCL) for nitrate of 10 milligrams per liter (mg/L) as nitrogen (N) for drinking water (USEPA, 1991).

The USEPA recently completed a five-year survey of pesticides in drinking water wells. The results indicate that nationally 4.2 percent of rural domestic wells contained one or more pesticides above detectable levels, and 2.4 percent contained nitrate above the federal MCL (USEPA, 1990). However, these percentages are significantly higher for areas with high ground-water vulnerability and high agricultural use (7.7 percent have nitrate above the MCL). The Monsanto Corporation also conducted a multistate well-water survey for the re-registration of alachlor. Their study found that 4.9 percent of wells contained nitrate above the MCL (Klein, 1991).

The Iowa Department of Natural Resources performed a statewide survey of agricultural chemicals in private drinking-water wells recently (Kross et al., 1990). For wells with depths less than 50 feet (ft), 35 percent of sampled wells contained nitrate above the MCL. Initial results from a pilot study conducted by the Illinois State Water Survey (ISWS) and Illinois State Geological Survey (ISGS) indicate that the frequency of detection increases as the depth of aquifer material from land surface decreases (Schock et al., 1992).

Aquifers beneath irrigated land are especially susceptible to contamination by pesticides and nitrate. Multiple and high value crops are grown with higher inputs of fertilizer, pesticides, and irrigation water. Soils under irrigation are often sandy and have low moisture-holding capacities, thus percolation rates through the unsaturated zone tend to be rapid. Depths to aquifer materials are also often shallow in irrigated areas. In Illinois, irrigated cropland increased from less than 1000 acres in 1960 to more than 325,000 acres in 1992 (U.S. Department of Commerce, 1994).

Research is needed to assess the amount of nitrogen in the subsurface hydrologic system and to examine its dynamics in irrigated areas with high vulnerability for leaching of nitrate due to soil properties and a shallow water table. Knowledge of the sources and transport of these contaminants from the sources to the regional ground-water flow system is essential in planning future well locations and evaluating existing water supplies. In addition, an understanding of the aqueous chemistry of contaminated aquifers will help in predicting contaminant fate and in designing remediation schemes.

The overall objective of this field-scale project was to monitor the movement and dynamics of nitrate in ground water in an irrigated system. Specific objectives include: (1) determine the chemical variability of the aquifer system on both temporal and spatial scales; (2) use chemical tracers to assess the migration of solutes both laterally and vertically due to the influence of pumping of an irrigation well; (3) ascertain if nitrate is recycled due to irrigation pumpage, and determine the amount of off-site transport of nitrate due to regional ground-water flow; and (4) determine aquifer properties from the tracer test data and validate a solute transport model of the system.

Acknowledgments

This report was funded by the Environmental Protection Trust Fund through the Illinois Department of Natural Resources. The views expressed herein are those of the authors and do not necessarily reflect the views of the sponsor or the Illinois State Water Survey.

Many people contributed to the successful completion of this project. Steve Wilson helped secure initial funding. Ken Rehfeldt was instrumental in locating a field site and helping restructure project objectives and methods. Andy Buck, Bryan Coulson, Joe Karny, and John Blomberg did the drilling. Joe Karny was the principal field assistant during water-quality sampling and tracer tests. Randy Locke also helped sampling. Andy Buck helped design the pump test, which he and Randy Locke helped set up. George Roadcap, Steve Burch, and Adrian Visocky did Global Positioning System (GPS) work. Chemical analyses were performed in the ISWS analytical chemistry laboratories, supervised by Loretta Skowron, with analyses by Dan Webb, Lauren Sievers, Saada Hamdy, and Tom Holm. Tom Holm also provided practical advice. Keith Hackley, ISGS, helped interpret tritium data. Randy Pepler and Van Bowersox provided precipitation data. Mark Sievers gave advice on purchasing water-level trackers, and he and Sean Sinclair prepared some of the graphics. Patti Hill prepared the appendices. Al Wehrmann helped guide preparation of the final report, which he, Tom Holm, and Randy Locke reviewed. We are especially indebted to Al Stauder and Dave Horvath for allowing us to use the Tree Nursery for this study.

PREVIOUS STUDIES

Nitrate in Agricultural Settings

Elevated concentrations of nitrate in soil and ground water almost always are the result of intense land-use activities, especially large-scale fertilizer applications, mineralization of organic nitrogen due to soil disturbance, high density animal operations, and septic systems (Keeney, 1986). Deposition of atmospheric nitrogen is also an important source of nitrogen to soil and ground water. The application of fertilizers to crops is the most widespread of the land-use sources. The greatest problems occur as a result of intensive row-cropping practices, intensive irrigated grain agriculture, and intensive irrigation and fertilization of shallow rooted crops on sandy soils (Hallberg and Keeney, 1993). Corn accounts for much of the nitrogen fertilizers applied in the United States. For example, in 1993 about 44 percent of the fertilizer applied was for corn, although only 22 percent of the cropland was in corn (Taylor, 1994). In addition, because much of the nitrogen applied to cornland in Illinois is not removed in harvested crops (Aldrich, 1980), a significant fraction is available for leaching.

In irrigated land, crops usually are heavily fertilized (Pratt, 1984). This is especially true in sandy soils, which tend to contain relatively small amounts of organic matter and, thus, are less productive than other soils. Leaching tends to be enhanced in irrigated systems, thus the potential for nitrate contamination of ground water is high in these areas (Keeney, 1982; Hubbard et al., 1984, 1986). High concentrations of nitrate have been observed in shallow ground water beneath irrigated areas in many regions, including Nebraska (Spalding et al., 1978) and Wisconsin (Saffigna and Keeney, 1977). A consistent conclusion of investigators studying nitrate movement through soils in irrigated systems is that irrigation rates must be carefully controlled to prevent excess leaching through the soil zone (Hergert, 1986; Prunty and Montgomery, 1991; Watts et al., 1991).

Chemigation, the direct addition of agrichemicals to irrigation water, potentially can pose special problems. Depending on the soil morphology, leaching may be enhanced or reduced as a result of chemigation compared to other fertilizer application methods (Schepers and Hay, 1987). If the backflow prevention device is defective or chemical handling is sloppy, accidental backflow into the irrigation well can occur and produce a pollutant point source.

The formation and movement of nitrate through soils is a complex process and depends on a number of factors, including soil morphology and structure, organic carbon content, organic nitrogen content, mineralogy and grain size, biogeochemical reaction rates, and agricultural management practices (Hubbard and Sheridan, 1989). The thickness of the unsaturated zone is an important variable that influences the travel time for solutes from the land surface to ground water.

One of the most important factors is recharge rates, or how much and how quickly water moves through the soil. Regardless of management practices and soil characteristics, there will be considerable leaching if heavy rainfall occurs soon after fertilizer application (Hubbard and Sheridan, 1989). A number of investigators have observed the greatest amount of leaching occurring in the late fall, winter, and early spring, when recharge is the greatest and plant uptake the least (Ritter et al., 1991).

The transport of nitrate that has leached below the root zone may be retarded by four different mechanisms. These are retention by soil or aquifer sediments and three microbially mediated reactions: assimilatory reduction, dissimilatory reduction to ammonium (NH_4^+), and denitrification (Korom, 1992). Assimilatory reduction is the incorporation of nitrogen into microbial biomass. In dissimilatory reduction and denitrification, nitrate is used as an electron acceptor during energetic reactions, usually oxidation of organic matter. The end products of the latter two processes are ammonium and nitrogen gas (N_2), respectively.

Retention on solids and denitrification are usually the most significant controls on nitrate concentrations in solution. Nitrate adsorption in soils and aquifer sediments is most strongly controlled by the organic matter content (Starr and Gillham, 1993). Nitrate is poorly adsorbed in sandy soils. Denitrification is an important process in many systems when oxygen is depleted, but generally it also is controlled by the amount and reactivity of organic matter. Thus, in systems with low amounts of organic matter, nitrate movement may be conservative, i.e., not noticeably retarded relative to the movement of water.

Nitrate is typically absent or present in low concentrations at depth in aquifers. This is primarily because ground water tends to be anoxic at depth, thus favoring denitrification and other nitrate-reduction reactions (Chapelle, 1993). For example, Postma et al. (1991) did not detect nitrate in deep water samples, which they correlated with a change in redox conditions with depth. Even if an aquifer is oxygenated at depth, if the primary source of nitrate is fertilizer applied over the past several decades, there may not have been sufficient time for the nitrate to migrate to depths greater than a few tens of meters. Many investigators have observed that nitrate concentrations are elevated near the water table and decrease with depth, especially in agricultural areas. Trudell et al. (1986) reported nitrate concentrations between 8 and 10 mg/L as nitrogen near the water table [about 1 meter (m) below the surface], decreasing to less than 0.1 mg/L at depths greater than 2 m. Ritter and Chirnside (1984), Hallberg (1989), Pedersen et al. (1991), Bjerg and Christensen (1992), and Geyer et al. (1992) found similar patterns at other sites, with nitrate levels highest near the water table and decreasing to below detection with depth.

This pattern is not always observed, however. Occasionally nitrate concentrations are depleted near the water table. Postma et al. (1991) found relatively depleted levels of nitrate near the water table in multi-level samplers (MLS) that were placed in forested land as opposed to arable land. They observed a plume of elevated nitrate concentrations in a fairly well-defined zone several meters below the water table, which they attributed to recharge from farmland hydrologically up-gradient. The nitrate-depleted zone near the water table was presumed to be from recharge from the noncultivated and thus nonfertilized land. Kalkhoff et al. (1992) report similar results from a site at which seven nested wells were placed. Nitrate concentrations were relatively low near the water table, increased with depth, then decreased rapidly with increased depth (> 5 m below the water table). They attributed the vertical differences to preferential flow in heterogeneous aquifer material.

Seasonal changes in nitrate concentrations have been documented in some shallow unconfined aquifers (Hallberg, 1986, 1987). Nitrate concentrations tend to increase when recharge through the soil occurs. Littke and Hallberg (1991) observed annual peak concentrations during the spring-early summer and/or fall recharge periods. Annual minima occurred during the spring snowmelt.

Management practices and weather patterns also strongly influence nitrate concentrations in ground water. Changing crop types, fertilization rates, application methods, and periods of drought and unusually wet conditions all will affect nitrate concentrations (Hallberg and Keeney, 1993). The most notable and widespread temporal trend in nitrate concentrations is the long-term increase since the 1950s and 1960s observed in a number of areas (Exner and Spalding, 1974; Spalding et al., 1978; Hallberg, 1986; Strelbel et al., 1989). This increasing trend is usually attributed to the increase in chemical fertilizer applications during this time. This increasing trend has not, however, been observed in Illinois ground water as a whole, although there are important limitations in the data (Illinois Department of Energy and Natural Resources, 1994). Nitrate contamination of ground water is probably more of a local rather than regional problem in Illinois (Illinois Department of Energy and Natural Resources, 1994).

Spatial variability in nitrate concentrations among individual wells also is observed commonly (Hallberg and Keeney, 1993). This variability is a reflection of the complex interactions between land-use practices, recharge characteristics, soil and aquifer hydraulic properties, and microbial activity, among other things. Because movement of water and solutes through soils and the saturated zone is often very slow, the application of fertilizer at the land surface may not be reflected in the ground water for several years. Thus the time frame for sampling ground water is an important consideration (Hallberg and Keeney, 1993).

Tracer Studies

Tracer studies in field settings are designed to investigate processes affecting reaction and movement of solutes or

particles in the subsurface. Conservative tracers (i.e., solutes that travel at the same rate as the advecting water) almost always are used in order to identify flow paths and rates and determine the amount of dispersion. Bromide (Br⁻), chloride (Cl⁻), and tritium (³H) are the most common conservative tracers used. Nonconservative tracers, whose movement is retarded relative to water, are used to determine how the physicochemical and biological systems affect the movement of solutes. These nonconservative tracers are often the chemicals or solutes whose subsurface fate is of interest.

The vast majority of tracer studies used to investigate the fate of agrichemicals consist of applying tracers at the land surface and monitoring movement in the unsaturated zone. Many of these studies simply monitor nitrate movement in the soil zone under different management practices (Hergert, 1986; Ritter et al., 1991; Prunty and Montgomery, 1991; Watts et al., 1991). Other studies have been designed to investigate transport processes in more detail. For example, Jabro et al. (1991) monitored bromide movement through a continuously saturated soil and concluded that nitrate could be preferentially transported to ground water in saturated soils. Jaynes et al. (1992) performed a tracer study investigating the effect of chemigation on leaching rates. They found that the movement of a conservative tracer (bromide) was enhanced under chemigation viz-a-viz side-dressing conditions.

Transport in the saturated zone also has been studied using tracers. The majority of these studies are designed to investigate the movement of organic or metal contaminants, such as at landfills, petroleum pipeline leaks, etc. For example, the U.S. Geological Survey (USGS) has performed several tracer tests using a number of tracers in a heavily instrumented site in Cape Cod, Massachusetts (Hess et al., 1992; Harvey et al., 1993). Harvey et al. (1993) investigated, among other things, bromide transport through the sandy aquifer at the Cape Cod site under natural-gradient conditions. Other comprehensive large-scale tracer studies include those reported by Sudicky (1986), Killey and Moltyaner (1988), Boggs et al. (1992), and Jensen et al. (1993). These studies all were performed under natural-gradient conditions.

Some tracer studies investigating denitrification in ground water have been performed. Trudell et al. (1986) injected 200 liters (L) of water spiked with nitrate and bromide into a sandy aquifer 3 m beneath the land surface and attempted to determine denitrification rates. They observed a decrease in nitrate concentrations of greater than 10 mg/L two weeks after injection. The loss of nitrate was preceded by removal of dissolved oxygen (DO) and coincided with production of bicarbonate (HCO₃⁻). Starr and Gillham (1989) performed similar experiments in the same aquifer and at a different site with a deeper water table (4 m vs. 1 m). Denitrification was not observed at this site, which the authors attributed to a lack of sufficient labile organic matter.

Tracer tests investigating denitrification rarely have been performed under pumping conditions, but some pertinent studies have attempted to enhance in-situ denitrification by injecting organic compounds into a contaminated

aquifer then pumping (Janda et al., 1988; Mercado et al., 1988). However, results from these tests are not particularly useful for transport calculations.

A tracer study was performed at the Sand Ridge State Forest, about 10 miles northeast of the field site used in this study, by ISWS investigators (Naymik and Sievers, 1983).

They injected several fluorescent dyes into the aquifer and followed transport under natural-gradient conditions. Plumes followed preferential pathways, as indicated by migration rates and plume geometries; these preferential pathways were concluded to be due to variability in the aquifer material that affected the flow direction and hydraulic gradient.

FIELD SITE AND METHODS

Field Site

The field site is located on the property of the Mason State Tree Nursery in Mason County, approximately 8 miles east of Havana, 17 miles west of San Jose, and 1.5 miles southeast of Topeka (figure 1). The tree nursery covers 240 acres and is bounded on the west by County Road 2400E and on the south by County Road 1750N. The field site is located in the southeast corner of the tree nursery property. The private fields near this corner of the tree nursery were planted in field corn in both 1993 and 1994.

Mason County is the most intensely irrigated county in Illinois. Nearly 89,000 acres were irrigated in 1989, which is about 25 percent of the total land in the county and more than a third of the state's total land under irrigation (Bowman and Kimpel, 1991). Potential sources of nitrate in the vicinity of the nursery include fertilizer (not applied at the nursery), soil organic nitrogen, and atmospheric nitrogen. There are no livestock facilities in the area and only isolated septic systems. Fertilizer in Mason County is typically applied with the irrigation water as 28 percent nitrogen solution (K. Donoho,

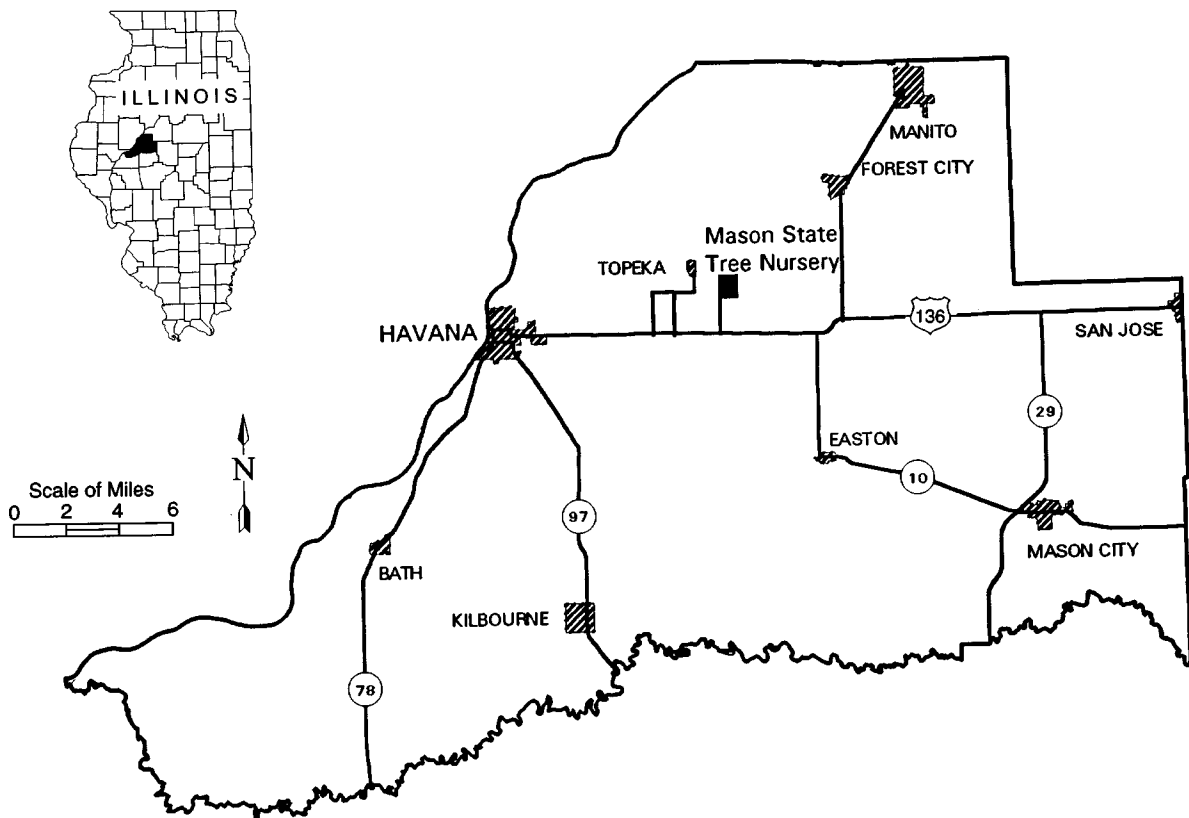


Figure 1. Location of field site at Mason State Tree Nursery in Mason County.

personal communication, 1995). Some farmers also apply anhydrous ammonia.

The main aquifer in this region is in the unconsolidated sand-and-gravel deposits of Pleistocene age overlying Pennsylvanian bedrock valleys. The water table is generally less than 20 ft below the surface, and frequently is less than 10 ft. The area is located at the confluence of the ancient Mississippi and Teays Rivers (Walker et al., 1965). The unconsolidated deposits were left by glacial meltwaters and are more than 100 ft thick in most of the region. The basal sand-and-gravel deposits have been named the Sankoty sand (Horberg, 1950). The Sankoty sand consists of about 75 percent quartz, 10-15 percent feldspar, and 10-20 percent crystalline and sedimentary rocks; it has a fairly distinct pinkish color derived from pink-tinted quartz and potassium feldspar (Walker et al., 1965). Overlying the Sankoty sand are Wisconsinan outwash deposits. These deposits consist of 60-75 percent quartz, 10-20 percent feldspar, and 20-30 percent sedimentary and crystalline rocks and minerals, with a light brown to yellowish brown color (Walker et al., 1965). Wisconsinan outwash deposits may reach thicknesses of about 100 ft.

At the tree nursery, all wells and MLSs were finished in the Wisconsinan outwash deposits. Even for the deepest wells (80 ft), the distinct pinkish color of the Sankoty sand was not observed. The Sankoty sand also is not described in the drillers' log for the irrigation well, which reached a depth of 115 ft (appendix A). The top 2-3 ft below the land surface consists of dark brown clayey sand, followed by 2-3 ft of

dark brown clay, termed "hardpan" by local drillers. Beneath this clay zone lies reddish, medium-grained sand to a depth of 40-50 ft. At greater depths, the deposits become more coarse and are gray in color, with some pebbles. The change in color from reddish to gray may represent reduction of iron oxyhydroxide coatings due to a change from oxidizing to reducing conditions. Gravel-sized pieces of coal were found in a number of zones, and they are especially abundant 35-40 ft beneath the surface.

The primary characteristic of the soils in Mason County is their low moisture-holding capacity. Fehrenbacher et al. (1984) described four soil associations in the area, all of which are sandy with low-to-moderate soil-moisture capacities. Soils of the Sparta-Dickinson-Onarga soil association are found at the tree nursery. Surface drainage is poorly developed in the region, and there is little overland runoff during rain events because the high permeabilities of these soils facilitate rapid precipitation recharge to the underlying sand-and-gravel aquifer (Bowman and Kimpel, 1991). These sandy soils also have low organic contents, necessitating application of fertilizer for adequate crop yields.

Multi-Level Samplers and Observation Wells

Multi-level samplers were constructed using 10-ft sections of 1.5-inch diameter polyvinyl chloride (PVC) pipe (figure 2). A PVC end-piece was cemented on the end of the section that would form the bottom part of the MLS. Couplers were cemented onto the top of each piece of pipe. Angled

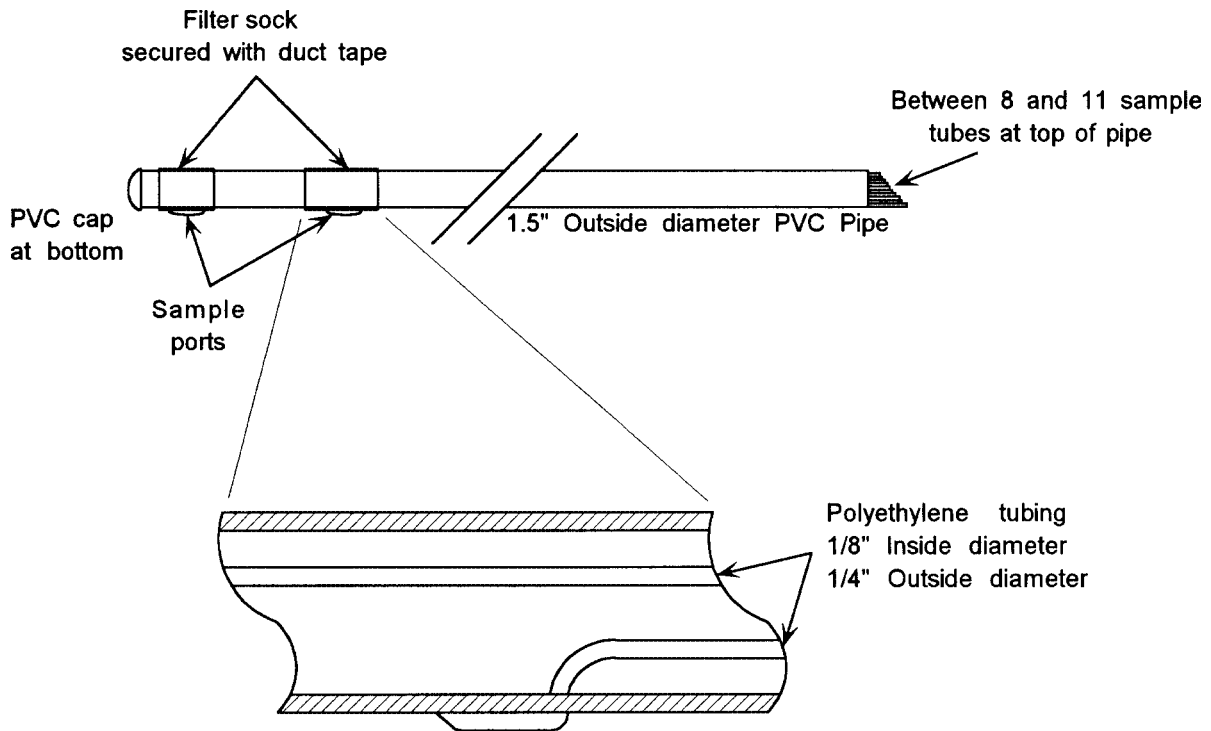


Figure 2. Construction of multi-level samplers.

holes were drilled at points where sampling ports were desired using a 17/64-inch drill bit. Lengths of polyethylene tubing (with an inside diameter of 1/8 inch and an outside diameter of 1/4 inch) were cut to the appropriate length (distance to the surface plus 5 ft); the bottom end was cut at an angle to increase inlet surface area and minimize the amount of tubing projecting outside the PVC pipe. The tubing was fed through the drilled holes upward until about an inch remained outside the drilled hole. This end was secured to the PVC pipe with duct tape, and the tubing end was left open. A section of 2-inch diameter filter sock with a thickness of 0.020 millimeter (mm) and a filtration opening size of 100-150 microns (Brainard•Kilman T-152) was pulled over the open end of the tubing then taped to the pipe above and below the open tip of the tubing. The tubing coming out the top end of the pipe was looped, secured, and marked with colored tape labeled with the tube length. Small holes (5/64-inch diameter) were drilled at the base of the lowermost section of pipe and covered with filter sock to ensure that the pipe would fill with water during installation and prevent it from being too buoyant.

The MLSs were installed using the ISWS drill rig with 4.25-inch inside diameter hollow stem auger and standard augering techniques. The auger hole was kept open by maintaining a constant head of water inside the augers. When the proper depth was reached, the drive head was disconnected from the augers to expose the top of the augers. The lowermost 10-ft section of the MLS was lowered through the top auger, keeping its top above the auger. Tubing from the lowermost section was threaded through a second section of pipe. The PVC cement was applied to the coupling, and the upper section was twisted into place to align the ports of each section. The two joined sections were held above the surface until they were bonded (two to three minutes), then lowered into the hole. Additional sections of the MLS were added in the same manner, and the pieces of tubing from lower sections were threaded through the upper sections until the bottom of the hole was reached.

After the MLS was in place, the tubing sticking out from the top was relabeled in three different ways and cut so about 1.5 inches protruded from the top. If the MLS extended too far above the surface, indicating a plug at the bottom of the hole, the augers were shaken to drop the MLS to the appropriate depth. The augers then were pulled out, without rotating, and removed. The unconsolidated sand collapsed around the MLS up to the water table (10-15 ft beneath the surface) as the augers were removed. Sand was shoveled back into the hole followed by bentonite chips the last 2 ft, then a final layer of sand. Caps for the MLSs were made from a 4-inch length of larger diameter PVC (2 inch) pipe and cap.

Observation, injection, and withdrawal wells also were installed using standard augering techniques. A minor variation was that the augers were rotated during removal. Two-inch threaded PVC schedule 40 well pipe was used with 10-slot screens for the observation and withdrawal wells and 20-slot screens for the injection wells. Four-inch well protectors were installed around the three observation wells.

We were constrained to a 25-ft buffer strip east of the irrigation well for locating the wells and MLSs to be used in the tracer experiments. Injection wells were placed between 105 and 110 ft from the irrigation well. Two parallel banks of five MLSs each (MLS-1 to MLS-5 and MLS-6 to MLS-10), which sampled to depths of approximately 45 ft and 30 ft, respectively, to the west of the injection wells, i.e., closer to the irrigation well. An additional MLS (MLS-11), which sampled to greater depths, was installed about 175 ft east of the irrigation well. After the preliminary tracer experiment, we put in an additional bank of four MLSs (MLS-12 to MLS-15), placed between the injection wells and the first bank of MLSs for better resolution. Two other MLSs (MLS-16 and MLS-17) also were placed between the initial two MLS banks. The final MLS (MLS-18) was installed near observation well 2 to assess up-gradient water chemistry. The locations of the wells and MLSs are shown in figure 3, and a summary of well characteristics is given in tables 1 and 2.

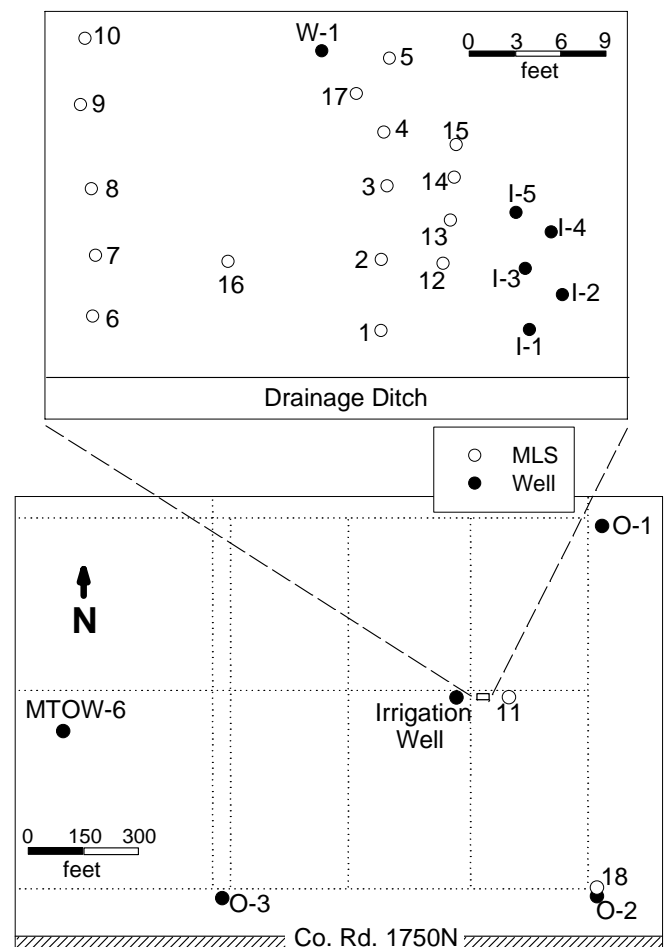


Figure 3. Location of wells and multi-level samplers on Mason State Tree Nursery property, including a detail of tracer test site. Previously existing ISWS monitoring well MTOW-6 was measured during the study.

Table 1. Well Construction Information

Wells	Depth (feet)	Screened interval* (feet)
Observation		
O-1	80	5
O-2	79	5
O-3	63	5
Injection		
I-1 to I-5	20	2.5
Withdrawal		
W-1	45	15
W-2	53	15

Note: *The screened interval is for the bottom length of the well.

Table 2. Multi-Level Sampler (MLS) Port Depths

Port no.	Depth below land surface (feet)				
	MLS-1 to MLS-10	MLS-11	MLS-12, 13, 14, 15, 17	MLS-16	MLS-18
11			16.5	11.5	
10			17.5	12.5	
9	7.0		18.5	13.5	9.5
8	10.0	13.5	19.5	14.5	14.5
7	13.0	23.5	20.5	15.5	19.5
6	17.0	33.5	21.5	16.5	24.5
5	23.0	43.5	22.5	17.5	29.5
4	27.0	53.5	23.5	18.5	34.5
3	33.0	63.5	24.5	19.5	39.5
2	38.0	73.5	25.5	20.5	49.5
1	44.5	83.5	27.0	22.0	59.5

All the wells, the MLSs, and the irrigation well were surveyed using both traditional optical techniques, which determine relative elevations, and a Leica Wilde System 200 GPS, which determined absolute spatial locations in three dimensions. The precision of the GPS is approximately ± 10 -20 mm. Lengths of well pipe above ground surface were measured manually.

Water-Level Data

The three observation wells were instrumented with current-sending water-level recorders (Telog model 2102) and 5 pounds per square inch (psi) transducers (Druck PTX-161 or PTX-164). The recorders were programmed to sample

every 30 or 60 minutes. Milliamp (mamp) readings from the transducers were converted to feet using the following relationship (G. Mayo, Telog Instruments, Inc., personal communication, 1993):

$$\text{feet} = 1.16115 \times \text{mamp} \quad (1)$$

Data were downloaded onto a laptop computer every four to eight weeks.

Manual water-level measurements were periodically taken using a water-level probe. Measurements were taken during most site visits from the injection and withdrawal wells and an observation well at the site that had been drilled as part of another ISWS study. Measurements were taken from the instrumented observation wells to help calibrate the transducer outputs.

Water-Quality Sampling

Complete chemical analyses were performed for all producing ports of MLS-3, MLS-7, and MLS-11 on five different sampling dates, approximately quarterly. Samples were collected from various MLSs on three other occasions solely for nitrate analysis.

Tubing from an MLS port was connected to a multi-channel peristaltic pump using silicone tubing and tubing connectors. Water was pumped through a flow cell, which filled from the bottom. Temperature, pH, conductivity, and redox and reference probes were fit into the cell. Water was pumped through the cell until these parameters had stabilized; pH and platinum electrode potential (Eh) took the longest times. A DO measurement then was taken by continually filling a bottle holding a DO probe. Samples were taken for transport back to the analytical laboratories at ISWS. Sampling dates, locations, and analytes are listed in table 3. Complete sampling and analytical methods and techniques are listed in appendix B.

Tracer Tests

The tracer solutions were mixed in a 200-gallon tank. Tracers were mixed with ground water taken from a well near the tree nursery offices. The tracers used were sodium bromide (NaBr) and sodium nitrate (NaNO_3). Tracers were added in amounts to obtain initial concentrations of approximately 300 mg/L Br^- and 100 mg/L nitrate as nitrogen (NO_3^- -N). The tracer solution was drained into one of the injection wells by gravity. No attempt was made to inject directly into the screened section of the well. Samples were taken periodically from the injection hose during injection to determine initial tracer concentrations. Injection of 200 gallons took about 1.5 hours. For the forced-gradient test, the irrigation pump was turned on following injection.

After injection, samples were taken from the injection well until tracer was no longer detected. Samples were taken from appropriate MLSs as the tracer plume migrated. Approximately 50 milliliters (mL) of sample were collected

Table 3. Sampling Dates, Locations, and Analytes

<i>Date</i>	<i>MLSs sampled</i>	<i>Parameters analyzed</i>
July 19, 1993	3, 7, 11	Complete* except DOC
Sep. 14, 1993	3	NO ₃ ⁻
Oct. 14, 1993	3, 7, 11	Complete except DOC
Nov. 17, 1993	1-5	NO ₃ ⁻
Feb. 4, 1994	3, 7, 11	Complete
Apr. 14, 1994	11, 18	Complete
Apr. 19, 1994	3, 7	Complete
July 19, 1994	3, 11	Complete plus ³ H
July 19, 1994	18	³ H
July 21, 1994	7, 17	Complete
Nov. 3, 1994	All	NO ₃ ⁻

Note: *Complete analysis includes temperature, pH, Eh, DO, conductivity, major cations (Ca, Mg, Na, K), major anions (Cl, SO₄²⁻, alkalinity), NO₃⁻, NO₂⁻, NH₄⁺, *o*-PO₄³⁻, metals and minor elements (Ag, Al, As, B, Ba, Be, Cd, Co, Cr, Fe, Hg, Li, Mn, Mo, Ni, P, Pb, S, Sb, Se, Si, Sn, Sr, Ti, Tl, V, Zn), TDS (for both 104 and 180°C), and DOC.

and brought back to the ISWS laboratories for each analysis. Single-junction combination ion-selective electrodes were used to analyze for bromide and nitrate (Cole-Parmer H-27502-04 and H-27502-30, respectively). Ionic strength adjustor was added to each sample: 1 mL per 50 mL of sample prior to analysis. Several standards between 0.1 and 300 mg/L were prepared by diluting 1000 mg/L standard solutions. Millivolt values for the standards were plotted on semilog paper, and a linear regression was performed to determine the slope. The regression coefficient (*r*²) for four standards was always greater than 0.99. Sample concentrations were calculated using the regression equation. The confidence interval ($\pm 2\sigma$) for electrode analysis was determined to be ± 4.4 percent; the absolute error increases with increasing concentration due to the logarithmic relationship.

Samples were collected from the MLSs until tracer was no longer detected in them. The test was concluded when the MLS furthest down-gradient contained no detectable tracer.

RESULTS AND DISCUSSION

Precipitation and Water-Level Data

The average annual precipitation in Mason County over the last 35 years is almost 40 inches (Bowman and Kimpel, 1991). Record amounts of rainfall were recorded in Mason County in 1993, and the summer and early fall were especially wet. A weather station maintained by the Soil Conservation Service is located at the tree nursery. Unfortunately, the equipment did not work for long periods of time during our study. The ISWS maintains about 20 raingages for the Imperial Valley Water Authority in Mason County. The raingage closest to the tree nursery is in Topeka, about 1.5 miles north; it has been in operation since September 1992. Weekly precipitation amounts from this station are shown in figure 4.

Because of the low moisture-holding capacities of the soils, percolation rates are rapid and thus the water table is sensitive to precipitation patterns. Ground-water levels were very high during 1993; in fact, considerable amounts of land were inundated in 1993 and into 1994. Peak ground-water levels were measured at the site in September and October 1993, and generally fell throughout the remainder of 1993 and all of 1994 (figure 4). Water levels varied by about 8 ft during the study period. To put a change of this magnitude in perspective, if water levels increased an average of 5 ft over

the entire county, more than 100 million gallons of additional water were moving through the aquifer. Because of the fluctuation of the water table, the top two ports in MLS-1 to MLS-10 were sometimes located in the unsaturated zone, especially during the latter period of the study.

Piezometric surface maps have been prepared for this region in the past by ISWS researchers, including Walker et al. (1965) and Sanderson and Buck (1995). The maps prepared by Sanderson and Buck (1995) during 1992 and 1993 show how sensitive the water table is to rainfall patterns in this region; the water table in a large part of the region changed by more than 5 ft in a year (figure 5). Regionally, flow is toward the Illinois River, which forms the northwest border of Mason County. There is local discharge into Culver Creek, which is approximately half a mile north of the tree nursery.

Ground-water flow directions and gradients at the field site were determined using water-level data from the three observation wells. Calculated flow directions varied between 13.7 and 21.8 degrees west from north, with gradients between 0.0024 and 0.0033 ft/ft. These slight changes in direction and gradient are probably inconsequential with respect to the tracer tests because fairly large volumes of tracer solution were injected, and tracer was monitored over relatively small distances.

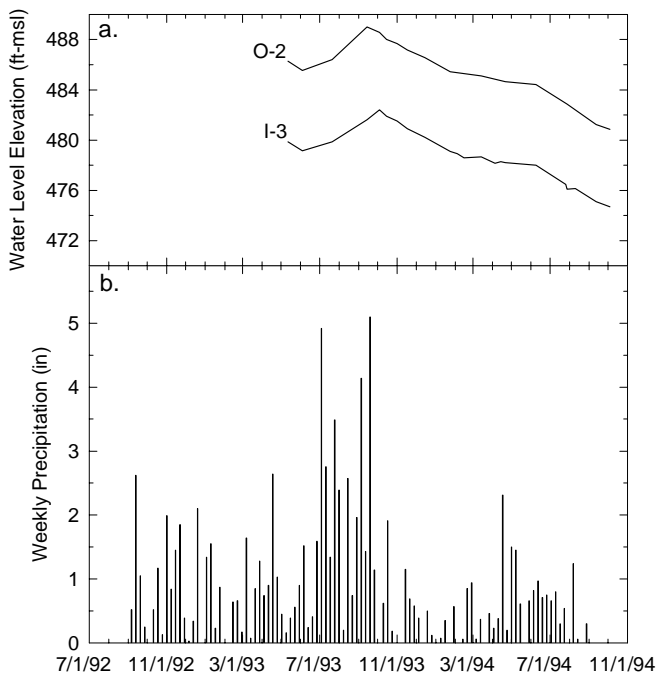


Figure 4. a. Ground-water levels for wells O-2 and I-3 at field site and b. weekly precipitation data from ISWS raingage in Topeka.

Water Quality

Vertical migration of solutes is significant in the aquifer. The precipitation data show how sensitive the water table is to precipitation patterns, and the chemical data presented here reinforce this observation.

Depth Profiles

Multi-level sampler-3, MLS-7, and MLS-11 were sampled quarterly from July 1993-July 1994. Although concentrations varied for the different sampling times, the trends of the vertical profiles were similar for the duration of the project. Complete water chemistry results are given in appendix B.

Vertical profiles of the chemical parameters for MLS-3 and MLS-7 fall into three general patterns: (1) low concentrations near the water table, elevated levels in a zone between about 10 and 30 ft below the water table, then decreased levels below this zone [nitrate, total dissolved solids (TDS), calcium (Ca), magnesium (Mg), dissolved inorganic carbon (DIC), strontium (Sr)]; (2) increasing concentrations or values with depth [pH, sodium (Na), Cl⁻, sulfate (SO₄²⁻), iron (Fe), manganese (Mn), zinc (Zn)]; and (3) decreasing concentrations or values with depth [Eh, DO, and silicon (Si)]. Profiles for these chemical parameters from MLS-3 at the

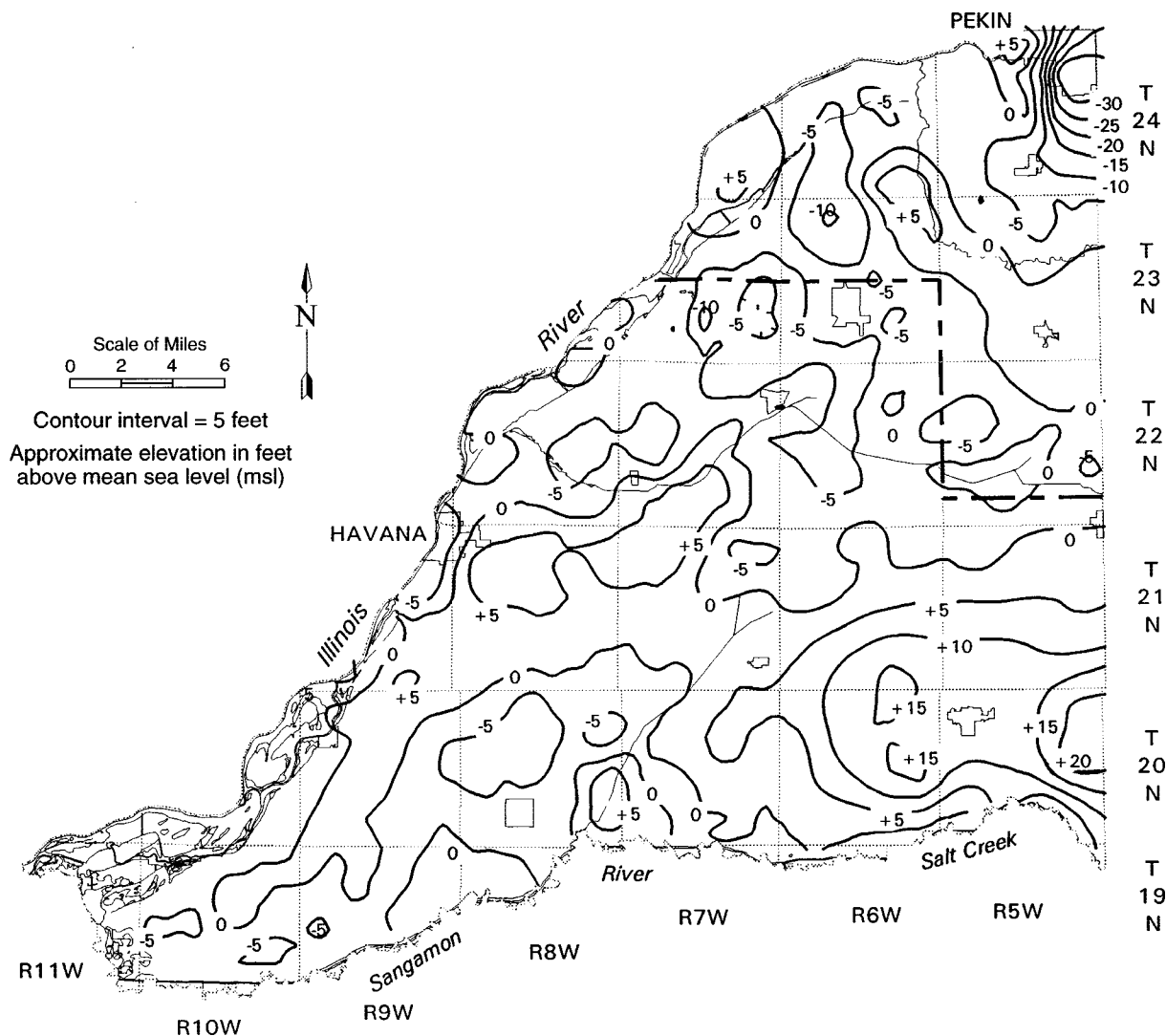


Figure 5. Change in potentiometric surface for Mason County between September 1992 and September 1993 (from Sanderson and Buck, 1995).

July 1994 sampling date are shown in figure 6. The following discussion will focus on these data unless otherwise specified.

The nitrate profile can be divided into three zones (figure 6a). Near the water table, concentrations were low. Between 20 and 35 to 40 ft below the surface, concentrations were elevated, occasionally exceeding the MCL. Below 35 to 40 ft, nitrate was generally below detection. These three zones represent areas at which recharge and/or chemical and biochemical reactions are different. The chemical signatures of each zone will be discussed in separate sections.

Near the Water Table. The relative depletion of nitrate near the water table, which is atypical in agricultural settings, is hypothesized to be due to infiltration of recharge water relatively depleted in nitrate at the nursery. Irrigation and fertilization practices at the tree nursery are typically much less than for the surrounding agricultural fields; for example,

no fertilizer was applied here in 1993 (A. Stauder, personal communication, 1993). To test this hypothesis, MLS-18 was installed near the southeast boundary of the tree nursery, about 600 ft to the southeast and hydraulically up-gradient from the other MLSs (figure 3), and sampled on the April 1994 date. The MLS-18 is about 20 ft down-gradient from fields planted in corn or soybeans. Nitrate concentrations in MLS-18 tended to be greater in the shallower ports than in MLS-3 (figure 7a). By plotting the concentrations from MLS-18 at depths lower than actuality, we can examine how the nitrate profile might look at MLS-18 if a layer of recharge water is added at the water table. When the MLS-18 data are plotted 10 ft below their actual depths, nitrate concentrations from MLS-3 and MLS-18 appear very similar for the top 30 ft below the surface; MLS-18 has elevated concentrations at deeper depths (figure 7b).

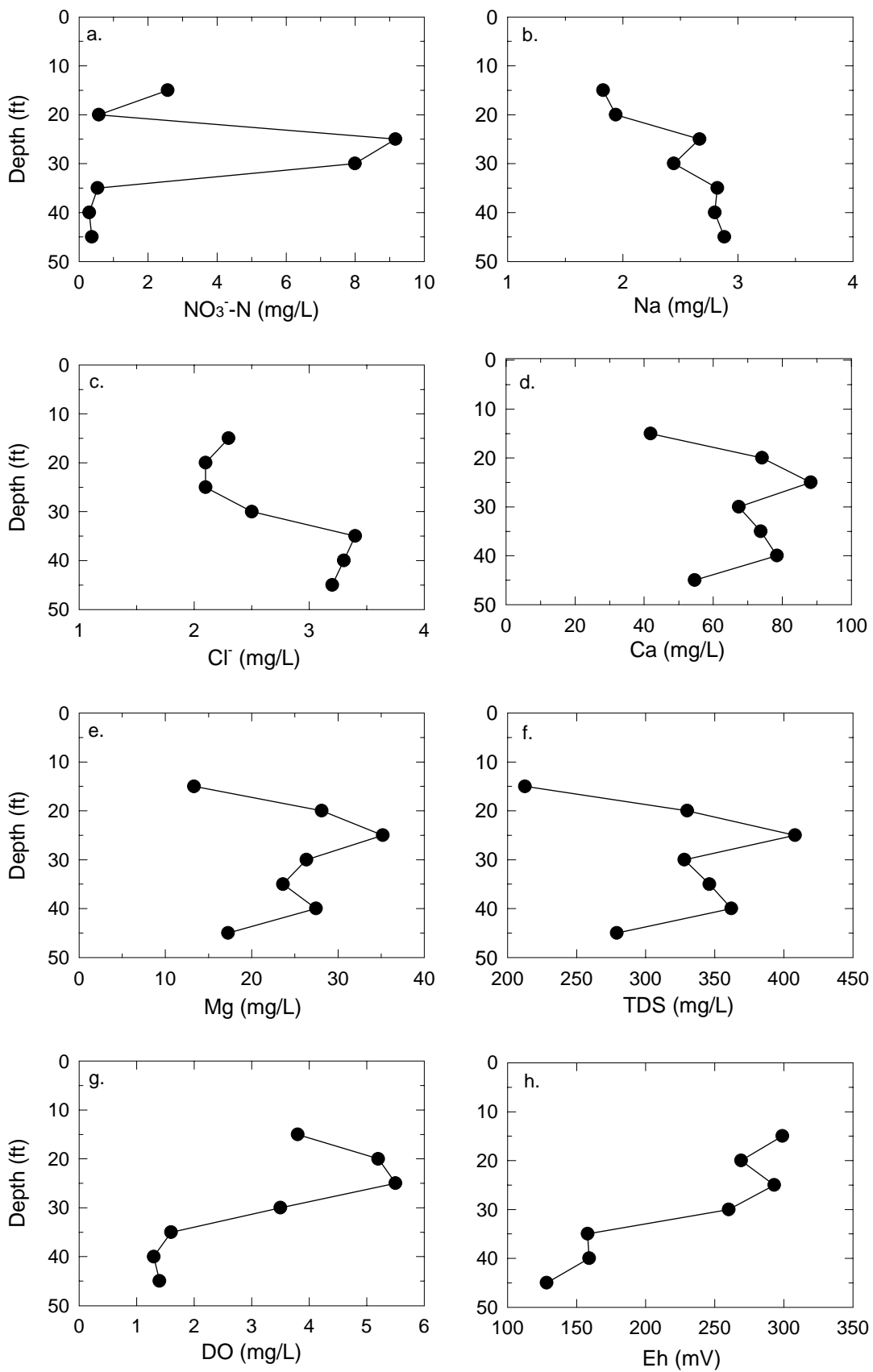


Figure 6. Concentrations of selected chemical parameters vs. depth from surface for MLS-3, July 1994: a. nitrate, b. sodium, c. chloride, d. calcium, e. magnesium, f. total dissolved solids, g. DO, and h. Eh.

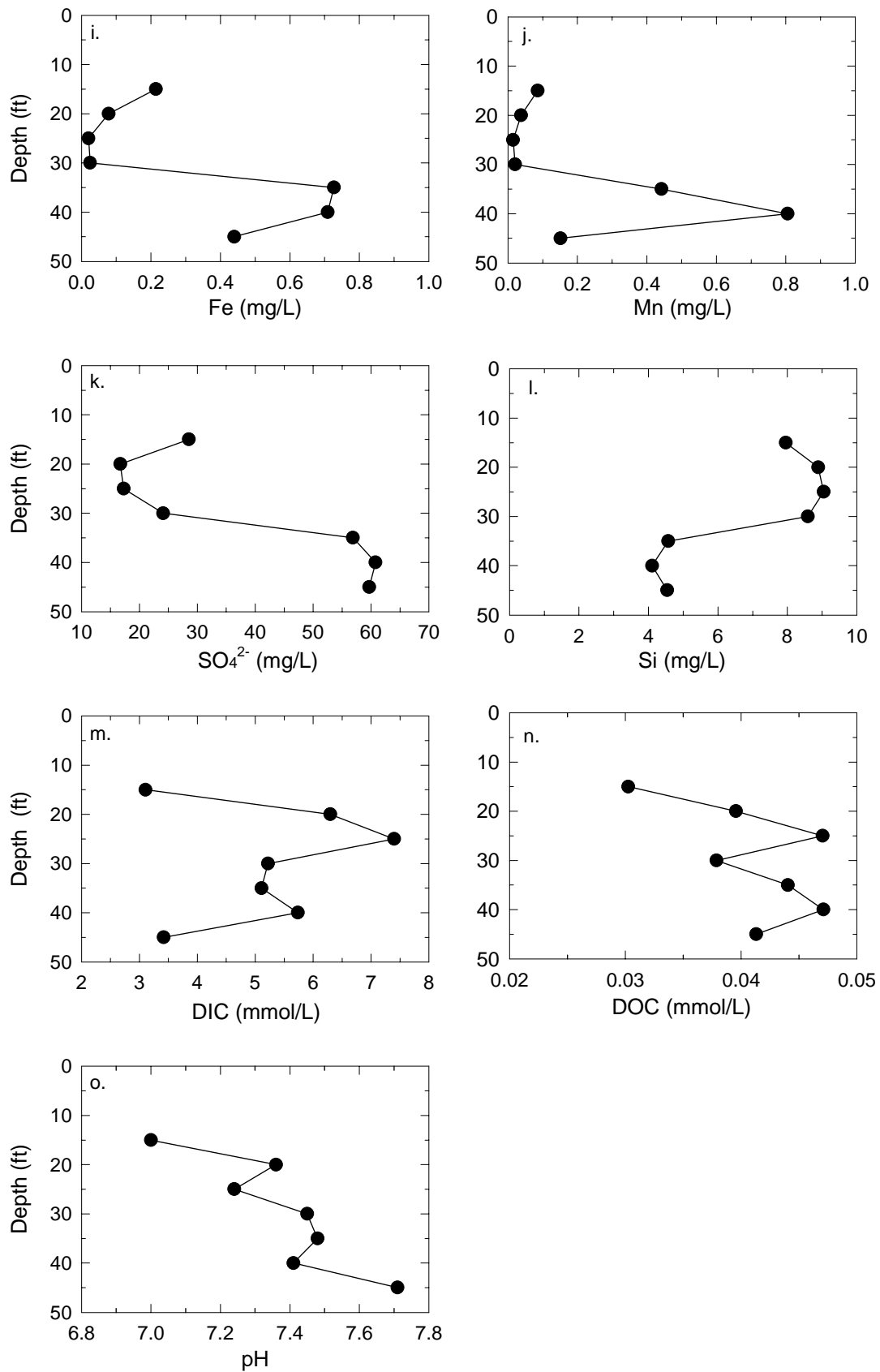


Figure 6 (concluded). Concentrations of selected chemical parameters vs. depth from surface for MLS-3, July 1994: i. iron, j. manganese, k. sulfate, l. silicon, m. dissolved inorganic carbon, n. dissolved organic carbon, and o. pH.

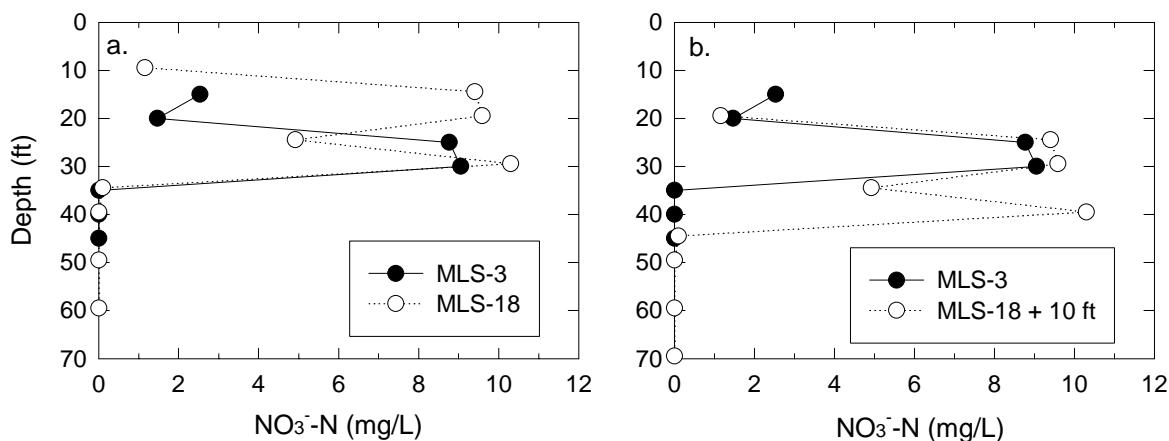


Figure 7. Nitrate concentrations vs. depth from surface for MLS-3 and MLS-18, April 1994:
 a. actual data and b. MLS-18 data plotted 10 feet deeper than actual.

Elevated concentrations of nitrate are found near the water table beneath fertilized fields, as expected; but recharge on the tree nursery fields low in nitrate causes a lens of water relatively depleted in nitrate to form near the water table. These results are similar to observations made by Postma et al. (1991), who found nitrate to be depleted near the water table beneath wooded land that was down-gradient from fertilized fields. In fact, plumes containing many types of contaminants are commonly observed to exhibit this behavior in unconfined aquifers. The fact that this lens of water is low in nitrate also suggests that atmospheric deposition is not an important source of nitrate in this area.

The fact that this lens has a thickness of approximately 10 ft less than 800 ft from agricultural fields up-gradient indicates how substantial recharge to the aquifer from precipitation is in this area. Assuming a ground-water velocity of about 0.6 ft/day, determined in tracer tests at the site, it would take approximately 3.6 years for a particle to travel from MLS-18 to MLS-3. During the 3.6 years prior to the April sampling date, approximately 160 inches of precipitation fell at the site. If the 10-ft thick lens is solely due to recharge from precipitation and irrigation is ignored, approximately 75 percent of the precipitation that occurred during the 3.6-year period recharged the aquifer. There are no evapotranspiration data for the tree nursery, but the average rate for soybeans in Mason County is about 8 inches/year (Bowman and Kimpel, 1991). There is very little overland runoff in the sandy soils in Mason County, thus the estimated recharge percentage of 75 percent seems reasonable.

This dilution at the water table between MLS-3 and MLS-18 also is observed for most of the other inorganic parameters. For many parameters, concentrations near the water table are much higher in MLS-18 than MLS-3. Calcium, Mg, SO₄²⁻, DIC, TDS, and Sr are all approximately twice as high in the uppermost part of MLS-18 versus MLS-3. Iron and manganese are about five and ten times higher, respectively. Dissolved organic carbon (DOC) concentrations are 10 times higher, and the pH is higher by about 0.5 pH units. However,

Eh and DO are lower in MLS-18. Although the decrease in concentrations for most of the parameters is likely due to dilution by dilute recharge water, the increase in DO may contribute to the loss of iron and manganese from solution by promoting mineral precipitation.

These results suggest several things. Conditions are more reducing near the water table next to the agricultural fields up-gradient from the tree nursery than at the tracer plot. This may reflect more intense biological activity in the agricultural fields up-gradient of the field site due to denser plantings and greater applications of fertilizer. Increased amounts of organic matter and nutrients stimulate microbiological activity, which depletes oxygen and makes conditions more reducing. Agricultural activities may also be contributing to increased levels of soil organic nitrogen mineralization, although the organic content of the soil is low compared to most other soils in Illinois. In addition to there being a source for nitrate in this area, there are also sources for many other major ions. Possible sources include irrigation water, fertilizers, and/or reactions in the unsaturated zone. The vertical profiles for many of the parameters do not match well between MLS-3 and MLS-18, even when the MLS-18 data are plotted 10 ft deeper, as with the nitrate data. It is obvious that there are mechanisms other than dilution affecting water quality in the aquifer. These include redox reactions, mineral dissolution/precipitation reactions, and spatial and temporal variability in the chemistry of the recharging water.

At the tree nursery, water recharging the aquifer should have the composition of rainwater that has reacted with materials in the unsaturated zone. In the agricultural fields, the recharge water composition should include a significant component of deep ground water. Water recharging most unconfined aquifers starts out as precipitation that migrates through the unsaturated zone. In this region, however, a significant portion of the recharging water starts out as irrigation water that has been pumped up from depth, generally greater than 70 ft beneath the land surface (Bowman and Kimpel, 1991). In Mason County, an average of 15 inches

of water is applied to the fields annually by irrigation; thus almost 30 percent of the water striking the land surface in irrigated fields during a year with average precipitation (37 inches) is from irrigation. This percentage is quite variable and is higher during drought years and lower during wet years. For example, during the drought year of 1988, the average amount of irrigation water applied was 23 inches (Bowman and Kimpel, 1991); however, many farmers did not irrigate in the flood year of 1993.

Precipitation is chemically dilute; most parameters typically have concentrations less than 1 mg/L (table 4). In addition, precipitation is undersaturated with respect to most common minerals. However, the irrigation water, which usually comes from fairly deep parts of the aquifer (> 60 ft), has greater concentrations of most chemical parameters. Water samples taken from the deepest port of MLS-11 (79 ft below the land surface) have an average chemical composition as shown in table 4. Most irrigation wells in Mason County are at least 70 ft deep (Bowman and Kimpel, 1991).

The composition of the water striking the land surface is important because the composition controls the identity and rates of chemical and biochemical reactions that occur in the

subsurface, such as mineral dissolution, ion exchange, biodegradation, etc. This variable composition of the recharge water complicates interpretation of the ground-water quality data.

The geochemical reaction-path code PHREEQE (Parkhurst et al., 1980) was used to simulate mixing of 30 percent deep aquifer water with 70 percent rainwater to account for the average annual input from irrigation. Chloride, which behaves conservatively in sand-and-gravel aquifers, has a mixed concentration of approximately 2.3 mg/L, which is close to the concentration found in the upper ports of MLS-18. Sodium, which also may behave conservatively in this system, has a mixed concentration of about 1 mg/L, slightly less than that observed for MLS-18. For the rest of the parameters, however, the simulation underpredicts the concentrations found in the uppermost ports of MLS-18. Therefore, simple mixing of irrigation water and rainwater cannot account for the observed composition of water near the water table.

Simulations also were run using the water composition from the top port of MLS-3 instead of rainwater. The rationale for this was that rainwater composition is highly altered during migration through the unsaturated zone, thus a water composition from MLS-3 would be more representative of the composition of water recharging the aquifer without irrigation contributions. The mixed water more closely resembles MLS-18 for this simulation but underpredicts the concentrations of most parameters, especially sulfate, alkalinity, and manganese. It appears that some other mechanisms must account for the observations. One obvious explanation is that there is considerable variability in the water chemistry in the aquifer, even at depths of 80 ft.

The mixed solutions from both simulations are calculated to be undersaturated with respect to calcite (CaCO_3) and dolomite [$\text{CaMg}(\text{CO}_3)_2$]. These common minerals are not specifically identified for these deposits, although it is likely that they do exist. If so, they would be expected to be dissolving and contributing Ca, Mg, and DIC into solution.

Saturation indices also were calculated for other water compositions using the thermodynamic geochemical code WATEQF (Plummer et al., 1976). The water sample from the shallowest port in MLS-3 is undersaturated with respect to calcite and dolomite. All other samples from MLS-3 are saturated with respect to these minerals. Similar results were found for MLS-18, except the shallowest port was undersaturated only with dolomite. Thus in the unsaturated zone, and/or a narrow zone just below the water table, enough calcium and magnesium are put into solution that dissolution of calcite and dolomite is no longer thermodynamically favored.

Nitrate Plume. The only significant sources of nitrate in this area are from fertilizer applications and possibly mineralization of soil organic nitrogen. The absence of nitrate in the lens near the water table at the nursery indicates that atmospheric deposition is not significant, and there are no sources of manure or septic effluent. The results from MLS-18 indicate that fertilizer applications in fields up-gradient

Table 4. Average Chemical Composition of Precipitation and Water Samples from MLS 11-1*

<i>Parameter</i>	<i>Rain</i>	<i>MLS 11-1</i>
Ca	0.26	62.8
Mg	0.031	17.1
Na	0.056	3.07
K	0.018	1.03
Cl ⁻	0.11	7.47
SO ₄ ²⁻	2.10	40.5
Alkalinity	**	189
DIC	**	4.15
Fe	**	0.49
Mn	**	0.22
pH	4.70	7.32 [†]
NO ₃ ⁻ -N	0.34	0.30
NH ₄ ⁺	0.45	0.04
Si	**	5.98
Sr	**	0.09
Al	**	<0.01
Ba	**	0.08
Zn	**	0.01
TDS	**	264
DOC	**	1.5
DO	**	1.4
Eh	**	123.9

Notes: *Alkalinity is as mg/L CaCO₃, DIC is in mmol/L, Eh is in mV, all other concentrations except pH are in mg/L. Precipitation data is from an ISWS sampling station in Monmouth, about 50 miles northeast of the study site.

** = Not reported.

[†] = Median value.

from the study site probably account for the nitrate plume found in the ground water.

Relationships between nitrate concentrations and other chemical parameters can be seen to some extent by comparing the depth profiles in figure 6. The correlations between nitrate and the other chemical parameters were calculated to determine if the concentrations of the other parameters could be related to nitrate. Because nitrate concentrations were below detection in some of the samples, a nonparametric test, the Spearman rank order correlation, was used (Helsel and Hirsch, 1992). Fairly strong positive correlations ($r_s > 0.6$) were found between nitrate and Mg, Sr, Ca, DIC, Si, DO, and TDS for the July 1994 samples (table 5). Fairly strong negative correlations ($r_s < -0.6$) were found between nitrate and pH and Fe. The worst correlations were for DOC, Na, potassium (K), and SO_4^{2-} . Similar results were found for all the sampling dates.

Some of these correlations are readily explained. For example, nitrate is a redox-sensitive species, and it would be expected to be correlated to some extent with other redox-sensitive species, such as DO and Fe. Nitrate concentrations also are influenced by biodegradation reactions; thus its correlation with DIC and pH, which are often affected by biodegradation reactions, is not unexpected. However, nitrate

was not correlated with DOC, another parameter affected by biodegradation.

The reason for the positive correlations between nitrate and Ca, Mg, Si, and Sr is most likely due to impurities in the fertilizers. Locally used fertilizers have elevated levels of many ions, including Ca, Mg, K, Na, Si, Cl⁻, SO_4^{2-} , and a number of trace elements (S. Panno, unpublished data, 1995).

Deep Aquifer. Low levels of nitrate at depth are not unexpected; this observation has been made in most unconfined aquifers. The two most likely explanations are that conditions at depth are conducive to denitrification reactions, or there has been inadequate time for nitrate to reach these depths.

The nitrate plume is probably dominated by aerobic bacteria, where detectable levels of DO are present. There is a sharp zonation in the water chemistry at the base of the nitrate plume, where the disappearance of nitrate occurs in conjunction with decreases in DO and Eh values (figures 6g and 6h) and the appearance of significant concentrations of dissolved Fe, Mn, and SO_4^{2-} (figures 6i, 6j, and 6k). These changes indicate a redoxcline, below which the redox conditions are significantly more reducing; below the redoxcline, nitrate is unstable and removed via denitrification or other nitrate reduction reactions. Both Eh and DO decrease with depth, and little or no nitrate is found in samples with low Eh (figure 8a). The accuracy of the DO results is somewhat uncertain because samples were taken using a peristaltic pump, which may cause degassing of samples (Rose and Long, 1988), and the samples were not shielded from the atmosphere; however, the fact that DO concentrations correlated with Eh values indicates that the DO data trends are accurate. The reason for the relative depletion of DO near the water table is uncertain, although it may have to do with the difference in crop type at the tree nursery compared with surrounding fields.

Dissolved Fe and Mn concentrations also indicate reducing conditions at depth (figures 6i and 6j). Both of these metals tend to form insoluble oxyhydroxide solid phases under oxidizing conditions but are much more soluble under reducing conditions (Stumm and Morgan, 1981). The change in color from reddish-brown to gray sand at depths about 40 ft suggests that iron reduction is occurring. Because iron-reduction reactions occur under conditions more reducing than nitrate-reduction reactions, the thermodynamics of the system are favorable for denitrification. A plot of nitrate versus dissolved iron for all samples taken at the July 1994 sampling date shows the relationship between these two parameters; the presence of one is usually marked by the absence of the other (figure 8b).

Nitrate may be denitrified in the presence of several terminal electron acceptors, most commonly organic carbon or sulfide (Appelo and Postma, 1993). If organic carbon is the primary electron acceptor, the disappearance of nitrate should occur in conjunction with a decrease in DOC and an increase in carbon dioxide (CO_2), which in turn might affect other geochemical reactions (e.g., decreased pH and increased solubility of carbonate minerals). If sulfide is the primary

Table 5. Spearman Rank Correlation Coefficients (r_s) and P Values for Nitrate with Other Chemical Parameters, July 1994*

<i>Parameter</i>	r_s	<i>P</i>
pH	-0.67	0.0
Ca	0.66	0.0
Mg	0.67	0.0
Na	-0.37	0.04
K	-0.45	0.01
Cl	0.53	0.0
SO_4^{2-}	-0.43	0.01
DIC	0.66	0.0
Fe	-0.66	0.0
Mn	-0.60	0.0
Al	0.09	0.65
Ba	0.09	0.63
Si	0.66	0.0
Sr	0.67	0.0
Zn	0.11	0.55
TDS	0.64	0.0
DO	0.21	0.0
Eh	0.59	0.0
DOC	0.65	0.24

Note: *Larger absolute r_s values indicate stronger correlations between variables. P values measure the probability of wrongly concluding that there is a true association between the variables (falsely rejecting null hypothesis). P values > 0.05 generally indicate there is no significant relationship.

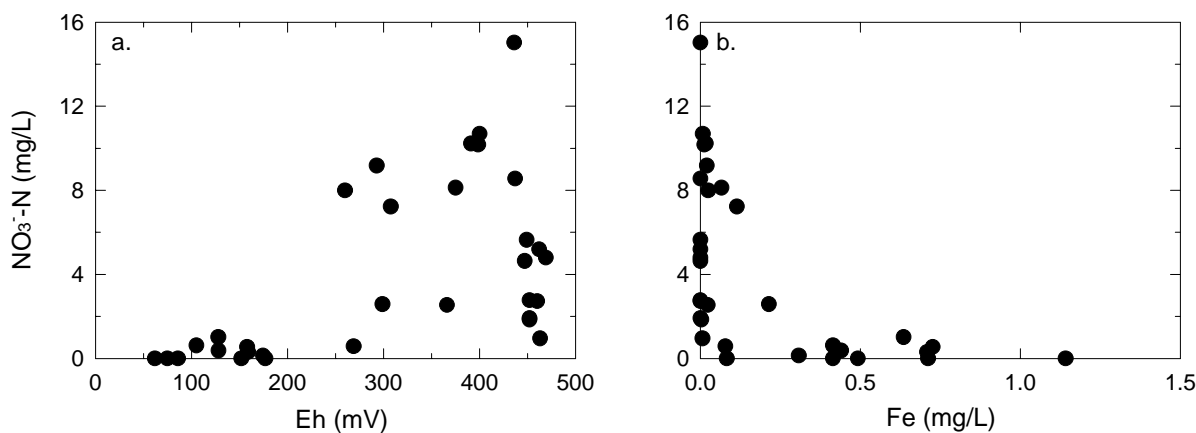


Figure 8. Nitrate vs. a. Eh and b. iron for MLS-3, MLS-7, MLS-11, and MLS-16, July 1994.

electron acceptor, then increases in iron and sulfate and a decrease in pH should be observed (van Beek et al., 1989).

The main limitation on denitrification rates in ground water is typically the lack of metabolizable organic matter (Starr and Gillham, 1993). This is especially true in well-sorted sand aquifers, in which amounts of organic matter tend to be small. Although DOC levels are not high in this aquifer, there appears to be sufficient amounts for denitrification (figure 6n). The DOC concentrations in the shallower parts of MLS-18 were much higher than for any of the other MLSs sampled (figure 9). The loss of DOC as ground water migrates is indirect evidence that biodegradation reactions are occurring.

The DIC concentrations, a measure of carbon dioxide production, also are higher down-gradient from MLS-18. There is a slight increase in DIC beneath the redoxcline in MLS-18. However, there was no increase in DIC at the base of the nitrate plume in MLS-3 in July 1994 (figure 6m), which would be expected if organic carbon was being oxidized to carbon dioxide. The DIC concentrations were slightly higher below the plume at other sampling dates, although concentrations always tended to be relatively low in the deepest ports (> 40 ft). The DIC concentrations tend to be greatest in the nitrate plume and just below it, suggesting an active bacterial population in this zone. The DOC and DIC concentrations are correlated ($r^2 = 0.48$), suggesting increased microbial activity in zones in which DOC levels are greatest. However, DIC is more strongly correlated with calcium and magnesium, with r^2 values of 0.95 and 0.95, respectively. These two cations have a high correlation with each other ($r^2 = 0.96$) and are commonly associated. The production of DIC decreases the pH and increases the solubility of carbonate minerals, such as calcite and dolomite. Below the nitrate plume, however, there is no increase in calcium and magnesium concentrations (figures 6d and 6e), and there is no significant change in saturation indices for carbonate minerals.

The possibility of denitrification in conjunction with sulfide oxidation is supported by the sulfate data, which

increase sharply at the base of the nitrate plume and at a rate consistent with the amount of nitrate removed (figure 6k). The increase in iron is much smaller (figure 6i), although it may be removed from solution by precipitation as oxyhydroxide minerals. Using the concept of electron equivalents developed by Postma et al. (1991), sulfide oxidation as measured by production of sulfate can account for between 50 and 95 percent of the nitrate reduced across the redoxcline for the various MLSs and sampling times.

However, sulfate concentrations decrease between MLS-18 and MLS-3, when they should be increasing if sulfide oxidation is occurring (figure 6k). Sulfate would not be expected to be sorbed significantly in a sand aquifer. Sulfate reduction is a potential sink for sulfate, but no sulfide was ever detected in these samples, and iron concentrations would be expected to be very low at depth if sulfide was being produced. The decrease in sulfate may be due to dilution effects as discussed previously; a significant source of sulfate to the aquifer is likely from impurities in fertilizers applied

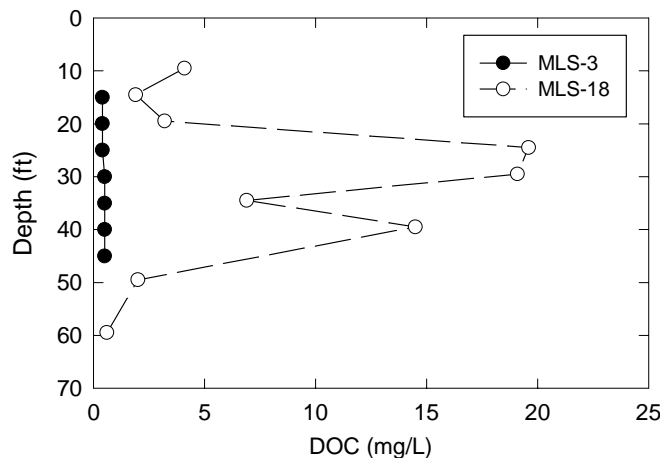


Figure 9. Dissolved organic carbon concentrations vs. depth from surface for MLS-3 and MLS-18, April 1994.

up-gradient of the tree nursery. Fertilizers commonly used in the area have high levels of sulfate, up to 10 percent by weight (S. Panno, unpublished data, 1995).

There are further drawbacks to the sulfide oxidation mechanism. The pH does not decrease below the nitrate plume, but increases slightly (figure 6o). Pyrite has not been identified in the aquifer material in this or other investigations. The gravel-sized coal that is especially abundant 35 to 40 ft beneath the surface, the approximate depth of the redoxcline, could be a source of sulfur, and sulfide minerals are commonly associated with coal.

There is evidence from the redox data that conditions for denitrification reactions are favorable, but the data are not unequivocal and mechanisms cannot be defined. It is possible that both organic matter and sulfide are being used as terminal electron acceptors during denitrification. In order to assess the second possibility for absence of nitrate at depth, i.e., that nitrate has had insufficient time to reach depth, tritium analyses were performed. In April 1994, samples were taken for tritium analysis to determine the relative age of the waters and estimate how rapidly water moves through the aquifer. Tritium is a radioactive isotope of hydrogen with a half-life of 12.3 years, and it has been used to date relatively young materials. It is produced in the atmosphere and enters ground water during recharge events. Tritium has been useful in hydrogeological studies in determining the presence or absence of recent water (water less than 50 years old) in an aquifer. This information helps in determining recharge areas and how rapidly water is transported to depth in an aquifer (Freeze and Cherry, 1979).

Before 1953, tritium concentrations in the atmosphere were between 5 and 10 tritium units (TU) (1 TU = 1 tritium atom per 10^{18} atoms of hydrogen). Water that entered an aquifer prior to 1953 contained levels less than 1 TU in 1994. If the source of tritium had been constant, levels should decrease the deeper or further down-gradient in an aquifer one goes. But, between 1953 and 1969, atmospheric testing of thermonuclear devices caused large amounts of tritium and other radioactive gases to be injected into the atmosphere. The highest historic levels of atmospheric tritium were measured in the early 1960s, peaking at over 6000 TU (figure 10). Since then, levels have decreased to about 10 TU at present in Illinois (Hackley et al., 1996). Based on the decay rate alone, water entering an aquifer in the early 1960s would still contain tritium levels in excess of 1000 TU. However, due to dispersion and mixing, tritium concentrations are generally much lower. Based on work by Daniels et al. (1989) in Indiana ground water, present tritium concentrations of water that recharged during the early 1960s would be about 75 TU (K. Hackley, personal communication, 1995).

Tritium was detected in all samples taken, even to a depth of nearly 80 ft below the surface (figure 11). Concentrations decrease from the water table to about 20 ft below land surface, then tend to increase with depth at greater depths. There is no observable "bomb pulse." All the samples tested appear to be less than 20 years old, indicating that rapid transport of solutes to considerable depths has occurred in this area. These results

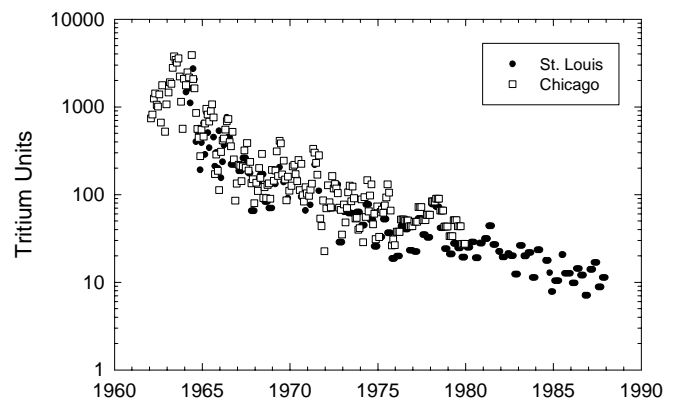


Figure 10. Tritium concentrations in precipitation from Chicago and St. Louis stations between 1960 and 1988 (from IAEA, 1994).

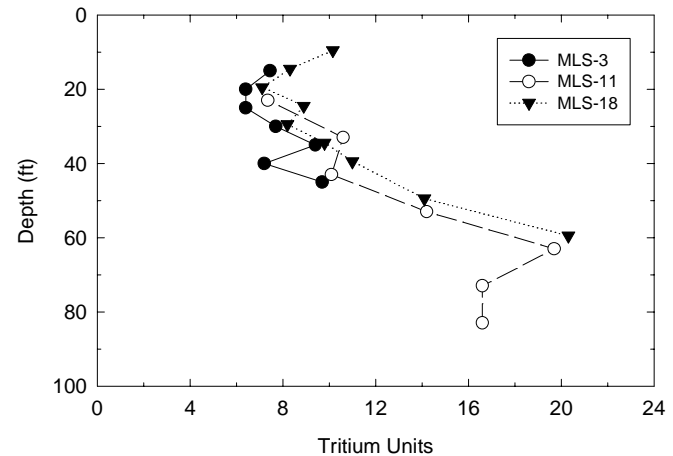


Figure 11. Tritium concentrations vs. depth from surface for MLS-3, MLS-11, and MLS-18, July 1994.

suggest that the widespread irrigation pumping in this region may be influencing solute transport in the aquifer.

Tritium is found throughout the water column but not nitrate, which suggests that denitrification is the principal factor restricting migration of nitrate to depth in this aquifer. This hypothesis is supported by the redox data. However, elevated levels of other parameters in the plume, such as calcium and magnesium, confound this interpretation. The concentrations at depth for calcium (~60 mg/L) and magnesium (~20 mg/L) are less than those found in the nitrate plume. If water is rapidly moving vertically through the aquifer as indicated by the tritium results, either there must be some process removing them at depth or their presence in the plume represents a fairly recent addition via recharge. The ground water is supersaturated with respect to calcite throughout the water column, except near the water table; but it is not clear why calcite precipitation would be enhanced at the base of the plume because the reaction is not affected by redox conditions.

Lateral Variability

In addition to the vertical variability in water chemistry, lateral variability also quite often was striking. The MLSs were sampled for nitrate in November 1993 (MLS-1 to MLS-5) and November 1994 (all MLSs). These data were plotted in cross sections to produce two- and three-dimensional pictures of nitrate concentrations (figures 12 and 13). The high-nitrate zone is readily apparent, but considerable variability can be observed. This is especially apparent in two of the profiles from November 1994. In figure 13b, three MLSs had no samples above 2 mg/L, but the other two had two each above 4 mg/L. The absence of elevated levels of nitrate in the other MLSs may be because none of their ports are located in the elevated zone, or for some reason the nitrate plume is confined to a narrow zone. The MLS-12 to MLS-15

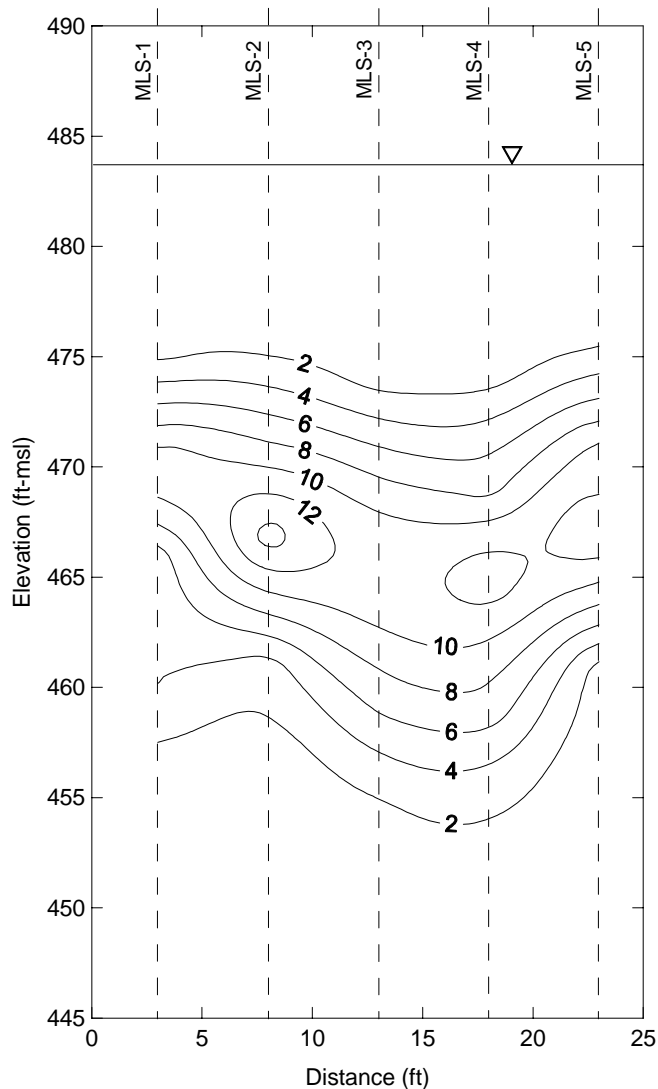


Figure 12. Vertical cross sections of nitrate concentrations as a function of depth for MLS-1 to MLS-5, November 1993. Contour interval 2 mg/L NO_3^- -N. No vertical exaggeration.

have ports every foot, and this fine resolution allowed considerable variability to be observed (figure 13a). It seems reasonable to conclude that the migration of solutes is significantly affected by heterogeneities in the aquifer.

Four MLSs were sampled in July 1994; three (MLS-3, MLS-7, and MLS-16) are within 20 ft of each other. The other, MLS-11, is approximately 80 ft east of MLS-3. The MLS-3 and MLS-7 have sample ports at approximately the same depths. Chemical trends for these two MLSs were generally the same, but there are some differences. For example, nitrate concentrations were greatest for both MLSs between 25 and 30 ft below land surface, but concentrations in MLS-7 were considerably greater for the 10 ft below this zone in MLS-7 than MLS-3 (figure 14a). Most of the other chemical parameters were similar for the two MLSs (see graphs in appendix C).

Multi-level sampler-16, which is approximately halfway between MLS-3 and MLS-7, samples the upper part of the aquifer with ports that are closer together than those in MLS-3 and MLS-7 (table 2). Chemical trends are similar for MLS-16, although more variability is observed because of the greater depth resolution. For example, between about 10 and 20 ft below the surface, nitrate concentrations fluctuate, and a higher peak concentration is observed than for the other two MLSs. This variability also is evident for most other parameters and reflects the common observation that, the more closely spaced the samples, the more heterogeneities and variability are observed.

For some parameters, concentrations in MLS-16 were significantly different than for MLS-3 and MLS-7 (appendix C). For example, DOC concentrations were quite a bit higher in MLS-16, sometimes twice the value for MLS-3 and MLS-7. Peak concentrations were highest in MLS-16 for several other parameters, including Ca, Mg, TDS, and DIC. The Eh values also were highest in MLS-16 (figure 14b), but DO concentrations decreased at shallower depths than for MLS-3 and MLS-7 (figure 14c).

The water-chemistry differences among MLS-3, MLS-7, and MLS-16 were relatively minor compared to differences with MLS-11. The MLS-11 has deeper and more widely spaced (10 ft) ports than the other three. Perhaps the most striking difference is that nitrate was not detected or had very low concentrations in all ports, including those that straddle the nitrate plume in MLS-3, MLS-7, and MLS-16 (figure 14a). The waters from MLS-11 also were significantly more reducing than for the other MLSs, as indicated by Eh and DO values (figures 14b and 14c). It appears that either denitrification reactions occur at much shallower depths in this region of the aquifer, or there is no source for a nitrate plume up-gradient from MLS-11. The latter explanation seems unlikely because the farm field up-gradient of MLS-11 is the same as that up-gradient of MLS-3 and MLS-7.

Other differences in water chemistry for MLS-11 included higher pH and lower Ca, Mg, TDS, and DIC concentrations (appendix C). The other chemical parameters had trends similar to the shallower MLSs.

These differences illustrate the chemical and physical heterogeneities in a system such as this one. Although we

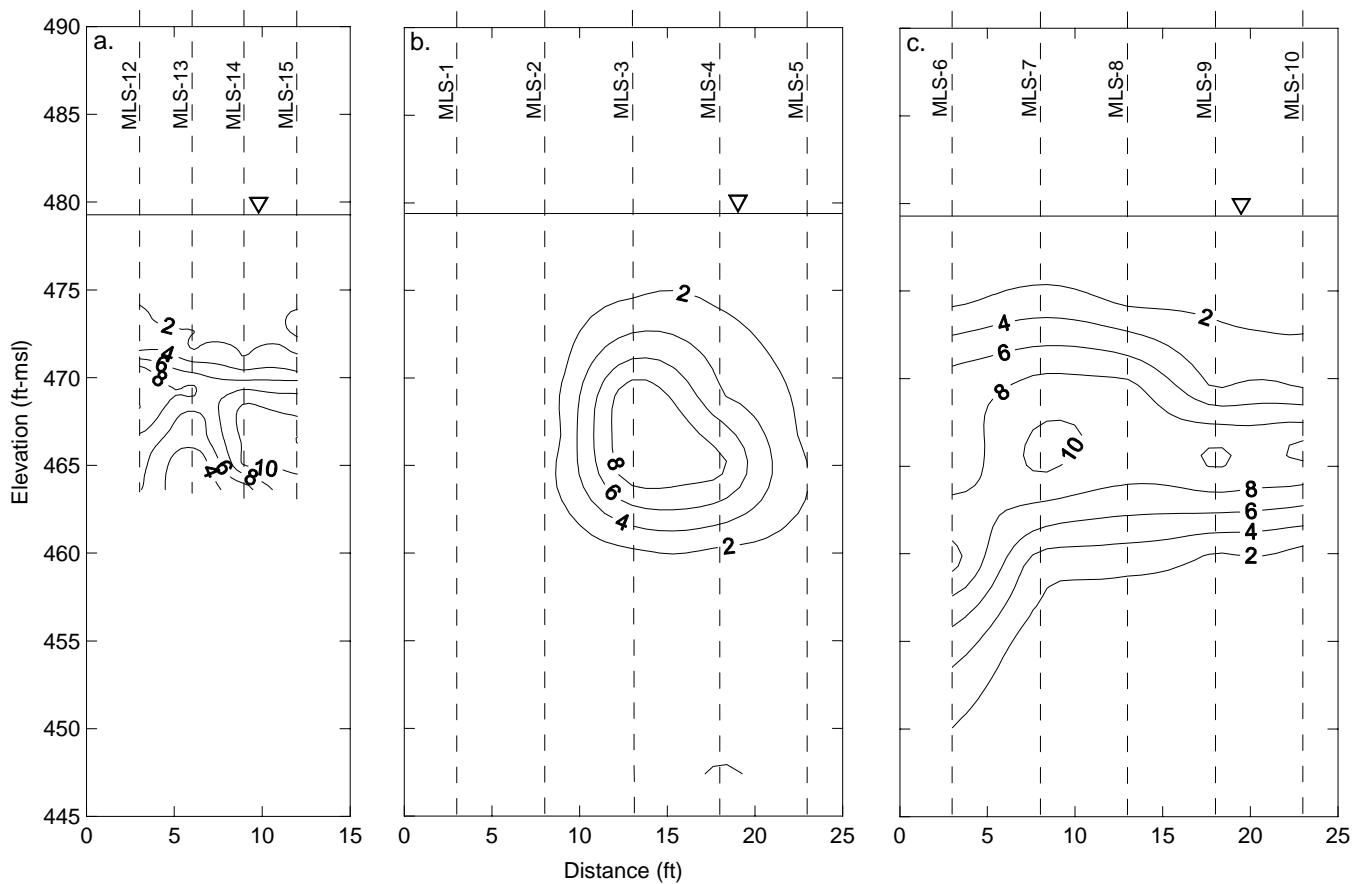


Figure 13. Vertical cross sections of nitrate concentrations as a function of depth for a. MLS-12 to MLS-15, b. MLS-1 to MLS-5, and c. MLS-6 to MLS-10, November 1994. Contour interval 2 mg/L NO_3^- -N. No vertical exaggeration.

might expect the chemical composition of precipitation and irrigation water to be relatively consistent, both temporal and spatial variability in fertilization inputs, soil organic nitrogen mineralization rates, and irrigation rates will lead to ground-water compositions that are not well mixed. In addition, spatial variability in hydraulic conductivity leads to preferred flow paths and variable ground-water velocities.

Temporal Changes

Water quality samples were taken quarterly between July 1993 and July 1994. The results discussed in this section will focus on MLS-3.

Concentration profiles for each MLS were similar for all the parameters throughout the study period, although actual concentrations for some of the parameters varied considerably. Nitrate in some ports varied by as much as 100 percent (figure 15a). Redox conditions also varied temporally. The Eh and DO were higher in fall and summer than in winter and spring (figures 15b and 15c). However, iron and manganese concentrations were greater in the winter and spring, especially winter (figure 15d and 15e).

Other parameters also varied over the seasons. Calcium, Mg, TDS, and DIC were higher in the winter and spring, sometimes over twice as high. Sodium, Cl^- , SO_4^{2-} , and pH all exhibited some temporal variability, although it was less well defined than the other parameters (appendix C).

Because of extremely wet conditions in 1993, little irrigation pumping was done that year, thus irrigation cannot be used to explain variability in water chemistry for 1993. The significant temporal variability probably is an illustration of the large impact agricultural practices have on ground-water chemistry in this region. For example, fertilizer is applied at specific times of year, and rapid percolation of precipitation and irrigation water to the water table suggests that the input of nitrate to ground water will be quite variable over the course of a year, a function of land-use activities and precipitation/irrigation events. Thus one might expect the water chemistry of migrating ground water to be quite variable in composition, especially for parameters that are affected by human activity.

Another possibility for changes in nitrate and redox conditions over time is the change in bacterial activity. Bacterial metabolism is affected by temperature, generally

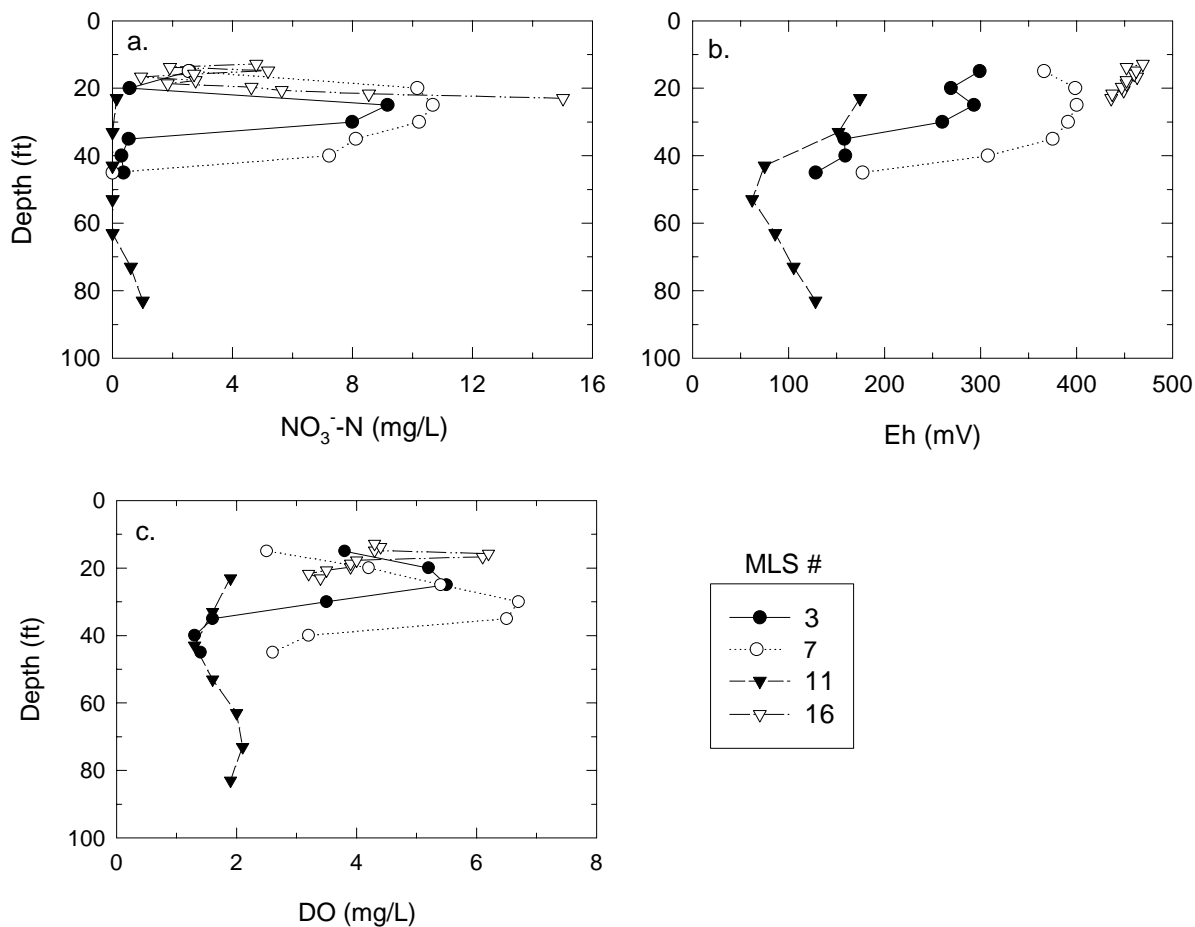


Figure 14. a. Nitrate, b. Eh, and c. DO vs. depth for MLS-3, MLS-7, MLS-11, and MLS-16, July 1994.

increasing with increasing temperature. Therefore, one would expect greater reaction rates in warmer weather and lower rates in winter. This is commonly observed in soils (Staver and Brinsfield, 1990) and has been reported in some shallow ground waters (Hallberg, 1987; Littke and Hallberg, 1991). However, at the tree nursery the data are equivocal for this hypothesis. Although the highest nitrate concentrations are observed for some ports in the winter, the highest concentrations are observed in July for others (figure 15a). The fact that little fertilizer is applied at the tree nursery and recharge is rapid confounds the interpretation. Because sampling was done for only one year, seasonal differences cannot be adequately assessed.

Summary of Water-Chemistry Data

The water-chemistry data strongly reflect the geology of the aquifer, soils, and agricultural activities, especially irrigation and fertilization practices. The lateral and temporal variability observed for most of the chemical parameters are most likely explained by the inconstant application of fertilizer and other agrichemicals at the surface, physical

heterogeneities in the aquifer leading to variability in permeability, and the rapid percolation of water to the saturated zone, which in turn is a function of precipitation events and scheduling of irrigation activities.

Denitrification appears to be an important process in this aquifer, limiting detectable nitrate concentrations to the upper 20 or 30 ft of the aquifer. The lateral variability highlights the heterogeneous nature of the microbial activity. In some areas (i.e., near MLS-11), denitrification appears to be important even near the water table. The lack of nitrate near the water table is probably because little fertilizer is applied at the tree nursery compared with surrounding agricultural fields, and recharge to the aquifer is rapid.

A simple conceptual model summarizing interpretation of the depth profile data, with special emphasis on nitrate, is illustrated in figure 16. Water with elevated levels of most ions probably resulting from fertilizer applications recharges the aquifer beneath cultivated fields (figure 16a). As the plume migrates down-gradient, recharge at the tree nursery is relatively low in solutes due to limited fertilizer application, causing the plume to sink (figure 16b and 16c). After the plume reaches a depth at which conditions are reducing

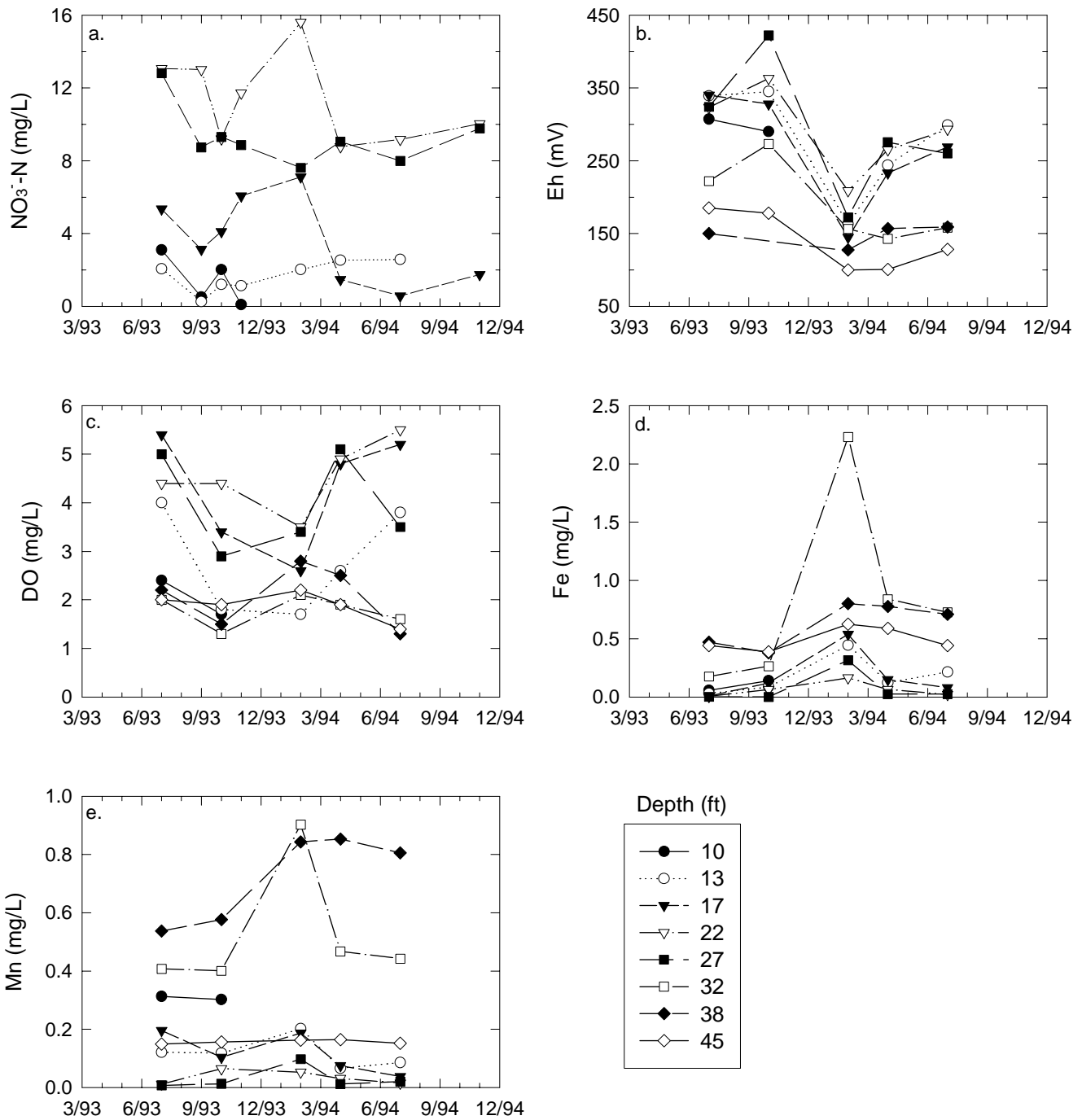


Figure 15. Temporal variation in chemical parameters for several ports of MLS-3: a. nitrate, b. Eh, c. DO, d. iron, and e. manganese at depth below land surface. Missing data from upper two ports represents times when ports were in unsaturated zone. Nitrate was not detected in lower ports.

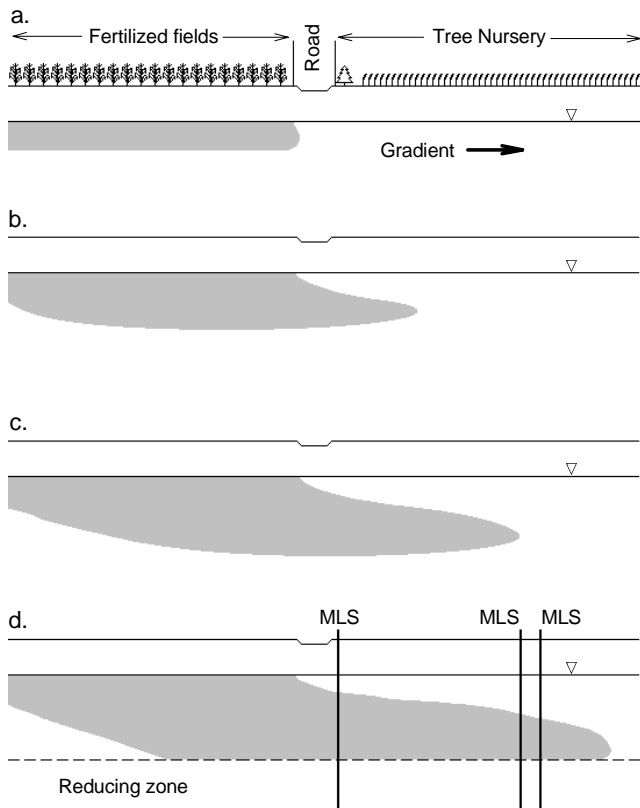


Figure 16. Cross-sectional view of simple conceptual model of water chemistry evolution at field site. Vertical exaggeration is approximately 10x.

enough so denitrification proceeds, nitrate is removed from solution and is generally not detected below this depth (figure 16d). Water with elevated levels of most ions, probably resulting from fertilizer applications, recharges the aquifer beneath cultivated fields. As the plume migrates down-gradient, recharge at the tree nursery is relatively low in solutes due to limited fertilizer application, causing the plume to sink. After the plume reaches a depth at which conditions are reducing enough so denitrification proceeds, nitrate is removed from solution and is generally not detected below this depth.

Tracer Experiments

Aquifer Properties

One goal of the tracer experiments was to determine certain aquifer properties, including hydraulic conductivity and longitudinal and transverse dispersivities. These properties are required for ground-water flow and solute transport modeling.

Hydraulic Conductivities. Naymik and Sievers (1983) estimated hydraulic conductivities at the Sand Ridge State Forest using several techniques, including the Hazen equation,

the Kozeny-Carmen equation, results from an ISWS aquifer test just north of their site using the Theis and Jacob methods, and results from published studies. Values from these methods were between 210 and 7600 gallons per day (gpd) per square foot (ft^2). Because the range of values was so great, they estimated hydraulic conductivity from a preliminary tracer experiment and calculated a value of 3000 gpd/ ft^2 . They also measured porosities between 0.25 and 0.29 at the same site and reported the drainable porosities to be between 0.23 and 0.28.

A preliminary tracer test was performed at the tree nursery under natural-gradient conditions to assess flow rates and directions and guide future installation of MLSs. This test was started on September 20, 1993. About 100 gallons of water spiked with sodium bromide to a final concentration of approximately 360 mg/L bromide, was injected by gravity into injection well I-3. To estimate the initial dimensions of the tracer plume, we assumed that the injected water advected vertically a total of 3 ft in the aquifer and that the effective porosity of the aquifer was 0.28. If the plume was initially cylindrical, it would have a radius of slightly more than 2 ft (figure 17). The irrigation pump was turned on for about 90 minutes after injection.

Breakthrough of bromide was first observed 22 days after injection in a port 22 ft beneath the surface in MLS-5 (figure 18). This was the only port at which bromide concentrations

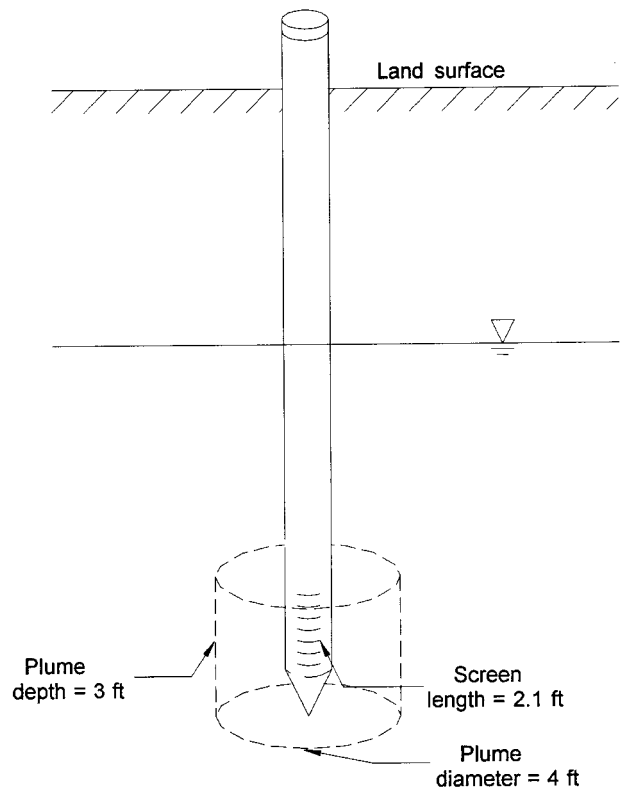


Figure 17. Idealized configuration of tracer plume.

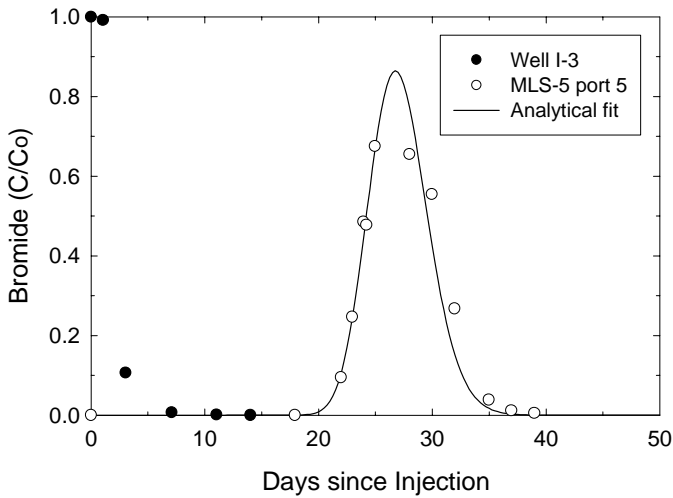


Figure 18. Bromide concentrations in MLS-5 port 5 as a function of days since injection for fall 1993 natural-gradient tracer test. Concentration of C was normalized to concentration of injected water (C_0).

above background were detected during the test, and the only port at a depth that was within the injection zone. The largest bromide concentration was observed 25 days after injection (~245 mg/L); bromide returned to background levels 45 days after injection.

The observation port data were fit using the analytical solution for the one-dimensional instantaneous injection model (Huang, 1991):

$$C(x,t) = \frac{M}{nS} \frac{1}{\sqrt{4\pi Dt}} e^{-[(xVt)^2/4Dt]} \quad (2)$$

where $C(x,t)$ is the tracer concentration at distance x and time t from the injection; D is the dispersion coefficient; V is the average linear (i.e., pore water) flow velocity; M is the tracer mass injected; n is the (effective) porosity; and S is the cross-sectional area of the flow for the injection. This model is not perfectly suited to the test data because the model is one-dimensional and assumes an instantaneous point source, but the results give a reasonable approximation of velocity and dispersion coefficient. The best model fit to the data was determined by trial and error, with $V = 0.595$ ft/day and $D = 0.052$ ft²/day, which gives a dispersivity of approximately 0.09 ft (figure 18). The maximum concentration calculated from this fit is 313 mg/L 26.9 days after injection.

Hydraulic conductivity can be determined from the tracer-test data using Darcy's Law:

$$K = \frac{nV}{dh/dl} \quad (3)$$

where K is hydraulic conductivity and dh/dl is the hydraulic gradient (head change over a distance). Using $V = 0.595$ ft/day, $n = 0.28$, and an average dh/dl of 0.0025, K is calculated to be approximately 500 gpd/ft².

Pump-test data collected from observation wells during the forced-gradient tracer test also were used to estimate the hydraulic conductivity of the aquifer. Transducers were installed in wells W-2 (located 25 feet west of MLS-7 and 60 feet east of the irrigation well), I-2, and O-2 (figure 3) during pumping and water-level data were collected every minute. Unfortunately, much of the data were lost due to equipment problems. Only recovery data from wells W-2 and I-2 were usable; K was estimated from these data.

Short duration time-drawdown data (toward the late stages of recovery) for observation well I-2 were analyzed using the Jacob method (Cooper and Jacob, 1946). The hydraulic conductivity estimated from the semi-logarithmic plot was 5500 gpd/ft², using a pumpage rate of 1200 gallons per minute (gpm) and an aquifer thickness of 110 ft. Because the recovery was not complete before the pump was turned on again, the quality of these data is questionable. The hydraulic conductivity of the sand-and-gravel aquifer in Mason and Tazewell Counties has been studied extensively. Hydraulic conductivities determined from pump tests in Illinois are stored in the Aquifer Properties Database at the ISWS. Values in the database suggest that the above estimated hydraulic conductivity falls in the upper ranges of the reported hydraulic conductivities.

The preliminary tracer test conducted in this study, as reported earlier, yielded a hydraulic conductivity of 500 gpd/ft². A potential reason for the differences between the hydraulic conductivities computed from our tracer test and those reported in the Aquifer Properties Database could be because of the gradual gradation of sand with depth. The sand at the top of the aquifer is finer than that from the pumped zone. During installation of wells and MLSs, the sand coarsened with depth; fine-to-medium grained sand was found in the top 40 ft and medium-to-coarse grained sand was found at greater depths. Naymik and Sievers (1983) intensively studied a site within the Sand Ridge State Forest approximately 5 miles north of the tree nursery. They observed nearly 30 ft of wind-blown fine-to-medium sand on the top of the coarser sand and gravel. The ISWS geologic logs for many sites in this area show a higher percentage of fine sand near the surface than at depth (S. Burch, personal communication, 1995).

Hydraulic conductivities also were estimated from transmissivity, and specific yield values were calculated from the pump test data. The calculated recovery (defined as the distance between water level and the point on the extended time-drawdown curve) for I-2 as a function of time was plotted on log-log graph paper. The Boulton-type curves for water-table aquifers were used for analysis using a method similar to that of Prickett (1965). From the family of curves, the curve with an r/D (ratio of the radial distance to the center of a well, r , to a drainage factor, D) value of 0.8 was chosen for early and late matches (see Boulton, 1963, for an explanation of

these terms). The estimated transmissivity and specific yield were 237,000 gpd/ft and 0.035, respectively. This gives an average hydraulic conductivity of approximately 2150 gpd/ft² for the entire thickness of the aquifer. A semilog (or straight line) method to compute aquifer parameters under water-table conditions has been presented by Neuman (1975). Using this method, the early and late data should appear as similar-sloped straight-line plots on semilog paper separated by a flat portion. The slopes of the late and early data can be used to compute the transmissivity. A transmissivity of 300,000 gpd/ft is obtained using the slope of the latter parts of the recovery data. The average hydraulic conductivity for the entire thickness of the aquifer estimated from this second method is 2730 gpd/ft². However, only 115 minutes of recovery data were available for analysis. Because of the uncertainties in analyses, we preferred to use a transmissivity value between 300,000 and 350,000 gpd/ft for this area, which falls in the mid-range of the reported values; the resulting hydraulic conductivity also matches well with that reported by Naymik and Sievers (1983) for the Sand Ridge site (3000 gpd/ft²). An overall specific yield of 0.1 was used for modeling purposes.

In standard time-drawdown or time-calculated recovery analysis, the pumped and observation wells are assumed to fully screen the aquifer depth. In our study, the pumped well, although fully penetrating the aquifer, was screened over only the final 35 ft of the aquifer. Similarly, the observation well was only 22 ft deep, and the screened interval was only the final 2 ft. Furthermore, the observation well (I-2) was approximately 110 ft away from the pumped well. The steady-state corrections to the drawdown, extrapolating the tabular values presented by Walton (1962), were approximately 3 percent. Corrections for transient conditions can be found in Hantush (1961) and Neuman (1975). However, these methods require the development of type curves for each well. Because the aquifer is not homogeneous (rather the aquifer material is graded from the top to the bottom), the additional effort in analysis is not justified.

The entire aquifer was conceptualized to be composed of layers of sand with hydraulic conductivities increasing with depth. A detailed graphical presentation of this is in the section Ground-Water Flow Modeling.

Dispersivities. The longitudinal and transverse dispersivities were estimated from results of the spring 1994 natural-gradient tracer test. A technique presented by Zou and Parr (1993) was used to determine dispersivities for a two-dimensional plume. Breakthrough data from I-3 and port 9 of MLS-13 were used for analysis. The elevations of port 9 of MLS-13 and I-3 are approximately the same as the injection zone (I-1). The centerline of the tracer plume passed through the two observation points. A trial-and-error procedure, as outlined by Zou and Parr (1993), was performed to estimate the direction of the centerline of the plume and ground-water velocity.

The computed pore water velocity was 0.75 ft/day between the injection well and the pair of observation points; and the longitudinal and the transverse dispersivities were estimated to be 0.285 and 0.041 ft, respectively. An analytical

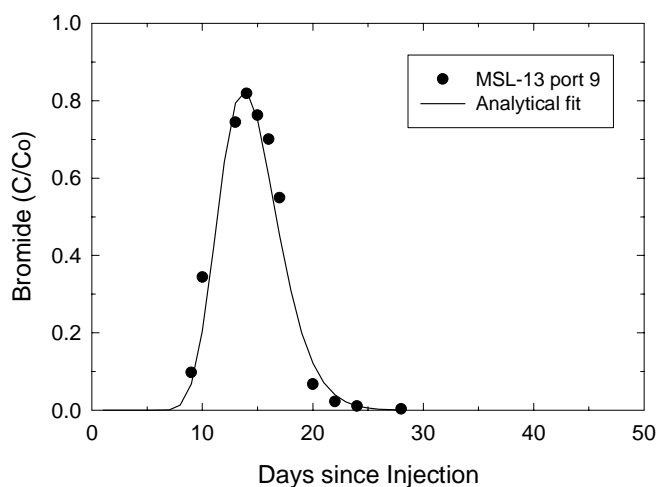


Figure 19. Bromide concentrations in MLS-13 port 9 as a function of days since injection for spring 1994 natural-gradient tracer test.

solution for the two-dimensional advection-dispersion equation (Zou and Parr, 1993) was used to fit the bromide breakthrough data (figure 19) for sample port 9 of MLS-13. The adjusted longitudinal and transverse dispersivities for fitting were 0.18 and 0.098 ft, respectively. The uniform rate of ground-water flow was slightly adjusted (0.69 ft/day) to match the breakthrough time.

Tracer Tests

Three tracer tests were performed during this study: two under natural-gradient conditions and one with the irrigation pump on.

Natural-Gradient Tracer Tests. It was obvious from the initial natural-gradient experiment that both lateral and vertical resolution at the test site were inadequate. Thus six additional MLSs (MLS-12 to MLS-17) with increased depth resolution (ports every foot) were installed soon after the preliminary test (figure 3).

The second natural-gradient test was started on April 19, 1994. About 200 gallons of ground water spiked with NaBr and NaNO₃ to final concentrations of approximately 320 mg/L Br⁻ and 100 mg/L NO₃⁻-N was injected by gravity into well I-1 and allowed to migrate under the natural hydraulic gradient. Using the same assumptions discussed previously, the radius of a cylindrical plume with a depth of 3 ft would be about 2.9 ft. The assumption of a depth of 3 ft is based on the observation that the largest maximum concentrations of tracer ($C/C_0 > 0.75$, where C is the tracer concentration in a well or MLS port and C_0 is the tracer concentration of the injection solution) were found in MLS ports over this depth range. Maximum concentrations were much less ($C/C_0 < 0.5$) for ports above and below this range. No wells were within 2.9 ft of the injection well. Tracer was never detected in the closest well, I-2, which is 3.4 ft from I-1.

Breakthrough of the tracers was first observed in well I-3, about 4.5 ft from I-1, starting one day after injection and peaking six days afterward. Tracers were first detected in MLS-13, approximately 9 ft from the injection well, nine days after injection. Nitrate was not observed to be retarded relative to bromide and appears to behave conservatively in this aquifer over the time frame of the experiments. Tracer was detected in every port of MLS-13 and MLS-14, with the greatest concentrations in ports sampling the zone in which tracer was injected (figure 20). The fact that tracer was detected in all ports, even the lowermost ports approximately 7 ft below the bottom of the injection well, indicates that there was significant vertical movement during injection. The breakthrough in these lower ports was delayed relative to the injection zone ports, due to longer flowpaths.

Maximum tracer concentrations were observed in MLS-13 between 14 and 17 days after injection, depending on the port, and reached background levels approximately 28 days after injection. Maximum concentrations in MLS-14 and MLS-5 were observed between 17 and 22 days and between 31 and 36 days, respectively, after injection, depending on the port. The ports in the body of the plume (i.e., with the highest tracer concentrations) tended to peak at the earlier times. Ground-water velocities calculated from these data were between 0.57 and 0.69 ft/day for MLS-13, 0.69 and 0.73 ft/day for MLS-14, and 0.69 ft/day for MLS-5. Thus the tracers are behaving conservatively as there is no decrease in solute transport rates with time or distance.

These results, especially from MLS-13, support the conclusions reached by Naymik and Sievers (1983) at the Sand Ridge site just north of this field site that, despite the relatively homogeneous sand aquifer, local physical heterogeneities strongly influence solute transport. The main part of the plume migrated over a depth range of about 3 ft; it was detected in ports that straddle the injection well's screened interval. The peak concentrations were lower in MLS-14 and MLS-5 than MLS-13, probably due to the effects of dispersion

increasing with increasing distance. The fact that tracer was found in all ports of MLS-13 and MLS-14 indicates that vertical dispersion under natural (i.e., nonpumping) conditions in this aquifer is significant.

Forced-Gradient Tracer Test. The tracer test under forced-gradient conditions was started on September 12, 1994. Approximately 200 gallons of water with initial concentrations of 288 mg/L Br⁻ and 77 mg/L NO₃⁻-N were injected into well I-3. Two hours after injection was complete, the irrigation pump was turned on. The pumping rate was determined by manometer to be about 1200 gpm. The pumped water was drained via irrigation pipes into several drainage basins and ditches north of the test site at least 500 ft away. Samples were taken from the injection well and MLSs every four to seven hours over the next three days, at the end of which the pump was shut off. The tracer plume was followed for the next month under natural-gradient conditions until it migrated off the experimental plot. As in the natural-gradient test, nitrate was not retarded relative to bromide.

The effect of pumping on tracer movement can be seen readily when comparing concentration data from the injection wells for the forced- and natural-gradient tests; the tracer is removed much more rapidly during the pumped test (figure 21). During the three-day pumping period, tracer was detected in MLS-12 and MLS-13 but not in MLS-2. Both MLS-12 and MLS-2 are aligned between I-3 and the irrigation well, 5.7 and 9.9 ft, respectively, to the west of I-3. The MLS-13 is slightly north of MLS-12 and 6.2 ft from I-3 (figure 3). Tracer was detected in MLS-12 and MLS-13 during the three days of pumping but not in MLS-2, thus the tracer cloud was transported horizontally less than 10 ft by the pumping (figure 22). As in the natural-gradient test, tracer was detected in all ports of MLS-12 and MLS-13.

By the end of the three days, the water level had dropped approximately 3 ft at the injection well (I-3), 3.7 ft at well W-2, and 15.5 ft at the pumping well. The formation of a cone of depression caused tracer to be detected in significantly greater

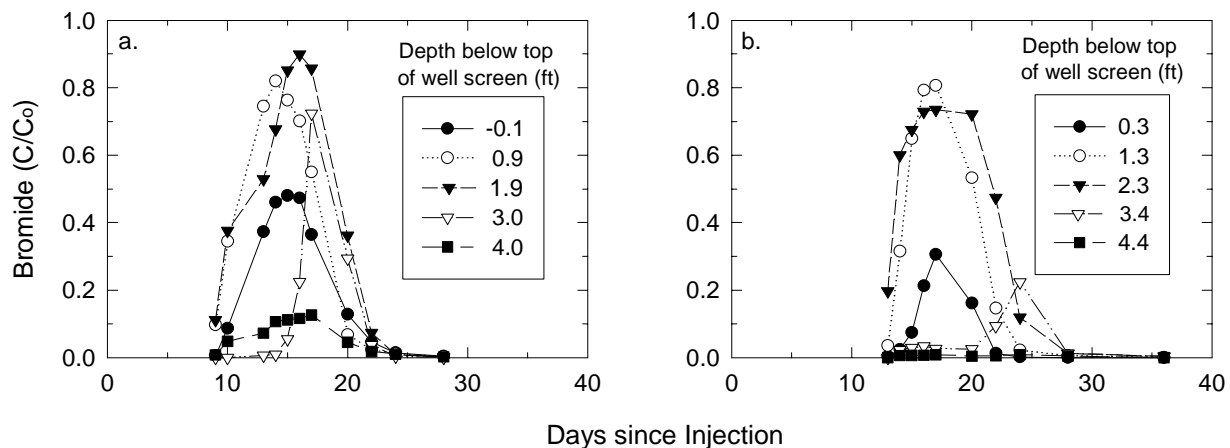


Figure 20. Breakthrough for normalized bromide data for five sampling ports of: a. MLS-13 and b. MLS-14 for natural-gradient tracer test, spring 1994.

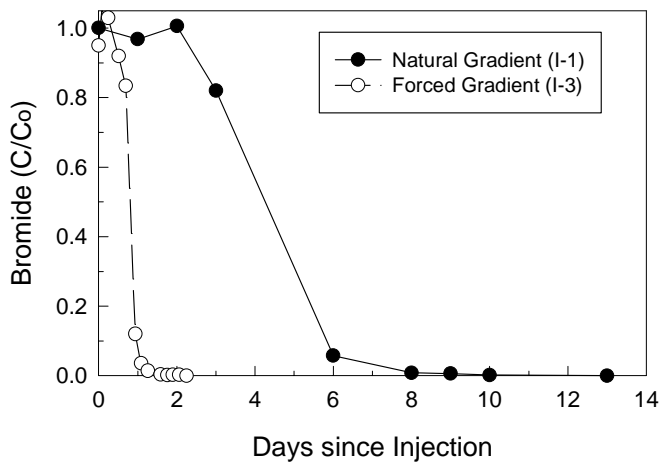


Figure 21. Normalized bromide data from the injection wells for the spring 1994 natural-gradient tracer test and forced-gradient tracer test.

concentrations in deeper ports during this test than in the natural-gradient tests (figure 23).

After the pump was turned off, the plume rose as water levels recovered. This is especially apparent in the shallower ports of MLS-12 and MLS-13, which had little or no detectable tracer during pumping, but had measurable levels after the pump was turned off (figure 22).

However, the plume did not return to its prepumping depth, as shown by comparing postpumping data with results from the natural-gradient test. Breakthrough was observed in MLS-17 two weeks after the pump was turned off (figure 24). Bromide concentrations were greater in deeper ports than for the strictly natural-gradient test (figure 25). Although these results are for different MLSs due to the pumping pulling the plume slightly to the west, the absolute depth of the plume is

deeper by between 1 and 2 ft after the pumping test. Because the plume did not rebound to prepumping levels after the pump was turned off, there must be a source of water recharging at the water table. The most likely source is drainage from the unsaturated zone.

Thus pumping did have a measurable effect on solute transport at a distance of about 100 ft from the well. It appears that pumping can have a significant effect on vertical movement of solutes in this aquifer, especially close to the well. Drawdown was observed at the most distant observation wells, about 0.66 ft at O-1 (782 ft from the irrigation well) and 0.48 ft at O-3 (1065 ft away). Thus if a drawdown of approximately 3 ft at I-2 caused about 2 ft of plume drawdown, we might expect some plume displacement throughout the zone of influence. It is expected that the duration of individual pumping episodes (i.e., how long the pump is run continuously) is very important in affecting solute movement, more so than the absolute amount of pumping done in a year.

Ground-Water Flow Modeling

A detailed ground-water modeling effort was undertaken for the vicinity of the study area to estimate the impact of pumping on water levels and the migration of conservative solutes toward the irrigation well. The area to be modeled was selected so that the effect of pumping on model boundaries would be insignificant. An area 12,600 ft (2.38 miles) long and 12,400 ft (2.35 miles) wide was selected for simulation (figure 26). Ground-water heads were simulated using the USGS model MODFLOW (McDonald and Harbaugh, 1988).

In order to accommodate the partially penetrating pumped and injection wells and the gradually varied hydraulic conductivity with depth, a model with five horizontal layers was prepared. The hydraulic conductivities were selected so the overall transmissivity of the aquifer satisfied the following relationship:

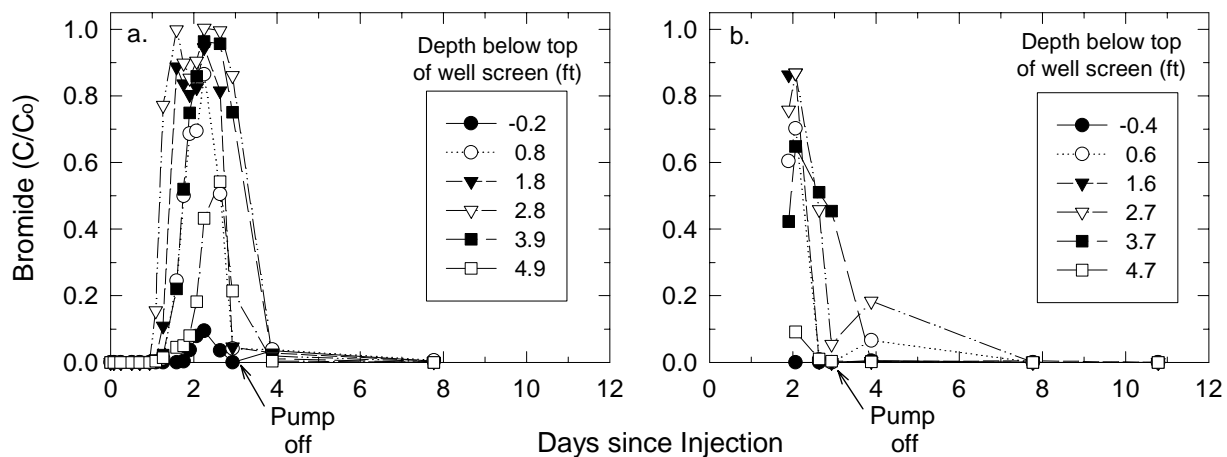


Figure 22. Breakthrough for normalized bromide data for six sampling ports of a. MLS-12 and b. MLS-13 for forced-gradient tracer test.

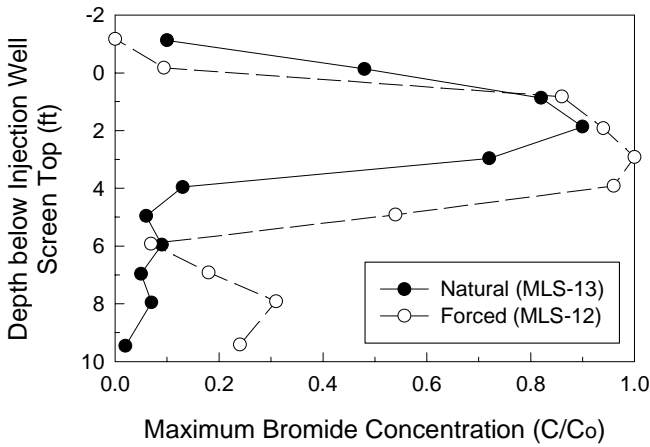


Figure 23. Maximum normalized bromide concentrations measured in all ports of MLS-13 (spring 1994 test) and MLS-12 (during test).

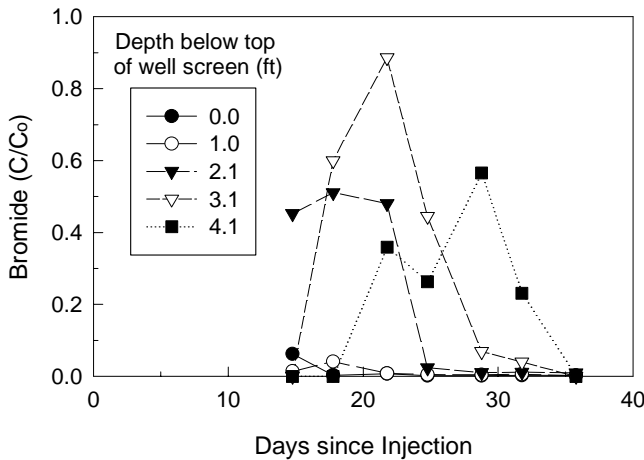


Figure 24. Breakthrough for normalized bromide data for five sampling ports of MLS-17 for forced-gradient tracer test after the pump was turned off.

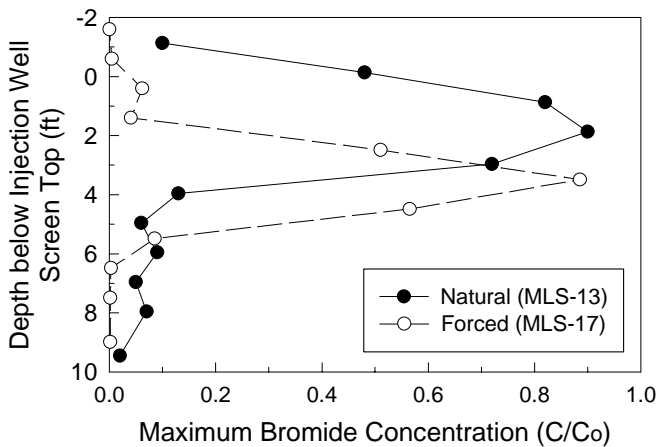


Figure 25. Maximum normalized bromide concentrations measured in all ports of MLS-13 (spring 1994 test) and MLS-17 (after test).

$$\bar{T} = \sum K_i b_i \quad (4)$$

where \bar{T} is the overall transmissivity (gpd/ft), K_i is the hydraulic conductivity (gpd/ft²) of an individual layer i , and b_i is the thickness of the layer. A cross section of the aquifer with selected hydraulic conductivities and layer elevations is shown in figure 27. The porosity and specific yield for each layer were selected to be 0.28 and 0.1, respectively. The study area was divided into 1600 cells (figure 26) with cell lengths or widths ranging from 100 to 600 ft. An extended memory version of the model, compiled on a Lahey compiler (F77-EM/32, version 6.0, Lahey Computer Systems, Incline Village, NV), was used for simulation.

The regional water levels for this area, measured in the fall of 1993 (Sanderson and Buck, 1995), were superimposed on the study area (figure 28). The hydraulic heads along the boundary of the model, obtained from the Sanderson and Buck (1995) water-table map, were used as constant head cells. The Mason-Tazewell drainage ditch in the northern part of the model area was treated as a river that could gain or contribute water depending upon the heads in the aquifer. The area to the north of the ditch was considered to be a no-flow boundary. The boundaries of the modeled area are shown in figure 26. Areal recharge was set to 0.00234 ft³/day, which was determined by Walker et al. (1965) based on an annual recharge of 10.3 inches calculated from selected flow channels for the area. Clark (1994) addressed the variability of recharge and calculated recharge (using flow tubes) ranging from 20 to 30 inches during the fall of 1992 in areas to the southeast of the tree nursery. However, during 1992, precipitation was higher than the mean annual precipitation for the area.

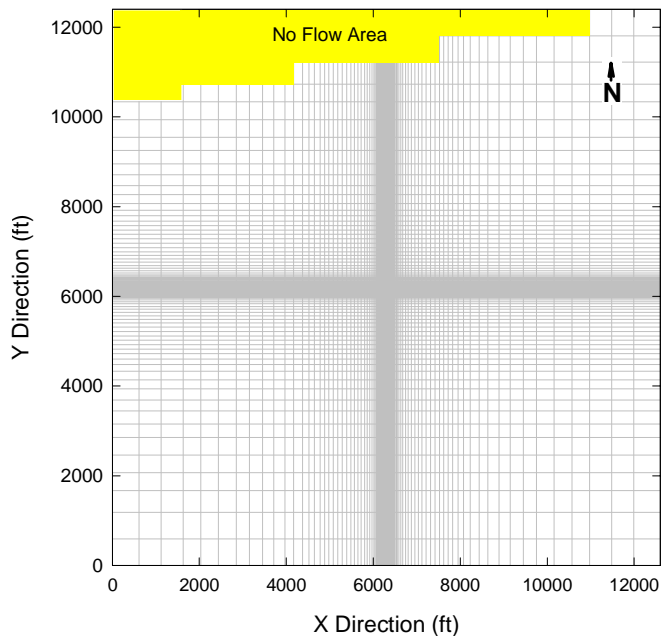


Figure 26. Model grid used for MODFLOW simulation of static potentiometric surface.

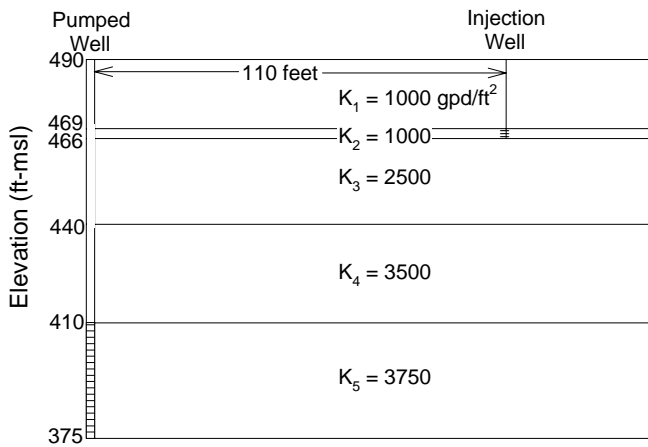


Figure 27. Cross section of aquifer showing model layers and hydraulic conductivity values used in MODFLOW simulation. Hashed areas represent well screens.

A steady-state simulation, with no irrigation pumping, was conducted to compare the water levels with those measured in the fall of 1993 when no irrigation occurred. The water levels predicted by the model matched measured values reasonably well (figure 29). Therefore, no alterations to the assigned hydraulic properties to the five layers were made for calibration purposes. A steady-state simulation was carried out using a constant-rate pumpage of 1200 gpm. The extent of the capture zone after five years of pumpage is shown by flow lines emanating out of the well, and the impact of drawdown on the boundaries was insignificant (figure 30).

A separate five-layer MODFLOW model was created for a smaller area with the irrigation pump at its center to show the impact of irrigation pumpage on tracer movement. The inner model was 1200 ft by 1200 ft with the same five layers. Model cell sizes ranged from 10 ft by 10 ft at the center (near the pumping well) to 100 ft by 100 ft at the far ends (figure 31). Recharge to the surface layer was the same as that in the outer (larger) model. The boundary was specified by constant-flux cells (wells). The fluxes entering or leaving the boundary cells were computed from the outer model after conducting a cell-by-cell flux balance computation. The system was modeled for the given rate of pumpage, boundary fluxes, recharge, and the given initial water level at observation well I-2. The water-level map for steady-state pumpage is shown in figure 32.

Ground-Water Transport Modeling

As reported earlier, tracer traveled horizontally less than 10 ft during the forced-gradient tracer test conducted in the early fall of 1994. In order to verify if the three days of pumping were sufficient to carry the tracer mass from the injection well to the pumped well, a distance of approximately 110 ft, two transport models were tested.

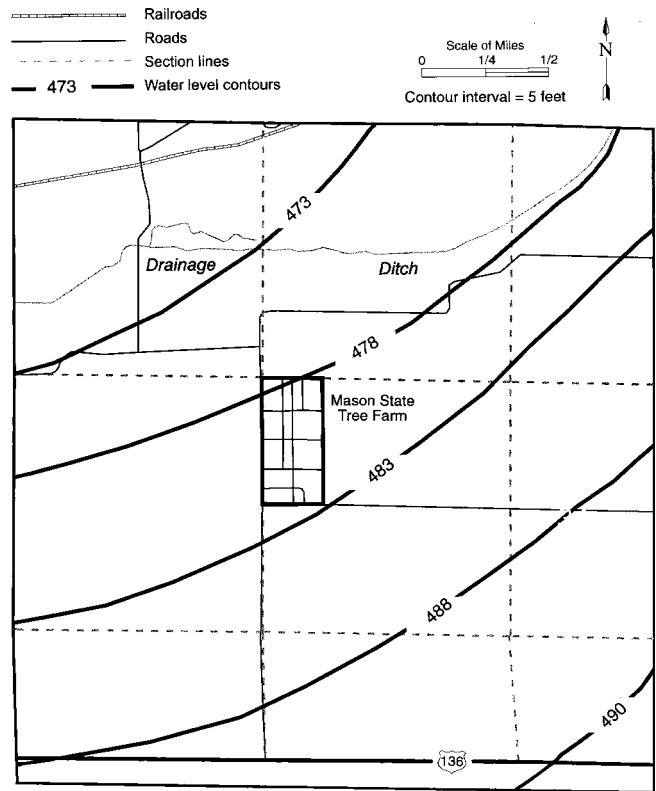


Figure 28. Area used in MODFLOW simulation showing model boundaries.

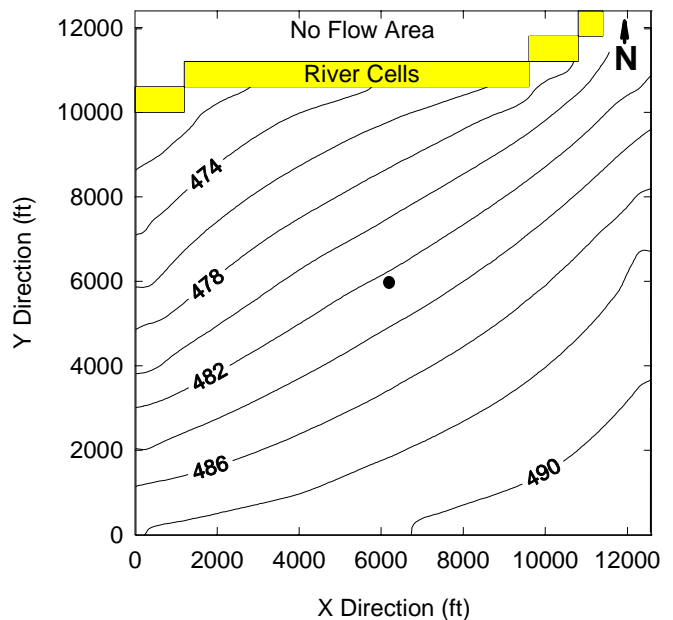


Figure 29. Potentiometric surface of field site under static conditions calculated using MODFLOW. Circle in center is location of irrigation well.

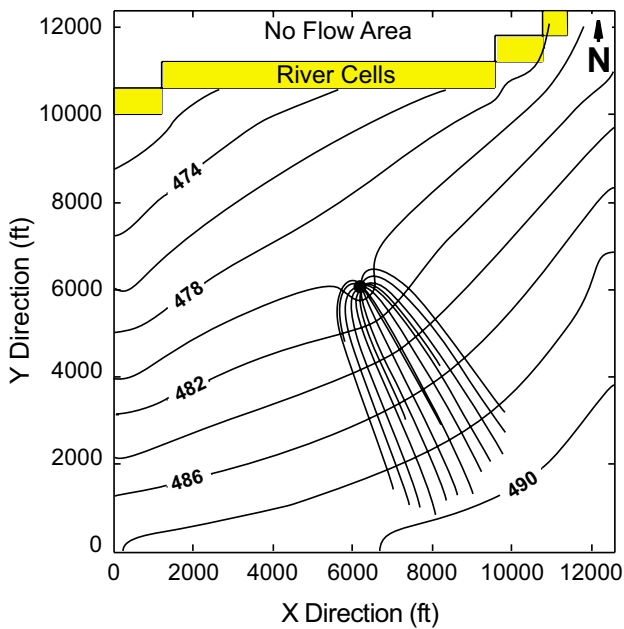


Figure 30. Potentiometric surface of field site under pumping conditions calculated using MODFLOW. Five-year particle pathlines are shown from the irrigation well.

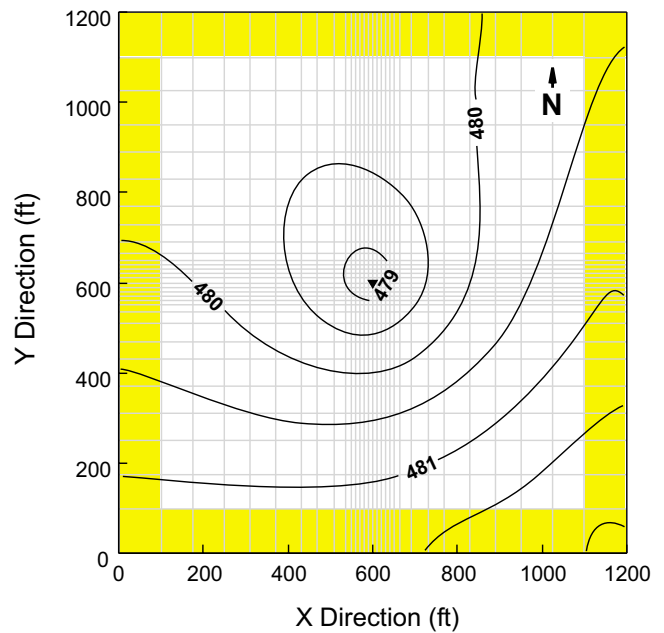


Figure 32. Potentiometric surface of field site under pumping conditions calculated using MODFLOW.

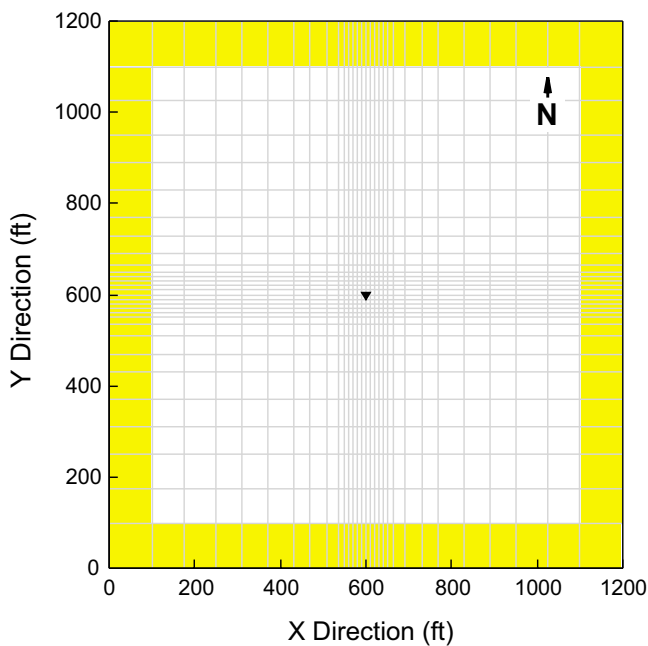


Figure 31. Inner model grid used for MODFLOW simulation of pumping conditions. Shaded cells are constant flux cells. Black triangle is location of irrigation well.

Random-Walk Model

A transport code (RAND3D) using the random-walk particle tracking technique was considered for simulation (Engineering Technology Associates, 1994a). The model is based on the original random-walk particle tracking model of Prickett et al. (1981). The RAND3D code uses a preprocessor, PREMOD3D (Engineering Technology Associates, 1994b), that uses the head data from MODFLOW to compute velocity. The RAND3D code is written in Microsoft QuickBasic language for personal computers, and for optimum performance the nodal dimension is limited to 10,000. A second limitation of the model is that it uses regularly spaced grids in the X and Y directions. The authors of RAND3D suggest that a larger model can be run in the system at the expense of program features and the number of particles. The inner model was discretized into 49 nodes (48 cells) in both the X and the Y directions, with model cells of 25 ft. Because RAND3D uses the head data from MODFLOW to compute velocity, a new MODFLOW model for the inner (1200 ft by 1200 ft) area was created. The model had five layers and 48 cells in the X and Y directions. In random-walk technique, an accounting box, equivalent to the area of one cell, is used to compute nodal concentrations based on the particle count and the particle mass. If the source area is much smaller than a single cell, dilution occurs while computing initial input nodal concentrations. However, the particle positions are correctly maintained for further advection and dispersion.

The natural-gradient tracer test was conducted in a small area. Because of the coarseness of the grid used in the RAND3D model to simulate the natural-gradient experiment,

a simple two-dimensional, random-walk particle tracking model (Prickett, 1994) was used for simulation. This model is a simpler version of the full three-dimensional code (Engineering Technology Associates, 1994b). A steady-state velocity distribution was used for simulation. For two-dimensional simulations, only the second layer (3 ft deep) was used. Uniform grids were used, and the grid spacing in the X and Y directions were 1.0 ft each. Solute mass was released from a square block 5.64 ft wide to account for the actual mass of the solute released. The plume was allowed to advect and disperse in the uniform regional flow. No sources or sinks were present. A simulated solute plume one, five, and ten days after injection is shown in figure 33.

The model results agree fairly well with the actual data. Bromide concentrations in I-1 were approximately 300, 18, and <1 mg/L at one, six, and ten days after injection (figure 21). Concentrations were approximately <1, 235, and 55 mg/L one, six, and ten days after injection in I-3. The model predicts a higher concentration after ten days. Bromide was first detected in MLS-13 nine days after injection at a maximum concentration of 36 mg/L (port 8) (figure 20). The maximum concentration after ten days was 120 mg/L.

A separate simulation was conducted using the three-dimensional RAND3D to study the impact of pumping on tracer migration. The velocity profile computed from the MODFLOW model was used as input to the RAND3D code. The edges of the velocity field computed by the model were truncated due to memory limitations. A constant dispersivity was assumed, and computed longitudinal and transverse dispersivities were input directly. Because bromide is a conservative tracer, the retardation coefficient was set to unity. The tracer was released from the injection well in a short

duration. The volume of the tracer was 200 gallons, with a bromide concentration of 320 mg/L. It was initially assumed that the tracer mass formed a cylindrical slug upon injection. However, the current version of RAND3D only allows the placement of particles on the outer walls of a cylinder. Because this would have distorted the initial concentration data, a fully mixed square prism was assumed to be the solute source, with sides 5.6 ft and a depth of 3.0 ft. A total of 2000 particles, each weighing 0.000267 lb (0.12 g) were released from this source.

The particles were allowed to advect and disperse in the flow for three days. The center of the plume moved approximately 10 ft in three days, considerably greater than was observed in actuality (figure 34). There was considerable spreading of the plume compared to that under natural-gradient conditions. From its shape and location, the plume was observed to have been pulled toward the well.

MODFLOW + MODPATH

The second method used for simulating particle movement was MODFLOW (McDonald and Harbaugh, 1988) and MODPATH/MODPATH-PLOT (Pollock, 1994). MODPATH uses the computed velocities from MODFLOW to advect the particles in the direction of the velocity vector. Dispersion effects are not simulated. A single particle was released from a cell in the second layer whose center was 110 ft from the pumping well. The positions of the particle (before it was captured by the well) in plan and cross-sectional views are shown in figure 35. The particles moved approximately 10 ft from the center of the cell toward the pumping well in three days. A slight sag in the particle trajectory in the horizontal

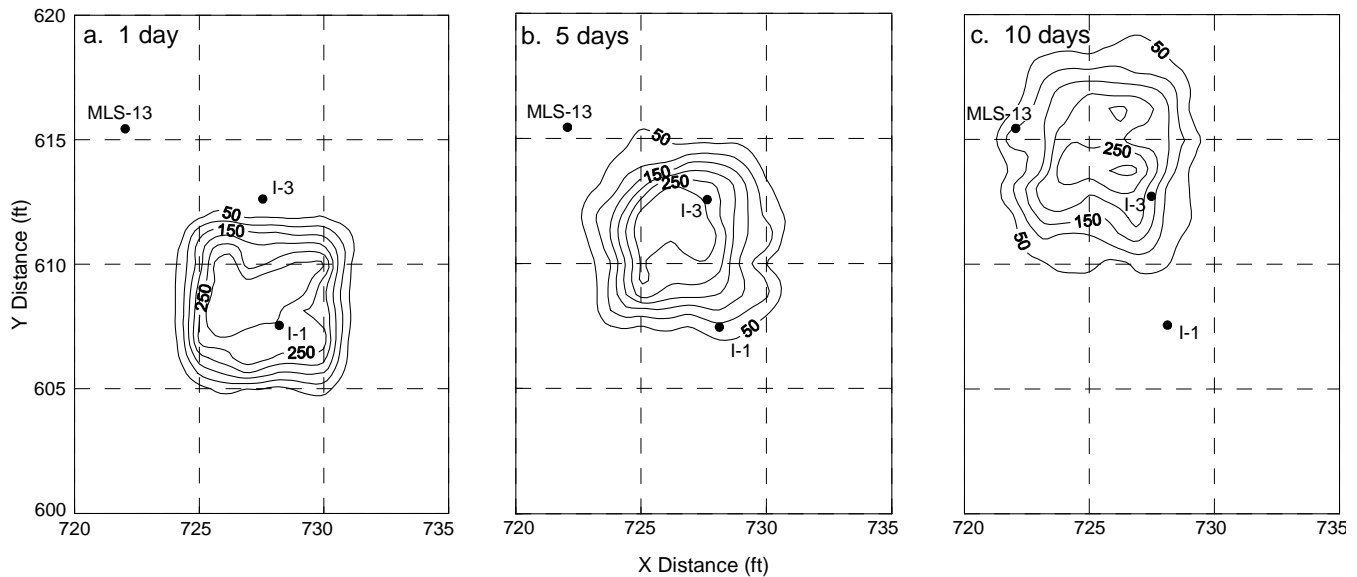


Figure 33. Plan view of bromide plume calculated by RAND3D for fall 1994 natural-gradient tracer test after: a. one, b. five, and c. ten days. Irrigation well is approximately 110 ft west of I-1. Contour interval is 50 mg/L.

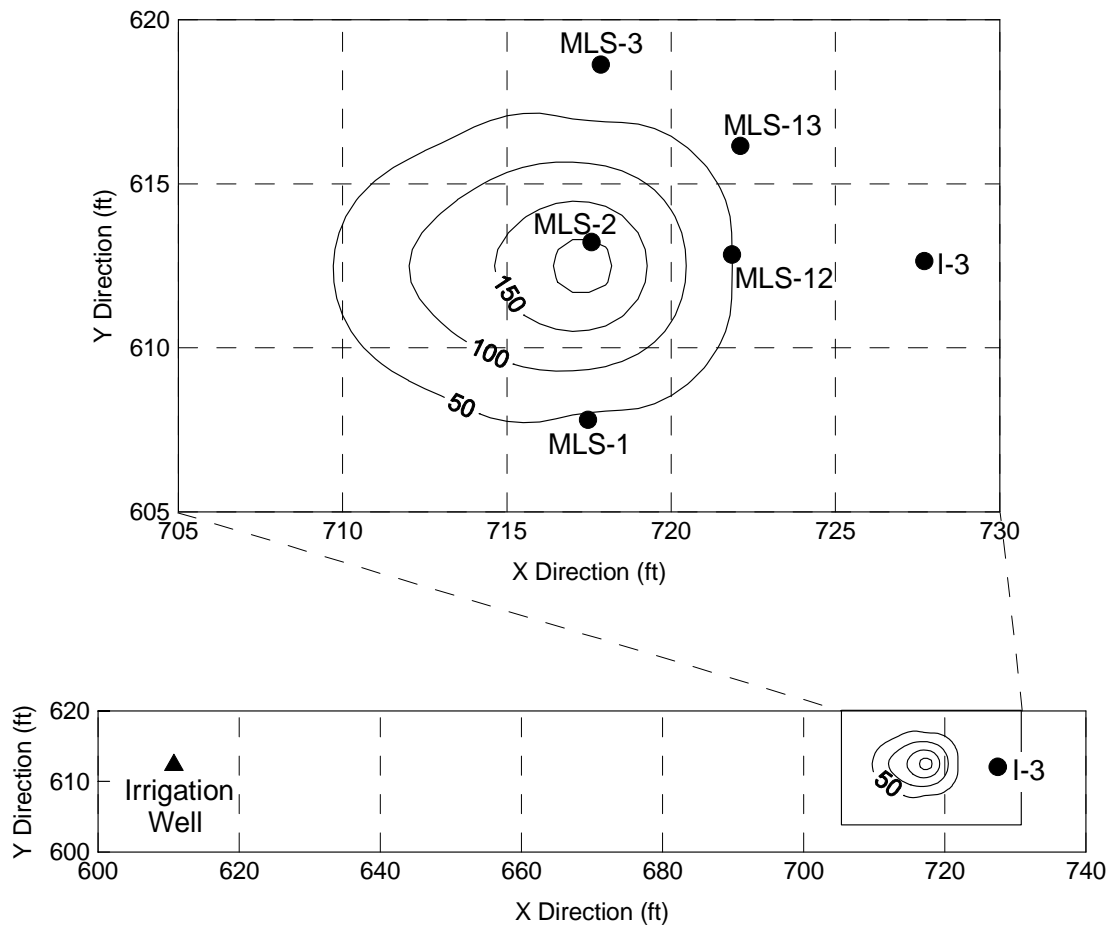


Figure 34. Bromide plume calculated by RAND3D for forced-gradient tracer test. Contour interval is 50 mg/L.

plane is due to the impact of the regional velocity gradient that is toward the northwest direction in the study area.

Effect of Pumping on Tracer Mass Recycling

Irrigation pumpage has been increasing in the Havana Lowlands in recent years (1993 excluded). Typical pumping rates for new irrigation wells range from 1000-1200 gpm. Because typical irrigation application lasts for a few days (equivalent to 1 to 2 inches of water application), an extreme scenario with three days of pumping was taken into consideration. It was assumed that the center-pivot irrigation system was built over a full section (1.0 square mile) and water was applied to an actual area of slightly over 500 acres. This amounts to approximately 2.8 inches of application over three days, or about 0.95 inches per day. The purpose of the simulation was to compute the quantity of water (hence nitrate) entering the pump after one, two, or three days of pumping. The concentration of nitrate as a function of depth is shown in figure 13. The amount of water from

individual layers with a depth-averaged concentration of nitrate for that layer was used to provide an estimate of nitrate in the pumped water.

A reverse-particle tracking analysis was conducted using MODPATH to examine the potential origin of the nitrate entering the pumped well. Thirty-five particles placed at the center of the well were allowed to advect backward with time. The particle positions after one, two, and three days are shown in figure 36. A majority of the volume of the pumped water is derived from the bottom two layers.

The volume fraction of water derived from each of the five layers was calculated from the particle positions after one, two, and three days (table 6). Based on these estimates, the concentration of nitrate in the pumped water for the first three days will be 0.08, 0.22, and 0.56 mg/L using the nitrate concentrations of MLS-7. Thus for a single irrigation application, the recycling effect on the surficial nitrate plume is minimal. However, more effort is needed to examine the impact of cycling pumpage on the movement of the nitrate plume.

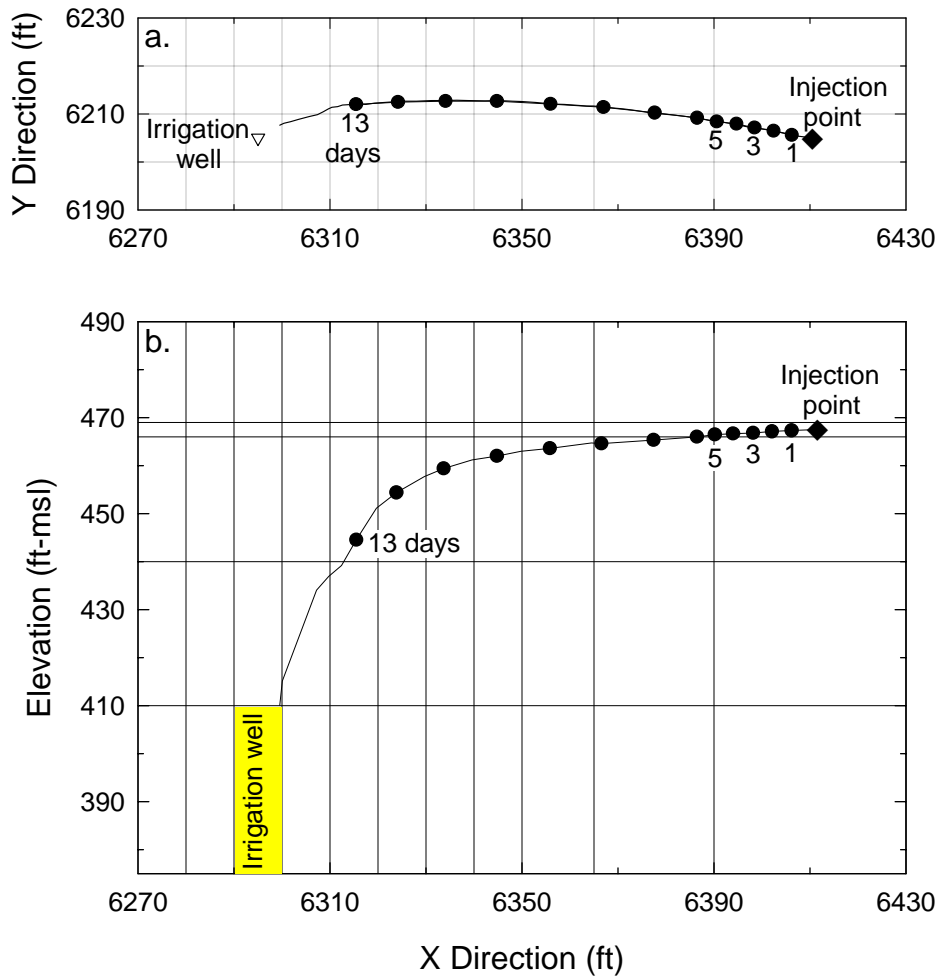


Figure 35. a. Plan and b. cross-sectional views of particle tracks calculated by MODPATH for forced-gradient tracer test. Each point represents one day. No vertical exaggeration in cross-sectional view (b).

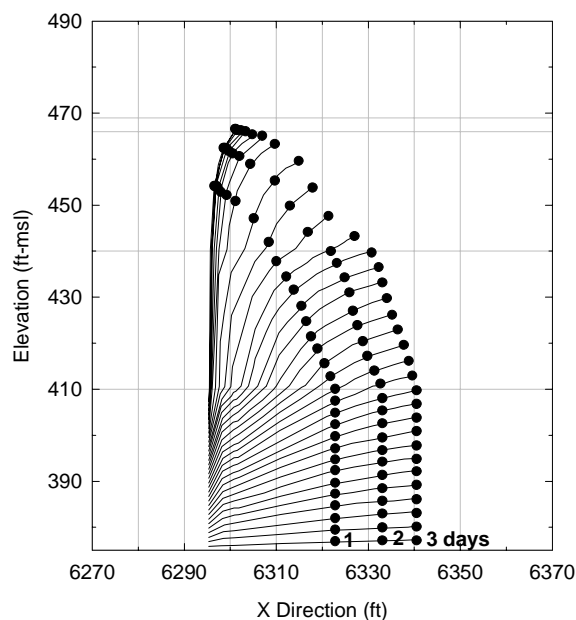


Figure 36. Three-day particle tracks calculated by MODPATH for reverse tracking simulation. Each point represents one day. No vertical exaggeration.

Table 6. Percent of Water from Each Layer after One, Two, and Three Days of Pumping*

Layer	Days		
	1	2	3
1	0.0	0.0	0.0
2	0.0	0.0	0.04
3	1.6	4.3	11.15
4	30.6	34.7	35.65
5	67.8	61.0	53.16

Note: *Calculated by MODPATH for reverse particle tracking simulation.

CONCLUSIONS

Water samples taken from the MLSs installed in a sandy aquifer reveal considerable heterogeneity in the aqueous chemistry both spatially and temporally. Recharge is rapid in this system, and the water chemistry of the recharge water is variable both spatially and temporally; it is especially influenced by agricultural practices.

Nitrate concentrations are elevated in a zone between approximately 15-30 ft beneath the surface, although in certain areas and at certain times this zone was not found. The maximum nitrate concentrations in this zone were slightly greater than 20 mg/L as nitrogen, well above the MCL. Nitrate was generally absent below 30 ft in the aquifer, probably due to denitrification reactions; the tritium data suggest that vertical movement of solutes in the aquifer is rapid, and that there has been enough time to transport surface-applied fertilizer to depths in excess of 100 ft. Reducing conditions are found at depth, indicating conditions are conducive to denitrification. Drinking water quality is not degraded by fertilization in this area because drinking-water wells are generally screened well below the zone of elevated nitrate concentrations. Homeowners with a well screened at a depth less than 50 ft should have the water periodically analyzed for nitrate.

One of the major objectives of this study was to determine the effect of irrigation pumping in sandy aquifers on solute transport. From the results of these tracer tests, the effects apparently are measurable but not substantial. The movement of tracer both vertically and horizontally was slight under

pumping conditions, on the order of several feet over the three days of pumping. It is clear that, at least for this well, solutes near the water table are not transported to the well screen during pumping, with the possible exception of solute very close to the well, perhaps within 20 ft. This conclusion was confirmed by solute transport modeling. Thus, the vast majority of nitrate applied in this area is not being recycled through the irrigation wells. For more shallow irrigation wells, more nitrate might be recycled, but it still seems likely that the amount would be a small percentage of the nitrate reaching the water table. Bowman and Kimpel (1991) reported that fewer than 9 percent of the irrigation wells in the Havana Lowlands (Mason County and southeast Tazewell County) are less than 70 ft deep, and fewer than 2 percent are less than 60 ft deep.

However, vertical movement of tracer due to pumping was observed, and the plume did not rebound after the pump was turned off. The vertical movement was between 1 and 2 ft about 100 ft from the irrigation well after three days of pumping. In Mason County, irrigation pumps operate about 30 days during an average year, although usually only for three days continuously. Thus, greater vertical movement of nitrate and pesticides might be expected during the irrigation season. The movement of solutes will be affected by the timing and duration of pumping cycles. The fact that nitrate was not found at great depths suggests some removal mechanisms, such as denitrification, must be occurring.

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Appendix A.
Driller's Log for Irrigation Well

White Ink Copies:
 Ill. t. of Public Health
 Yellow Copy: Well Contractor
 Golden Copy: Well Owner

Well Construction Report

THIS FORM MUST BE COMPLETED WITHIN 30 DAYS
 OF WELL COMPLETION AND SENT TO
 THE ILLINOIS DEPARTMENT OF PUBLIC HEALTH
 DIVISION OF ENVIRONMENTAL HEALTH
 525 WEST JEFFERSON STREET
 SPRINGFIELD, ILLINOIS 62761

GEOLOGICAL AND WATER SURVEYS WELL RECORD

9. Driller Grosch Irrigation Co. License No. 102-003340
 10. Well Site Address Topeka, IL
 11. Property Owner Mason State Tree Nursery #9
 12. Permit No. 018074 Date Issued 5/22/90
 13. Location: County MASON

Sec. <u>337A</u>					
Twp. <u>22N</u>					
Rge. <u>7W</u>					

1. Type of Well
- a. Bored Hole Diam. 30 in. Depth 115 ft
 Buried Slab: Yes No
 Drive Pipe Diam. in. Depth ft
- b. Driven Finished in Drift In Rock
 (KIND) FROM (Ft.) TO (Ft.)
- | | | |
|----------|---|----|
| Concrete | 0 | 20 |
|----------|---|----|
- d. Grout:

14. Water from at depth ft to ft
 15. Casing and Liner Pipe Kind and Weight From (ft) To (ft)

14	Plain Steel	0	80
14	SS Screen	80	115

NE QUADRANT

2. Well furnishes water for human consumption? Yes No
3. Date well drilled 10/24/90
4. Permanent pump installed? Yes No
 Manufacturer Date Type
- Location
- Capacity gpm. Depth of setting ft.
5. Well top sealed? Yes No Type
6. Pitless adapter installed? Yes No
 Manufacturer Model No.
- How attached to casing?
7. Well disinfected? Yes No
8. Pump and equipment disinfected? Yes No

IMPORTANT NOTICE

This State Agency is requesting disclosure of information that is necessary to accomplish the statutory purpose as outlined under Public Act 85-0863. Disclosure of this information is mandatory. This form has been approved by the Forms Management Center.

16. Screen: Diam. 14 in. Length 35 ft. Slot Size 0.55
 17. Size hole below casing in. 18. Ground Elev. 445 ft msl.
 19. Static level 14 ft below casing top which is 50 ft. above ground level. Pumping level 24 ft. pumping gpm for hours.

20. Earth Materials Passed Through

	Depth of Top	Depth of Bottom
TOP SOIL	0	4
Fine to med. sand, brown	4	25
Medium sand to med. gravel, brown	25	40
Med. sand to fine gravel, blue	40	80
Med. sand to med. gravel, blue	80	115
Fine to coarse gravel & rocks	115	140

Continue on separate sheet if necessary.

Signed *Paul Grosch* Date 11/12/90

PRESS FIRMLY WITH BLACK PEN OR TYPE
 Do Not Use Felt Pen

Appendix B.
Water Chemistry Data

Table B1. Sample Preservation and Analytical Techniques for Water Chemistry Parameters Analyzed

<i>Parameter</i>	<i>Sample preservation*</i>	<i>Analytical technique</i>
pH, Eh, DO, temperature, conductivity	Field analysis	Electrode
Cl ⁻ , SO ₄ ²⁻ , F ⁻ , NO ₂ ⁻ , o-PO ₄ ³⁻	Filtered**	Ion chromatography
Ca, Mg, Na, K, Ag, Al, As, B, Ba, Be, Cd, Co, Cr, Cu, Fe, Hg, Li, Mn, Mo, Ni, P, Pb, S, Sb, Se, Si, Sn, Sr, Ti, V, Zn	Filtered** and acidified with 0.5% HNO ₃ to pH < 2.0	Inductively coupled plasma
NH ₄ ⁺	Filtered** and acidified with 0.05% H ₂ SO ₄ to pH < 2.0	Colorimetric
Alkalinity	None	Titrametric
TDS	Filtered**	Dried, filtered** residue
NO ₃ ⁻ -N	1. Filtered** and acidified with 0.05% H ₂ SO ₄ to pH < 2.0 2. None	1. Ion chromatography 2. Electrode
Br ⁻	None	Electrode
DOC	None	Ultraviolet promoted oxidation
DIC	N/A	Calculated from alkalinity and pH measurements

Notes: *All samples stored on ice for transport back to ISWS laboratories and refrigerated prior to analysis.
 ** Samples passed through 0.45 μm nylon or polysulfone filters.
 N/A = not applicable.

Table B2. Complete Chemical Analyses, July 19, 1993

<i>MLS</i>	$^{\circ}\text{C}$	<i>pH</i>	<i>Ca</i>	<i>Mg</i>	<i>Na</i>	<i>K</i>	<i>Cl</i>	SO_4^{2-}	<i>Alka-</i> <i>linity</i>	<i>DIC</i>	$\text{NO}_3\text{-N}$	NH_4^+	<i>Fe</i>	<i>Mn</i>	<i>Al</i>	<i>Ba</i>	<i>Si</i>	<i>Sr</i>	<i>Zn</i>	<i>DO</i>	<i>Eh</i>	<i>TDS</i>
3-1	20.6	7.95	45.2	14.0	334	0.62	3.9	60.4	115	236	<0.02	<0.02	0.440	0.149	0.012	0.013	4.89	0.052	0.008	2.0	185	249
3-2	18.5	7.76	52.8	15.0	209	0.95	2.9	64.4	136	282	<0.02	<0.02	0.470	0.537	0.007	0.013	3.79	0.063	0.006	2.2	150	258
3-3	19.8	7.77	51.5	14.5	277	0.57	2.8	34.8	125	259	<0.02	<0.02	0.176	0.407	0.007	0.016	3.87	0.059	0.006	2.0	222	265
3-4	19.2	7.70	69.6	27.5	241	0.61	3.5	21.6	219	457	12.9	<0.02	0.000	0.007	0.007	0.030	9.37	0.120	0.007	5.0	324	314
3-5	17.6	7.45	72.6	30.8	229	0.60	2.9	21.0	247	533	13.1	<0.02	0.006	0.011	0.007	0.029	8.71	0.155	0.007	4.4	323	368
3-6	19.4	7.30	71.9	22.5	647	1.41	4.1	31.2	232	516	5.42	0.02	0.006	0.196	0.010	0.076	7.60	0.160	0.011	5.4	340	325
3-7	19.4	6.93	55.4	11.2	232	1.35	1.8	25.9	159	401	2.06	<0.02	0.030	0.121	0.010	0.039	6.38	0.113	0.012	4.0	339	233
3-8	19.2	7.21	70.7	16.9	6.61	1.71	2.9	39.7	211	480	3.16	0.04	0.057	0.313	0.004	0.098	5.98	0.168	0.008	2.4	307	314
7-1	18.5	7.64	60.1	15.2	234	1.23	3.1	50.8	156	328	0.361	0.03	0.863	0.460	0.011	0.025	3.56	0.073	0.017	1.0	135	289
7-2	17.8	7.29	87.3	26.3	214	1.24	3.8	26.0	272	606	6.78	<0.02	0.070	0.308	0.009	0.050	7.71	0.295	0.042	1.2	286	355
7-3	18	7.62	83.1	30.5	252	1.23	4.9	17.9	274	577	9.94	0.03	0.005	0.095	0.000	0.042	9.70	0.178	0.034	5.6	356	350
7-4	19.5	7.38	115	46.1	299	1.31	4.0	16.4	403	881	15.6	<0.02	0.004	0.042	0.010	0.044	10.0	0.205	0.034	5.1	352	512
7-5	18.1	7.24	121.9	49.4	3.12	0.77	2.8	19.3	418	943	20.1	<0.02	0.004	0.060	0.009	0.041	9.44	0.205	0.040	5.7	365	542
7-6	19.7	7.38	93.2	40.7	2.50	1.01	3.1	17.8	358	7.82	8.13	<0.02	0.010	0.058	0.007	0.034	9.64	0.225	0.034	6.5	368	418
7-7	20.6	7.28	90.5	20.6	1.26	1.49	1.6	17.3	291	650	2.08	<0.02	0.026	0.322	0.015	0.052	8.40	0.224	0.044	2.8	341	343
7-8	21.5	7.18	89.0	18.5	1.20	1.64	1.6	14.6	278	638	3.84	<0.02	0.007	0.289	0.011	0.061	6.80	0.212	0.037	2.2	371	341
11-1	20.3	7.15	72.8	13.5	3.08	0.83	5.4	35.0	191	442	0.11	<0.02	0.051	0.195	0.017	0.117	5.69	0.103	0.013	1.3	276	288
11-2	21.2	7.39	66.6	18.7	3.66	0.65	10.0	48.1	184	401	0.05	<0.02	0.111	0.187	0.013	0.074	6.12	0.085	0.011	1.1	251	280
11-3	21.3	7.75	59.9	15.3	3.68	0.48	9.1	57.1	152	3.16	<0.02	<0.02	0.134	0.177	0.008	0.036	5.62	0.058	0.014	1.9	244	266
11-4	20.7	7.95	46.0	15.0	3.20	0.60	2.7	66.4	111	227	<0.02	<0.02	0.474	0.139	0.005	0.013	5.01	0.052	0.017	1.5	190	258
11-5	20.9	7.28	59.4	12.1	2.93	0.77	2.6	61.1	143	3.19	<0.02	<0.02	0.004	0.268	0.008	0.092	3.91	0.076	0.015	2.2	350	296
11-6	19.4	7.34	73.5	14.4	2.80	1.08	2.4	58.6	183	403	<0.02	<0.02	0.005	0.300	0.022	0.110	3.87	0.098	0.013	1.3	288	288
11-7	19.8	7.09	88.9	11.1	1.56	2.04	0.8	12.3	252	5.95	<0.02	<0.02	0.044	1.133	0.008	0.120	4.47	0.162	0.017	1.0	285	283
11-8	22.9	7.06	66.2	11.8	0.94	2.39	0.7	6.8	210	5.02	0.11	<0.02	0.003	0.010	0.013	0.136	3.79	0.092	0.010	2.0	355	225

Notes: Each MLS is identified by number and port. Alkalinity is as mg/L CaCO₃, DIC is in mmol/L, Eh is in mV, all other concentrations except pH are in mg/L.
 $^{\circ}\text{C}$ = degrees Centigrade.

Table B3. Complete Chemical Analyses, October 14, 1993

MLS	°C	pH	Ca	Mg	Na	K	Cl	SO ₄ ²⁻	Alka- linity	DIC	NO ₃ -N	NH ₄ ⁺	Fe	Mn	Al	Ba	Si	Sr	Zn	DO	Eh	TDS	Conduc- tivity
3-1	15.6	7.73	48.5	15.1	3.11	0.98	3.4	64.2	133	2.77	<0.02	<0.02	0.389	0.156	<0.013	0.013	4.86	0.057	0.059	1.9	178	231	300
3-2	14.9	7.53	54.2	15.7	2.24	0.83	3.0	62.5	142	3.03	<0.02	<0.02	0.379	0.576	0.013	0.012	3.89	0.066	0.024	1.5	*	244	320
3-3	15.1	7.60	53.2	14.9	2.55	1.24	3.0	57.2	142	3.00	<0.02	<0.02	0.264	0.401	<0.013	0.013	4.05	0.061	0.017	1.3	273	249	315
3-4	15.4	7.55	61.2	24.6	2.27	0.86	3.3	28.9	194	4.12	9.30	<0.02	<0.003	0.013	<0.013	0.018	9.18	0.106	0.014	2.9	422	293	408
3-5	15.7	7.52	60.5	23.0	2.37	0.57	4.5	24.9	185	3.95	9.19	<0.02	0.062	0.065	<0.013	0.026	7.93	0.121	0.017	4.4	363	271	400
3-6	16.2	7.48	70.0	27.2	2.68	1.12	5.2	32.5	122	2.62	4.11	<0.02	0.120	0.103	<0.013	0.027	8.22	0.146	0.013	3.4	328	322	430
3-7	17.4	7.10	51.3	14.9	1.36	1.17	3.4	17.3	161	3.79	1.20	<0.02	0.086	0.118	<0.013	0.029	4.30	0.177	0.016	1.8	345	205	310
3-8	17.2	7.38	58.0	16.0	2.28	1.44	3.3	29.1	183	3.99	2.01	0.03	0.140	0.302	0.013	0.033	5.32	0.124	0.014	1.7	290	241	355
7-1	15.9	7.51	66.4	17.5	2.62	0.90	3.7	54.5	191	4.08	<0.02	<0.02	1.33	0.494	<0.013	0.028	3.45	0.072	0.017	1.1	142	278	395
7-2	15.5	7.28	79.5	27.2	2.78	1.18	4.7	37.3	258	5.76	6.20	<0.02	0.494	0.402	0.027	0.036	7.26	0.125	0.024	1.1	210	331	500
7-3	15.4	7.43	79.8	28.6	2.73	1.24	4.2	26.1	279	6.04	7.40	<0.02	0.319	0.155	0.014	0.041	8.55	0.141	0.025	3.5	260	*	510
7-4	15.6	7.28	110	45.4	3.30	1.04	3.4	17.7	408	9.11	12.6	<0.02	0.050	0.024	<0.013	0.079	9.99	0.190	0.049	4.3	333	*	660
7-5	15.7	7.28	89.9	34.2	2.68	0.82	3.6	23.4	312	6.97	8.70	<0.02	0.259	0.127	<0.013	0.040	8.06	0.142	0.029	3.8	267	388	520
7-6	16.7	7.38	84.2	35.5	2.62	1.05	4.2	20.3	307	6.71	7.30	0.03	0.146	0.072	<0.013	0.045	10.6	0.309	0.037	4.0	292	266	510
7-7	18.1	7.41	59.0	11.8	1.00	1.26	2.5	11.4	190	4.13	0.20	<0.02	0.249	0.254	<0.013	0.043	3.21	0.206	0.020	0.8	211	216	335
7-8	18.7	7.26	69.1	13.3	0.85	1.35	2.7	8.5	224	5.03	0.09	<0.02	0.141	0.274	<0.013	0.070	4.40	0.147	0.020	0.7	279	229	375
11-1	18.4	7.30	61.9	18.1	3.28	0.87	8.9	45.6	185	4.11	<0.02	0.05	1.22	0.238	<0.013	0.075	6.16	0.080	0.012	1.3	128	253	410
11-2	17.4	7.56	59.0	18.6	3.25	1.09	9.2	45.4	181	3.84	<0.02	0.02	0.750	0.244	<0.013	0.053	5.70	0.075	0.010	1.1	132	254	385
11-3	17.6	7.94	56.1	14.6	3.54	0.65	10.2	50.0	152	3.12	<0.02	<0.02	0.379	0.168	<0.013	0.029	5.44	0.052	0.009	1.1	122	311	350
11-4	17.0	7.98	46.6	15.1	3.13	0.64	2.7	68.2	119	2.43	<0.02	<0.02	0.494	0.138	<0.013	0.017	4.67	0.052	0.011	1.3	104	216	305
11-5	16.9	7.83	54.8	15.9	2.86	1.20	2.9	63.5	150	3.10	<0.02	<0.02	0.511	0.290	<0.013	0.024	3.86	0.061	0.006	0.9	115	230	348
11-6	17.1	7.40	80.4	28.6	2.75	1.87	3.7	58.1	276	6.01	<0.02	<0.02	0.194	0.385	<0.013	0.069	3.29	0.084	0.008	0.6	191	314	480
11-7	17.5	7.59	44.1	8.22	0.67	2.82	1.7	6.8	154	3.26	<0.02	<0.02	0.209	0.400	<0.013	0.049	2.78	0.075	0.010	0.4	145	156	265
11-8	18.9	7.80	32.9	10.2	0.27	3.38	1.9	2.1	136	2.81	<0.02	0.34	0.010	0.104	<0.013	0.058	2.26	0.082	0.010	*	265	125	230

Notes: Each MLS is identified by number and port. Alkalinity is as mg/L CaCO₃, DIC is in mmol/L, Eh is in mV, conductivity is in µmho/cm, all other concentrations except pH are in mg/L.

* = Not reported.

°C = degrees Centigrade.

Table B4. Complete Chemical Analyses, February 4, 1994

<i>MLS</i>	$^{\circ}\text{C}$	<i>pH</i>	<i>Ca</i>	<i>Mg</i>	<i>Na</i>	<i>K</i>	<i>Cl</i>	SO_4^{2-}	<i>Alka-</i> <i>linity</i>	<i>DIC</i>	$\text{NO}_3\text{-N}$	NH_4^+	<i>Fe</i>	<i>Mn</i>	<i>Al</i>	<i>Ba</i>	<i>Sr</i>	<i>Zn</i>	<i>DO</i>	<i>Eh</i>	<i>TDS</i>	<i>DOC</i>	
3-1	12.9	7.68	58.1	19.1	3.01	<0.58	3.4	61.6	166	347	<0.02	<0.02	0.625	0.163	<0.011	0.019	5.05	0.066	0.008	2.2	94.0	250	*
3-2	11.1	7.48	81.1	26.3	2.92	0.78	3.7	59.7	250	537	<0.02	<0.02	0.799	0.843	<0.011	0.021	3.84	0.106	0.008	2.8	127.3	335	*
3-3	11.8	7.06	166	54.3	4.61	1.16	22.1	58.2	576	1376	<0.02	<0.02	2.23	0.903	<0.011	0.042	5.51	0.200	0.008	2.1	156.3	665	*
3-4	11.9	7.24	104	41.4	4.24	0.87	6.7	27.3	509	1148	7.61	<0.02	0.314	0.098	<0.011	0.034	9.73	0.180	0.007	3.4	172.0	450	*
3-5	11.2	7.18	110	45.6	3.84	<0.58	4.4	21.4	389	893	15.6	<0.02	0.163	0.053	<0.011	0.085	9.56	0.250	0.011	3.5	208.7	487	*
3-6	10.8	7.22	94.2	34.2	4.56	1.12	4.3	31.3	309	701	7.10	<0.02	0.537	0.187	<0.011	0.047	7.98	0.189	0.008	2.4	144.5	396	4.7
3-7	9.5	7.15	68.0	18.4	5.03	1.61	3.1	33.3	186	431	2.02	<0.02	0.443	0.202	<0.011	0.052	5.35	0.167	0.008	1.7	163.0	251	3.2
7-1	9.7	7.48	76.4	21.8	2.92	0.83	4.6	57.7	226	485	<0.02	<0.02	1.44	0.528	<0.011	0.027	3.41	0.082	0.009	1.1	95.0	309	3.0
7-2	10.5	7.41	77.5	25.3	2.51	0.66	3.9	31.9	246	535	5.62	<0.02	0.573	0.356	<0.011	0.030	6.47	0.118	0.012	0.6	105.0	322	3.3
7-3	11.2	7.41	86.8	31.6	2.65	0.92	3.5	21.6	311	676	7.65	<0.02	0.332	0.127	<0.011	0.046	8.98	0.157	0.019	5.6	146.3	370	3.9
7-4	11.2	7.39	77.6	30.9	2.46	0.85	4.2	14.9	268	585	12.0	<0.02	0.067	0.025	<0.011	0.061	9.36	0.138	0.023	4.5	179.3	331	3.7
7-5	10.9	7.30	82.2	30.3	2.36	<0.58	3.8	16.6	279	620	9.48	<0.02	0.210	0.085	<0.011	0.039	7.84	0.133	0.013	3.9	113.0	337	3.9
7-6	10.3	7.19	102	40.2	2.50	<0.58	3.5	17.1	371	849	6.13	<0.02	0.117	0.048	<0.011	0.048	11.3	0.376	0.039	4.6	166.0	412	5.1
7-7	9.2	7.40	59.1	12.8	0.938	1.67	3.2	16.2	186	405	1.25	<0.02	0.150	0.226	<0.011	0.034	3.15	0.172	0.011	1.2	160.5	212	3.2
7-8	8.6	7.32	75.3	16.9	0.975	1.04	3.0	15.2	237	524	1.20	<0.02	0.202	0.264	<0.011	0.064	3.93	0.178	0.013	3.9	172.0	261	3.4
11-1	9.0	7.32	65.6	19.2	3.18	1.17	8.8	39.1	202	447	<0.02	0.04	0.696	0.281	<0.011	0.079	6.06	0.087	0.007	0.7	23.3	255	3.0
11-2	9.9	7.60	58.2	19.6	3.39	1.00	9.4	47.7	175	369	<0.02	<0.02	0.449	0.237	0.016	0.049	6.02	0.074	0.024	0.0	21.7	258	2.5
11-3	10.6	7.92	60.4	15.2	3.82	0.84	11.5	52.0	150	308	<0.02	<0.02	0.430	0.178	<0.011	0.030	5.40	0.057	0.029	0.0	73.3	240	2.0
11-4	10.6	7.96	46.2	14.9	3.34	0.65	3.5	69.0	112	228	<0.02	<0.02	0.525	0.138	0.011	0.017	4.90	0.053	0.031	0.2	53.0	211	3.6
11-5	10.4	7.78	58.0	17.4	3.02	0.97	2.7	66.4	154	319	<0.02	<0.02	0.684	0.244	<0.011	0.028	4.17	0.067	0.025	0.0	60.0	247	2.1
11-6	10.4	7.36	72.5	24.3	3.31	1.10	3.8	65.0	224	492	<0.02	<0.02	0.260	0.583	<0.011	0.037	3.44	0.086	0.014	0.3	120.5	301	2.8
11-7	9.4	7.33	64.3	16.1	1.06	2.61	3.1	11.7	212	468	<0.02	<0.02	1.46	0.685	0.021	0.091	3.90	0.148	0.083	0.0	141.0	229	3.2
11-8	7.8	7.23	63.4	17.6	0.301	2.09	2.3	13.5	208	471	1.44	<0.02	0.007	0.002	<0.011	0.145	3.09	0.177	0.019	5.5	176.0	220	3.6

Notes: Each MLS is identified by number and pot. Alkalinity is as mg/L CaCO_3 , DIC is in mmol/L, Eh is in mV, all other concentrations except pH are in mg/L.

* = Not reported.

$^{\circ}\text{C}$ = degrees Centigrade.

Table B5. Combined Complete Chemical Analyses, April 14 and 19, 1994

<i>MLS</i>	$^{\circ}\text{C}$	<i>pH</i>	<i>Ca</i>	<i>Mg</i>	<i>Na</i>	<i>K</i>	<i>Cl</i>	<i>SO₄²⁻</i>	<i>Alka- linity</i>	<i>DIC</i>	<i>NO₃-N</i>	<i>NH₄⁺</i>	<i>Fe</i>	<i>Mn</i>	<i>Al</i>	<i>Ba</i>	<i>Si</i>	<i>Sr</i>	<i>Zn</i>	<i>DO</i>	<i>Eh</i>	<i>TDS</i>	<i>DOC</i>	<i>Conduc- tivity</i>
3-1	16.5	7.48	56.2	16.8	2.88	0.56	3.8	66.5	167	3.59	<0.02	<0.02	0.588	0.164	<0.011	0.015	4.60	0.065	0.013	1.9	100.5	278	0.5	330
3-2	16.7	7.16	79.2	26.8	2.81	0.77	3.6	65.3	262	6.05	<0.02	0.02	0.776	0.853	<0.011	0.018	3.92	0.102	0.011	2.5	157.0	358	0.5	450
3-3	16.4	7.31	71.7	24.2	2.80	0.85	4.0	65.5	234	5.19	<0.02	0.02	0.838	0.468	<0.011	0.017	3.89	0.085	0.012	1.9	142.5	331	0.5	410
3-4	16.0	6.95	105	44.9	3.17	0.72	4.9	21.6	412	10.30	9.05	<0.02	0.024	0.012	<0.011	0.030	10.5	0.192	0.006	5.1	275.5	485	0.5	570
3-5	15.4	7.01	89.4	36.9	2.87	0.735	5.0	23.6	335	8.16	8.77	<0.02	0.063	0.031	<0.011	0.044	8.31	0.200	<0.006	4.9	265.0	421	0.4	490
3-6	15.2	7.11	74.6	29.2	2.25	0.786	2.6	30.4	297	6.97	1.47	<0.02	0.149	0.075	<0.011	0.024	7.55	0.147	0.006	4.8	233.0	352	0.4	440
3-7	14.4	6.88	42.3	12.7	1.81	1.520	2.7	33.6	135	3.49	2.53	<0.02	0.121	0.066	<0.011	0.027	7.70	0.111	0.007	2.6	243.7	218	0.4	245
7-1	17.1	7.22	66.4	19.0	2.71	0.835	4.4	65.6	208	4.72	<0.02	<0.02	1.27	0.510	<0.011	0.019	3.03	0.074	0.011	1.7	127.0	306	0.5	490
7-2	16.8	7.24	67.7	23.7	2.46	0.931	4.1	44.0	240	5.42	5.75	0.02	0.368	0.285	<0.011	0.019	5.74	0.102	0.030	1.9	179.3	323	0.5	420
7-3	16.7	7.22	67.5	25.7	2.34	1.147	3.6	23.9	252	5.72	10.7	<0.02	0.156	0.071	<0.011	0.022	7.60	0.126	0.030	4.5	278.5	341	0.4	430
7-4	16.1	7.16	75.3	30.4	2.49	1.390	4.0	18.1	300	6.93	12.0	<0.02	0.041	0.018	<0.011	0.064	8.42	0.144	0.034	6.1	294.7	396	0.4	450
7-5	15.6	7.25	61.1	24.5	2.07	0.775	3.5	13.8	229	5.15	12.1	<0.02	0.014	0.007	<0.011	0.030	6.87	0.102	0.023	7.6	296.7	306	0.6	395
7-6	15.9	7.00	88.7	37.0	2.49	0.783	7.3	18.2	359	8.78	11.2	<0.02	0.007	0.003	<0.011	0.033	10.1	0.366	0.041	4.9	296.7	440	0.8	525
7-7	15.8	7.17	55.3	11.0	1.04	1.950	2.8	25.1	183	4.21	2.03	<0.02	0.068	0.104	<0.011	0.033	4.78	0.163	0.031	2.9	248.5	245	0.8	300
11-1	19.5	7.64	60.1	16.9	2.97	1.07	8.5	39.8	201	4.22	<0.02	0.06	0.563	0.255	<0.011	0.065	5.75	0.079	0.012	2.0	46.3	256	0.9	370
11-2	20.1	7.96	54.4	18.3	3.32	0.53	9.9	50.9	177	3.62	<0.02	<0.02	0.393	0.232	<0.011	0.033	5.53	0.070	0.026	2.6	53.7	249	0.6	360
11-3	18.2	8.18	41.3	12.7	3.04	0.71	3.2	69.2	113	2.29	<0.02	<0.02	0.456	0.124	<0.011	0.013	4.55	0.049	0.022	2.1	80.5	203	0.5	320
11-4	17.9	8.25	52.6	15.3	2.81	0.66	3.3	66.3	154	3.12	<0.02	<0.02	0.652	0.215	0.011	0.018	3.87	0.063	0.033	1.2	67.3	238	0.6	275
11-5	17.6	8.08	55.5	16.0	2.59	0.72	3.1	66.2	162	3.30	<0.02	<0.02	0.192	0.480	0.011	0.021	2.84	0.065	0.047	1.0	95.3	244	0.8	320
11-6	18.6	7.86	54.9	13.2	3.42	0.33	8.2	60.4	150	3.09	<0.02	<0.02	0.365	0.164	0.019	0.022	5.39	0.054	0.055	3.4	148.0	237	0.6	345
11-7	15.9	7.67	62.2	13.9	0.71	2.20	2.8	19.5	210	4.40	0.07	<0.02	0.652	0.599	0.012	0.077	2.85	0.145	0.011	0.7	128.3	228	1.2	310
11-8	13.5	7.31	54.7	14.3	0.34	2.46	1.5	18.5	179	3.97	4.24	<0.02	0.004	0.002	<0.011	0.153	4.78	0.153	0.010	4.8	238.0	207	1.2	280
18-1	16.3	7.95	60.0	14.2	3.78	0.00	11.3	69.6	154	3.16	<0.02	<0.02	0.338	0.197	0.027	0.030	5.32	0.059	0.109	1.1	131.7	257	0.6	285
18-2	17.6	8.03	45.3	12.6	3.26	0.00	3.6	73.6	110	2.24	<0.02	<0.02	0.460	0.149	0.013	0.015	5.22	0.052	0.033	2.6	122.5	209	2.0	280
18-3	17.2	8.05	43.6	13.7	2.92	0.70	3.9	71.7	119	2.43	<0.02	<0.02	0.193	0.137	<0.011	0.017	4.16	0.058	0.011	0.7	108.7	214	14.5	280
18-4	17.4	8.00	48.6	14.3	3.01	0.70	2.8	63.7	141	2.88	0.10	<0.02	0.184	0.194	<0.011	0.020	3.18	0.061	0.009	1.1	128.7	218	6.9	300
18-5	18.0	8.01	46.8	14.5	2.52	0.58	2.2	39.3	130	2.66	10.3	<0.02	0.157	0.179	0.012	0.020	4.83	0.066	0.012	3.5	135.5	220	19.1	310
18-6	17.6	7.99	52.5	14.8	3.01	0.62	4.8	56.2	143	2.92	4.92	<0.02	0.267	0.205	0.012	0.021	4.87	0.077	0.017	1.9	114.7	243	19.6	320
18-7	16.5	8.04	50.3	13.8	1.80	1.02	1.2	42.6	132	2.69	9.59	<0.02	0.034	0.434	<0.011	0.017	3.98	0.104	0.025	1.5	168.0	224	3.2	295
18-8	16.4	7.72	72.0	27.1	1.85	0.92	2.7	72.4	210	4.38	9.40	<0.02	0.018	0.717	0.014	0.033	4.27	0.124	0.038	1.1	231.7	351	1.9	440
18-9	16.1	7.29	96.7	22.6	2.47	0.75	1.5	73.1	293	6.53	1.16	<0.02	0.530	2.080	<0.011	0.056	5.80	0.234	0.038	1.8	212.3	402	4.1	500

Notes: MLS is identified by number and port. Alkalinity is as mg/L CaCO₃, DIC is in mmol/L, Eh is in mV, conductivity is in $\mu\text{mho/cm}$, all other concentrations except pH are in mg/L.
 $^{\circ}\text{C}$ = degrees Centigrade.

Table B6. Combined Complete Chemical Analyses, July 19 and 21, 1994

MLS	°C	pH	Ca	Mg	Na	K	Cl	SO ₄ ²⁻	Alka- linity	DIC	NO ₃ -N	NH ₄ ⁺	Fe	Mn	Al	Ba	Si	Sr	Zn	Eh	TDS	DOC	Conduc- tivity
3-1	16.2	7.71	54.6	17.3	2.88	0.78	3.2	59.7	164	3.42	0.37	<0.02	0.440	0.152	<0.011	0.032	4.54	0.065	0.109	128.3	279	0.5	335
3-2	15.7	7.41	78.4	27.4	2.80	1.05	3.3	60.8	264	5.74	0.30	<0.02	0.709	0.805	<0.011	0.021	4.11	0.102	0.018	129.0	362	0.6	460
3-3	15.3	7.48	73.8	23.6	2.82	0.66	3.4	56.9	238	5.11	0.54	<0.02	0.726	0.442	<0.011	0.037	4.57	0.089	0.035	158.0	346	0.5	425
3-4	15.5	7.45	67.4	26.3	2.44	0.88	2.5	24.1	242	5.22	7.99	<0.02	0.025	0.020	<0.011	0.028	8.58	0.122	0.014	260.0	328	0.5	480
3-5	16.0	7.24	88.2	35.2	2.67	<0.58	2.1	17.3	328	7.40	9.17	<0.02	0.021	0.015	<0.011	0.060	9.04	0.218	0.011	293.0	408	0.6	555
3-6	16.2	7.36	74.1	28.1	1.94	0.71	2.1	16.7	287	6.30	0.57	<0.02	0.078	0.038	<0.011	0.043	8.89	0.153	0.046	269.0	330	0.5	470
3-7	16.5	7.00	41.8	13.4	1.83	1.40	2.3	28.6	127	3.11	2.57	<0.02	0.215	0.086	<0.011	0.038	7.96	0.102	0.016	299.0	213	0.4	245
7-1	19.0	7.46	61.4	19.5	2.77	1.52	5.0	67.3	188	4.05	<0.02	<0.02	1.14	0.525	0.136	0.028	3.02	0.074	0.129	177.0	298	0.4	380
7-2	19.2	7.34	76.8	27.7	2.56	1.16	5.9	41.0	270	5.95	7.22	<0.02	0.115	0.268	0.011	0.022	6.90	0.134	0.081	307.7	360	0.5	495
7-3	17.4	7.46	82.5	31.0	2.54	1.06	7.1	24.3	296	6.38	8.11	<0.02	0.067	0.039	<0.011	0.031	8.91	0.174	0.047	375.0	368	0.5	500
7-4	17.8	7.43	77.9	30.1	2.42	1.02	6.6	18.6	277	6.00	10.22	<0.02	0.018	0.011	<0.011	0.050	9.19	0.162	0.036	391.0	345	0.5	495
7-5	18.5	7.36	91.9	36.5	2.67	0.66	15.3	18.6	320	7.02	10.68	<0.02	0.008	0.006	0.016	0.029	8.23	0.160	0.039	400.0	413	0.5	520
7-6	18.4	7.25	92.8	38.1	2.57	<0.58	10.7	16.5	342	7.70	10.16	<0.02	0.013	0.008	<0.011	0.031	10.7	0.348	0.052	398.7	424	0.5	550
7-7	18.8	7.45	57.5	14.6	1.34	1.51	4.1	17.3	177	3.82	2.54	<0.02	0.024	0.071	<0.011	0.026	7.51	0.143	0.034	366.0	231	0.4	315
11-1	19.5	7.57	53.6	17.7	2.83	1.21	5.8	43.1	167	3.54	1.01	0.02	0.636	0.143	<0.011	0.075	6.04	0.082	0.013	138.7	267	0.6	390
11-2	20.0	7.84	52.9	19.0	3.13	1.02	6.4	44.0	172	3.55	0.61	<0.02	0.415	0.171	<0.011	0.059	5.73	0.076	0.012	112.3	284	0.5	385
11-3	19.3	8.00	54.9	13.5	3.50	0.89	5.8	58.3	142	2.90	<0.02	<0.02	0.414	0.160	0.043	0.035	5.17	0.051	0.023	105.0	259	0.4	365
11-4	19.1	8.11	42.9	13.9	3.15	0.85	2.9	61.9	134	2.73	<0.02	<0.02	0.492	0.131	<0.011	0.025	4.57	0.049	0.018	67.5	226	0.4	320
11-5	18.8	7.91	53.6	16.4	2.88	0.92	2.4	59.0	184	3.78	<0.02	<0.02	0.712	0.216	<0.011	0.029	3.66	0.062	0.013	108.0	268	0.5	350
11-6	19.7	7.59	54.8	16.1	2.92	1.56	2.4	59.2	162	3.42	<0.02	<0.02	0.083	0.489	<0.011	0.030	2.71	0.060	0.009	152.0	278	0.4	370
11-7	18.5	7.51	62.8	16.3	0.962	2.03	2.3	13.3	226	4.83	0.14	<0.02	0.309	0.618	<0.011	0.085	4.42	0.155	0.007	174.5	257	0.6	360
16-1	17.1	7.26	112	43.7	2.74	<0.58	9.5	17.6	398	8.93	15.02	<0.02	<0.003	<0.001	0.013	0.050	9.10	0.248	0.015	436.0	488	0.6	560
16-2	17.7	7.19	107	41.3	2.68	<0.58	6.2	15.6	399	9.13	8.55	<0.02	<0.003	0.001	<0.011	0.034	9.35	0.199	0.015	437.0	464	0.5	600
16-3	17.3	7.19	105	40.1	2.63	<0.58	5.5	14.2	397	9.08	5.64	<0.02	<0.003	<0.001	<0.011	0.023	9.65	0.224	0.013	449.0	448	0.8	590
16-4	17.3	7.21	97.9	37.9	2.52	<0.58	5.7	14.5	385	8.76	4.63	<0.02	<0.003	0.001	<0.011	0.030	9.66	0.252	0.015	447.0	421	0.8	550
16-5	19.0	7.26	91.0	38.7	2.75	<0.58	4.4	10.4	379	8.51	1.84	<0.02	0.003	<0.001	<0.011	0.032	10.6	0.250	0.013	452.0	393	0.7	535
16-6	19.0	7.26	89.1	36.7	1.99	<0.58	4.8	10.8	361	8.10	2.77	<0.02	<0.003	<0.001	0.013	0.053	11.0	0.338	0.011	452.0	379	0.7	520
16-7	17.8	7.39	84.7	33.0	1.68	<0.58	3.3	7.7	348	7.59	0.95	<0.02	0.007	<0.001	0.012	0.060	12.1	0.430	0.006	463.0	373	0.6	475
16-8	18.5	7.48	79.9	32.8	1.88	<0.58	3.2	10.3	324	6.96	2.72	<0.02	<0.003	<0.001	0.012	0.034	11.3	0.303	0.011	460.0	350	0.6	465
16-9	20.9	7.37	78.1	25.4	1.92	<0.58	5.9	15.6	267	5.85	5.19	<0.02	<0.003	<0.001	0.011	0.040	8.89	0.174	0.011	462.0	334	0.6	450
16-10	19.9	7.60	53.3	11.1	1.20	<0.58	2.3	12.5	168	3.55	1.91	<0.02	<0.003	<0.001	0.011	0.032	8.38	0.112	0.013	452.0	189	0.5	290
16-11	18.7	7.33	65.6	21.8	1.88	<0.58	5.7	15.5	206	4.55	4.79	<0.02	<0.003	<0.001	0.011	0.052	9.41	0.139	0.012	469.0	281	0.5	365

Notes: Each MLS is identified by number and port. Alkalinity is as mg/L CaCO₃, DIC is in mmol/L, Eh is in mV, conductivity is in μmho/cm, all other concentrations except pH are in mg/L.
°C = degrees Centigrade.

**Table B7. Nitrate Data (mg/L) from November 1993 and November 1994
and Tritium Data (TU) from April 1994**

<i>11/17/93</i>		<i>11/3/94</i>								<i>4/12/94</i>	
<i>Nitrate data</i>		<i>Nitrate data</i>								<i>Tritium data</i>	
<i>MLS</i>	<i>NO₃⁻-N</i>	<i>MLS</i>	<i>NO₃⁻-N</i>	<i>MLS</i>	<i>NO₃⁻-N</i>	<i>MLS</i>	<i>NO₃⁻-N</i>	<i>MLS</i>	<i>NO₃⁻-N</i>	<i>MLS</i>	<i>Tritium</i>
1-1	<0.02	1-1	0.12	8-1	0.24	13-1	2.07	17-1	9.78	3-1	9.7
1-2	<0.02	1-2	0.03	8-2	0.06	13-2	2.10	17-2	10.5	3-2	7.2
1-3	0.2	1-3	0.06	8-3	2.56	13-3	5.88	17-3	10.9	3-3	9.4
1-4	52.2	1-4	0.09	8-4	9.06	13-4	3.60	17-4	10.8	3-4	7.7
1-5	21.8	1-6	0.08	8-5	8.84	13-5	7.64	17-5	3.62	3-5	6.4
1-6	17.4	2-2	0.07	8-6	0.35	13-6	8.31	17-6	2.04	3-6	6.4
1-7	2.4	2-3	0.07	9-1	0.20	13-7	5.14	17-7	1.20	3-7	7.6
1-8	0.6	2-4	0.10	9-2	0.10	13-8	1.74	17-8	2.38	11-1	16.6
2-1	0.3	2-5	0.21	9-3	1.20	13-9	2.11	17-9	2.30	11-2	16.6
2-2	<0.02	2-6	0.11	9-4	10.8	13-10	1.50	17-10	1.76	11-3	19.7
2-3	<0.02	2-8	0.10	9-5	3.57	13-11	0.35	17-11	1.37	11-4	14.2
2-4	37.0	3-2	0.03	9-6	1.19	14-1	4.71	18-1	<0.02	11-5	10.1
2-5	66.2	3-3	0.04	10-1	1.33	14-2	10.6	18-2	<0.02	11-6	10.6
2-6	13.1	3-4	9.77	10-2	0.26	14-3	10.3	18-3	0.04	11-7	7.4
2-7	0.7	3-5	10.0	10-3	1.27	14-4	10.8	18-4	0.06	18-1	20.3
2-8	0.9	3-6	1.74	10-4	10.9	14-5	10.7	18-5	9.72	18-2	14.1
3-1	<0.02	4-1	2.22	10-5	2.95	14-6	8.86	18-6	8.65	18-3	11.0
3-2	<0.02	4-2	0.04	10-6	1.16	14-7	4.87	18-7	9.61	18-4	10.3
3-3	<0.02	4-3	<0.02	11-1	0.06	14-8	1.86	18-8	10.3	18-5	8.2
3-4	39.3	4-4	8.82	11-2	<0.02	14-9	1.91			18-6	8.9
3-5	51.9	4-5	4.06	11-3	<0.02	14-10	1.31			18-7	7.1
3-6	26.8	4-6	1.51	11-4	<0.02	14-11	1.26			18-8	8.3
3-7	5.0	5-1	1.29	11-5	<0.02	15-1	9.59			18-9	9.5
3-8	0.4	5-2	0.33	11-6	<0.02	15-2	10.6				
4-1	<0.02	5-3	1.15	11-7	0.04	15-3	9.85				
4-2	<0.02	5-4	1.40	11-8	0.05	15-4	10.5				
4-3	<0.02	5-5	1.36	12-1	6.35	15-5	8.93				
4-4	32.8	5-6	0.70	12-2	6.82	15-6	7.47				
4-5	58.6	5-8	0.82	12-3	7.04	15-7	3.69				
4-6	31.5	6-1	1.22	12-4	9.21	15-8	1.70				
4-7	8.5	6-2	4.86	12-5	10.1	15-9	2.09				
4-8	1.7	6-3	11.0	12-6	9.53	15-10	2.60				
5-1	0.3	6-4	6.03	12-7	9.52	15-11	1.02				
5-2	0.3	6-5	7.10	12-8	4.26	16-1	7.20				
5-3	0.3	6-6	0.13	12-9	2.09	16-2	5.09				
5-4	46.7	7-1	<0.02	12-10	2.58	16-3	1.95				
5-5	59.8	7-2	<0.02	12-11	1.80	16-4	2.47				
5-6	5.1	7-3	2.56			16-5	0.95				
5-7	0.3	7-4	10.8			16-6	0.89				
5-8	0.3	7-5	9.06			16-7	0.98				
		7-6	2.06			16-8	0.75				

**Table B8. Bromide Data from the Natural-Gradient Tracer Test
Started September 20, 1993**

<i>Date</i>	<i>Time</i>	<i>Days</i>	<i>Hours</i>	<i>I-3</i>	<i>I-5</i>	<i>MLS-5-5</i>
9/20/93	11:00 AM	0.0	0.0	363	*	*
9/21/93	12:20 PM	1.1	25.3	360	*	
9/23/93	12:05 PM	3.0	73.1	38.6	7.21	
9/27/93	12:45 PM	7.1	169.8	2.45	2.69	
10/1/93	11:30 AM	11.0	264.5	0.52	0.33	
10/4/93	11:00 AM	14.0	336.0	*	*	
10/8/93	8:55 AM	17.9	429.9			*
10/12/93	10:00 AM	22.0	527.0			34.4
10/13/93	9:55 AM	23.0	550.9			89.5
10/14/93	9:30 AM	23.9	574.5			176.3
10/14/93	3:55 PM	24.2	580.9			173.4
10/15/93	10:00 AM	25.0	599.0			245.1
10/18/93	10:45 AM	28.0	671.8			237.7
10/20/93	9:45 AM	29.9	718.8			201.3
10/22/93	9:40 AM	31.9	766.7			97.1
10/25/93	10:05 AM	35.0	839.1			14.0
10/27/93	9:40 AM	36.9	886.7			4.15
10/29/93	10:10 AM	39.0	935.2			1.90

Notes: Results are in mg/L. Blank fields indicate well or port not sampled.

* Below detection.

Table B9. Bromide Data from the Natural-Gradient Tracer Test Initiated April 19, 1994

Date	Days	Well		MLS										
		I-1	I-3	13-1	13-2	13-3	13-4	13-5	13-6	13-7	13-8	13-9	13-10	13-11
4/19/94	0	318												
4/20/94	1	308	0.25											
4/21/94	2	320	1.56											
4/22/94	3	261	9.6											
4/25/94	6	18.4	235											
4/27/94	8	2.60	174											
4/28/94	9	1.89	93.4	*	*	*	0.92	2.36	2.46	*	35.6	30.9	1.70	5.60
4/29/94	10	0.53	54.7	*	*	*	6.31	4.21	15.2	*	120	110	27.4	6.78
5/2/94	13	*	1.34	0.57	0.83	8.06	16.3	13.3	23.1	1.82	168	237	119	17.0
5/3/94	14		*	0.84	1.70	7.42	24.5	17.1	34.0	2.97	215	261	146	31.9
5/4/94	15			2.48	4.23	12.9	27.4	20.1	35.6	17.4	271	243	153	29.1
5/5/94	16			3.29	20.9	13.9	28.7	19.5	36.8	71.4	286	223	150	19.8
5/6/94	17			5.63	18.8	16.4	28.9	18.0	40.0	230	273	175	116	23.3
5/9/94	20			3.91	6.30	9.88	18.0	7.62	14.3	93.3	115	21.4	40.6	21.8
5/11/94	22			3.62	4.98	5.19	9.14	3.34	5.75	12.0	23.5	7.15	14.5	10.0
5/13/94	24			2.55	1.75	1.83	4.87	2.85	3.17	1.13	2.20	3.40	4.35	0.74
5/17/94	28			1.28	1.85	0.72	0.52	0.61	0.64	*	0.55	0.98	1.31	0.89
5/25/94	36			1.01	*	*	*	*	*	*	*	*	*	*

Date	Days	MLS											
		14-1	14-2	14-3	14-4	14-5	14-6	14-7	14-8	14-9	14-10	14-11	
5/2/94	13			*	*	*	*	*	62.8	11.1	0.89	0.44	
5/3/94	14	*	*	*	*	0.55	1.94	6.99	191	100	7.57	*	
5/4/94	15	1.04	1.30	1.08	0.98	0.97	2.13	9.21	215	207	23.6	0.55	
5/5/94	16	1.36	2.32	1.87	1.61	1.29	2.20	10.8	232	252	67.6	1.75	
5/6/94	17	2.59	2.59	2.61	2.22	1.62	2.59	8.36	234	257	97.3	6.10	
5/9/94	20	0.96	1.97	1.82	1.47	1.03	1.42	7.96	230	170	51.3	6.41	
5/11/94	22	1.77	2.15	2.13	1.80	1.06	1.60	29.6	151	46.5	3.93	3.42	
5/13/94	24	1.23	1.73	1.57	1.49	1.99	2.82	71.2	37.7	6.94	0.50	3.04	
5/17/94	28	1.60	2.05	2.05	1.67	1.08	1.37	3.98	3.95	1.81	*	75.0	
5/25/94	36	0.78	0.70	0.80	0.75	*	*	1.08	1.00	*	*	*	

Date	Days	MLS									
		5-3	5-4	5-5	5-6	5-7	17-6	17-7	17-8	17-9	17-10
5/17/94	28		*	106	*	*					
5/20/94	31	0.45	0.35	246	0.58	0.30	0.35	6.02	0.40	11.6	0.66
5/23/94	34	*	*	215	0.52	*	*	*	*	1.63	*
5/25/94	36	1.38	*	213	1.44	*	*	*	*	*	*
5/27/94	38	0.57	*	138	0.78	*					
5/31/94	42	*	0.75	18.0	*	*					
6/2/94	44	0.58	*	4.66	0.81	*					
6/9/94	51	*	*	1.37	0.63	*					

Notes: Results are in mg/L. Blank fields indicate well or port not sampled.
 * Below detection.

Table B10. Nitrate Data from the Natural-Gradient Tracer Test Initiated April 19, 1994

<i>Date</i>	<i>Days</i>	<i>Well</i>				<i>MLS</i>								
		<i>I-1</i>	<i>I-2</i>	<i>I-3</i>	<i>I-5</i>	<i>13-1</i>	<i>13-2</i>	<i>13-3</i>	<i>13-4</i>	<i>13-5</i>	<i>13-6</i>	<i>13-7</i>	<i>13-8</i>	<i>13-9</i>
4/19/94	0	433				47.4	42.3	48.9	53.9	53.9	57.5	57.9	50.6	31.0
4/20/94	1	427		44.7										
4/21/94	2	423	19.3	45.1										
4/22/94	3	408	16.3	50.0										
4/25/94	6	81.1		431										
4/27/94	8	55.3		326										
4/28/94	9	44.3		185	20.0	39.7	44.6	49.2	53.0	57.7	62.9	57.2	111	94.7
4/29/94	10	47.8		128		41.0	47.2	48.8	62.7	60.1	75.6	58.9	236	212
5/2/94	13	36.6		46.1	23.2	41.1	51.8	60.2	73.4	79.2	93.9	68.3	277	377
5/3/94	14			44.4		43.4	57.8	65.1	82.2	80.5	104	68.5	328	389
5/5/94	16					48.4	87.1	70.3	87.1	87.1	104	148	427	344
5/6/94	17					51.3	69.5	72.6	88.2	78.1	107	356	420	284
5/9/94	20					52.6	72.1	78.0	78.6	72.7	82.9	190	219	93.7
5/11/94	22					51.9	76.7	80.1	64.2	60.9	64.2	71.7	92.3	69.5
5/13/94	24					53.9	66.5	73.8	55.0	55.5	58.6	56.9	60.3	60.3
5/17/94	28					47.5	53.8	52.0	47.1	47.5	51.6	53.3	58.1	51.6
5/25/94	36					47.6	69.4	55.8	56.3	57.5	62.5	63.3	61.7	50.7

<i>Date</i>	<i>Days</i>	<i>MLS</i>												
		<i>13-10</i>	<i>13-11</i>	<i>14-1</i>	<i>14-2</i>	<i>14-3</i>	<i>14-4</i>	<i>14-5</i>	<i>14-6</i>	<i>14-7</i>	<i>14-8</i>	<i>14-9</i>	<i>14-10</i>	<i>14-11</i>
4/19/94	0	15.9	26.1	44.1	55.1	55.1	54.6	55.5	57.5	50.6	65.8	57.5	20.2	22.7
4/28/94	9	23.5	42.0											
4/29/94	10	58.2	35.3											
5/2/94	13	174	46.1			57.7	56.4	56.9	56.9	58.9	150	64.6	17.8	23.4
5/3/94	14	212	61.8	53.7	57.3	57.3	56.1	58.5	61.8	79.8	312	187	21.7	20.0
5/4/94	15				58.5	57.2	56.0	58.5	63.1	78.8	348	336	39.8	18.1
5/5/94	16	206	37.7	55.5	59.2	59.2	58.0	59.2	63.2	83.5	370	397	104	19.4
5/6/94	17	178	36.7	58.4	60.2	61.8	60.2	59.7	61.8	80.2	372	402	136	26.2
5/9/94	20	82.2	54.3	61.1	60.5	62.7	63.3	60.5	65.5	73.7	382	289	100	37.1
5/11/94	22	38.6	34.6	56.2	56.8	58.8	58.8	58.3	60.1	100	302	124	13.7	27.7
5/13/94	24	14.0	10.7	56.9	57.9	59.6	60.3	60.3	65.6	163	116	68.4	9.60	29.1
5/17/94	28	14.5	16.9	56.1	56.6	58.6	60.6	57.3	59.8	61.6	67.0	40.9	7.44	13.7
5/25/94	36	17.2	27.6	56.7	61.7	65.2	87.4	72.4	78.1	72.4	63.3	51.3	8.89	15.5

Table B10 (Concluded)

<i>Date</i>	<i>Days</i>	<i>MLS</i>												
		<i>3-4</i>	<i>3-5</i>	<i>3-6</i>	<i>3-7</i>	<i>4-4</i>	<i>4-5</i>	<i>4-6</i>	<i>4-7</i>	<i>5-3</i>	<i>5-4</i>	<i>5-5</i>	<i>5-6</i>	<i>5-7</i>
5/4/94	15					38.2	51.0	13.8	18.5					
5/5/94	16	52.1	52.7	11.6	18.2									
5/6/94	17					37.1	47.7	13.6	17.4					
5/9/94	20	46.6	57.5	10.6		48.2	49.7	13.7	20.5					
5/11/94	22	43.2	56.2	9.77	16.0	36.5	45.5	13.7	15.8					
5/13/94	24	44.6	62.4	10.1	17.3	39.6	47.5	13.9	15.7					
5/17/94	28					42.1	45.9	12.5	14.9					
5/20/94	31					41.8	44.5	11.3	14.8	5.03	49.9	373	17.4	9.63
5/23/94	34					54.1	51.2	10.5	16.7	6.79	62.4	443	22.9	12.4
5/25/94	36									6.22	59.7	308	20.6	14.0
5/27/94	38									6.73	57.1	260	19.5	11.0
5/31/94	42									6.29	51.4	82.4	19.4	10.6
6/2/94	44									5.86	71.5	67.5	19.0	8.76
6/9/94	51									8.71	51.3	50.2	18.9	10.8

<i>Date</i>	<i>Days</i>	<i>MLS</i>										
		<i>15-1</i>	<i>15-2</i>	<i>15-3</i>	<i>15-4</i>	<i>15-5</i>	<i>15-6</i>	<i>15-7</i>	<i>15-8</i>	<i>15-9</i>	<i>15-10</i>	<i>15-11</i>
5/20/94	31	49.5	38.5	49.9	53.3	49.9	47.4	53.3	51.6	5.24	9.97	14.3

<i>Date</i>	<i>Days</i>	<i>MLS</i>										
		<i>17-1</i>	<i>17-2</i>	<i>17-3</i>	<i>17-4</i>	<i>17-5</i>	<i>17-6</i>	<i>17-7</i>	<i>17-8</i>	<i>17-9</i>	<i>17-10</i>	<i>17-11</i>
5/23/94	34	53.4	54.1	66.0	54.5	54.5	56.9	51.2	44.1	33.1	9.94	16.3
5/25/94	36	58.6	56.7	68.0	61.7	54.4	59.2	74.8	64.4	40.6	9.19	9.66
5/27/94	38	55.0	51.2	61.6	42.2	43.6	49.5	49.1	52.1	33.2	8.40	17.5

Note: Results are in mg/L. Blank fields indicate well or port not sampled.

**Table B11. Bromide Data from the Forced-Gradient Tracer Test
Started September 12, 1994**

<i>Date</i>	<i>Time</i>	<i>Hours</i>	<i>Days</i>	<i>Well</i>		<i>MLS</i>								
				<i>I-3</i>		<i>12-1</i>	<i>12-2</i>	<i>12-3</i>	<i>12-4</i>	<i>12-5</i>	<i>12-6</i>	<i>12-7</i>	<i>12-8</i>	<i>12-9</i>
9/12/94	3:15 PM	0.0	0.0	272	*	*	*	*	*	*	*	*	*	*
9/12/94	5:10 PM	1.9	0.1	301	*	*	*	*	*	*	*	*	*	*
9/12/94	9:05 PM	5.8	0.2	295	*	*	*	*	*	*	*	*	*	*
9/13/94	3:30 AM	12.3	0.5	263	*	*	*	*	*	*	*	*	*	*
9/13/94	8:00 AM	16.8	0.7	239	*	*	*	*	*	*	*	*	*	*
9/13/94	1:45 PM	22.5	0.9	34.3	*	*	*	*	*	*	1.60	*	*	*
9/13/94	5:15 PM	26.0	1.1	10.2	*	0.64	0.88	0.44	0.92	0.38	44.3	2.93	0.29	
9/13/94	9:30 PM	30.3	1.3	3.94	2.46	7.03	6.84	1.94	4.10	6.12	221	31.2	3.33	
9/14/94	5:18 AM	38.1	1.6	0.96	11.7	29.9	22.8	7.29	12.9	63.1	286	254	70.3	
9/14/94	9:22 AM	42.1	1.8	0.53	16.3	34.1	24.0	9.10	14.4	149	257	240	143	
9/14/94	12:47 PM	45.5	1.9	0.55	20.7	41.5	29.9	10.8	23.1	214	244	230	197	
9/14/94	4:48 PM	49.6	2.1	0.70	22.6	50.0	33.8	13.2	52.1	246	259	236	199	
9/14/94	9:16 PM	54.0	2.3	*	37.3	75.7	51.3	19.9	124	276	287	271	248	
9/15/94	6:27 AM	63.2	2.6		69.9	89.6	52.4	19.9	155	274	285	234	145	
9/15/94	1:36 PM	70.4	2.9		36.5	64.0	40.7	12.9	61.5	215	247	13.1	11.8	
9/16/94	12:27 PM	93.2	3.9		2.15	4.01	3.31	*	1.06	3.01	5.36	7.95	10.9	
9/20/94	9:43 AM	186.5	7.8		*	0.67	0.67	*	*	*	0.49	0.94	1.58	

<i>Date</i>	<i>Time</i>	<i>Hours</i>	<i>Days</i>	<i>MLS</i>												
				<i>12-10</i>	<i>12-11</i>	<i>13-1</i>	<i>13-2</i>	<i>13-3</i>	<i>13-4</i>	<i>13-5</i>	<i>13-6</i>	<i>13-7</i>	<i>13-8</i>	<i>13-9</i>		
9/12/94	3:15 PM	0.0	0.0	*	*											
9/12/94	5:10 PM	1.9	0.1	*	*											
9/12/94	9:05 PM	5.8	0.2	*	*											
9/13/94	3:30 AM	12.3	0.5	*	*											
9/13/94	8:00 AM	16.8	0.7	*	*											
9/13/94	1:45 PM	22.5	0.9	*	*											
9/13/94	5:15 PM	26.0	1.1	*	*											
9/13/94	9:30 PM	30.3	1.3	*	*											
9/14/94	5:18 AM	38.1	1.6	*	*											
9/14/94	9:22 AM	42.1	1.8	0.94	*											
9/14/94	12:47 PM	45.5	1.9	10.5	*						121	217	247	173		
9/14/94	4:48 PM	49.6	2.1	22.4	*	96.6	88.4	33.5	60.4	26.3	186	249	249	201		
9/14/94	9:16 PM	54.0	2.3	27.0	*											
9/15/94	6:27 AM	63.2	2.6	10.1	*	49.8	61.5	16.1	15.0	2.98	146	131	0.78	2.46		
9/15/94	1:36 PM	70.4	2.9	*	*	86.4	16.9	5.20	7.17	0.92	130	15.5	*	*		
9/16/94	12:27 PM	93.2	3.9	10.0	6.99	0.99	0.56	*	0.60	0.40	0.73	52.5	0.69	18.9		
9/20/94	9:43 AM	186.5	7.8	0.91	1.63	*	*	*	*	*	*	*	1.30	*		
9/23/94	10:00 AM	258.8	10.8			*	*	*	*	*	*	*	*	*		

Table B11 (Concluded)

Date	Time	Hours	Days	MLS												
				13-10	13-11	17-1	17-2	17-3	17-4	17-5	17-6	17-7	17-8	17-9		
9/14/94	4:48 PM	49.6	2.1	*	*											
9/14/94	9:16 PM	54.1	2.3													
9/15/94	6:27 AM	63.3	2.6	*	*											
9/15/94	1:36 PM	70.4	2.9	*	*											
9/16/94	12:27 PM	93.3	3.9	1.4	2.29											
9/20/94	9:43 AM	186.5	7.8	*	*											
9/23/94	10:00 AM	258.8	10.8	*	*											
9/27/94	9:35 AM	354.4	14.8			*	*	*	*	*	*	130	3.89	17.8		
9/30/94	9:43 AM	426.5	17.8			*	*	*	*	*	172	146	11.6	0.83		
10/4/94	9:43 AM	522.5	21.8			*	*	0.42	2.42	103	254	138	2.23	2.15		
10/7/94	9:43 AM	594.5	24.8			*	0.36	0.75	13.0	75.3	127	6.68	1.32	0.44		
10/11/94	9:43 AM	690.5	28.8			0.49	0.35	0.87	24.5	162	19.7	2.92	1.14	0.62		
10/14/94	9:43 AM	762.5	31.8			0.44	*	0.84	2.97	66.0	11.3	3.27	1.65	0.86		
10/18/94	9:43 AM	858.5	35.8			0.40	0.49	*	0.40	*	*	2.59	0.47	0.69		

Date	Time	Hours	Days	MLS												
				17-10	17-11	3-3	3-4	3-5	3-6	4-3	4-4	4-5	4-6			
9/20/94	9:43 AM	186.5	7.8			*	1.15	55.3	0.53							
9/23/94	10:00 AM	258.8	10.8			*	*	109	3.02	*	*	*	*			
9/27/94	9:35 AM	354.4	14.8	*	*	*	*	232	*	*	*	43.8	1.16			
9/30/94	9:43 AM	426.5	17.8	*	*	*	0.35	5.90	*	*	0.33	182	0.36			
10/4/94	9:43 AM	522.5	21.8	1.30	*	*	*	2.06	*	0.35	1.11	229	0.90			
10/7/94	9:43 AM	594.5	24.8	*	*	*	*	1.98	*	*	0.83	13.4	*			
10/11/94	9:43 AM	690.5	28.8	*	*	*	0.32	0.24	*	*	0.72	32.5	*			
10/14/94	9:43 AM	762.5	31.8	*	*					*	0.77	17.8	*			
10/18/94	9:43 AM	858.5	35.8	*	*					*	0.44	*	*			

Notes: Results are in mg/L. Blank fields indicate well or port not sampled.

* Below detection.

Appendix C.
Graphs of Water-Chemistry Data

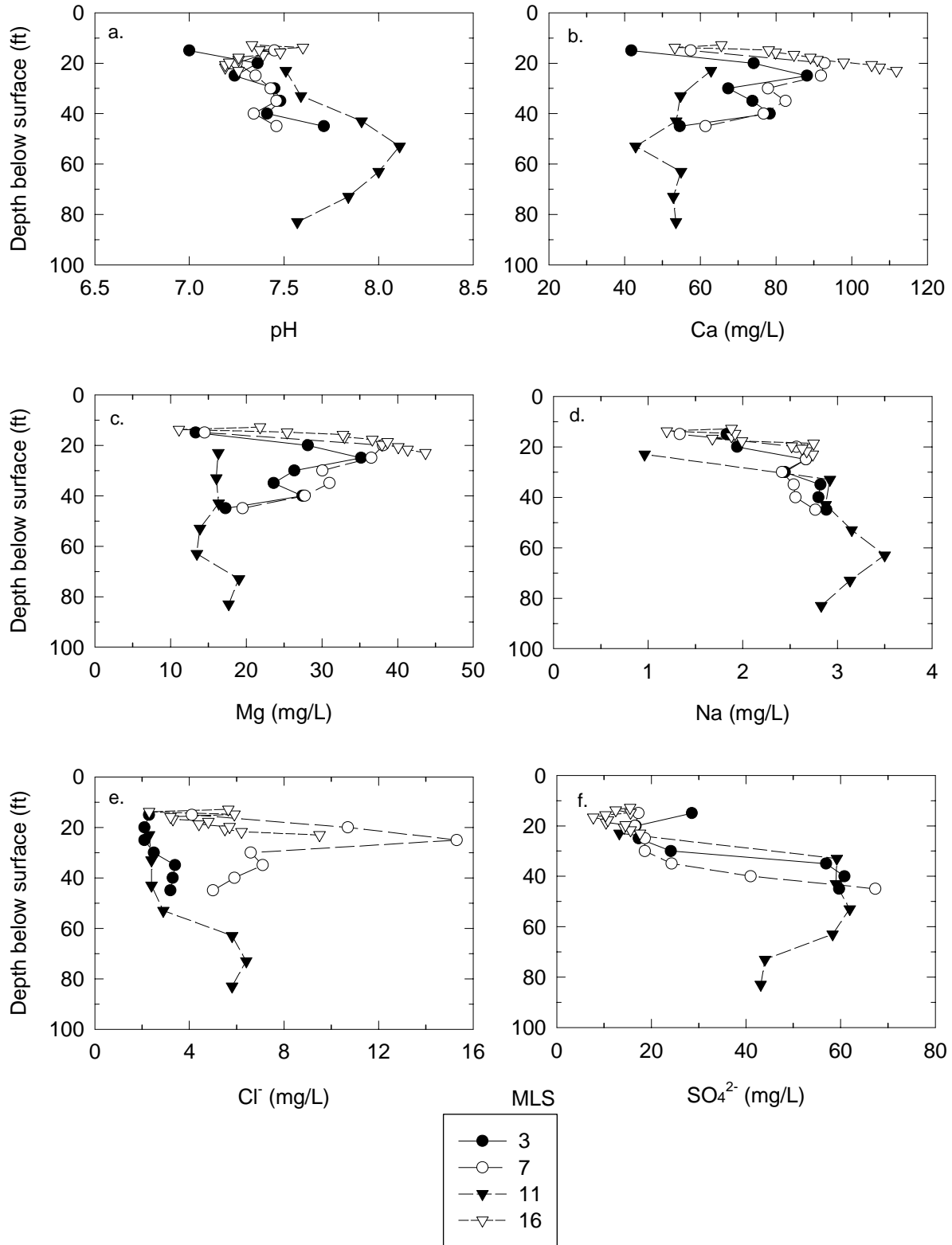


Figure C1. Concentrations of parameters vs. depth for MLS-3, MLS-7, MLS-11, and MLS-16, July 1994: a. pH; b. calcium; c. magnesium; d. sodium; e. chloride; f. sulfate.

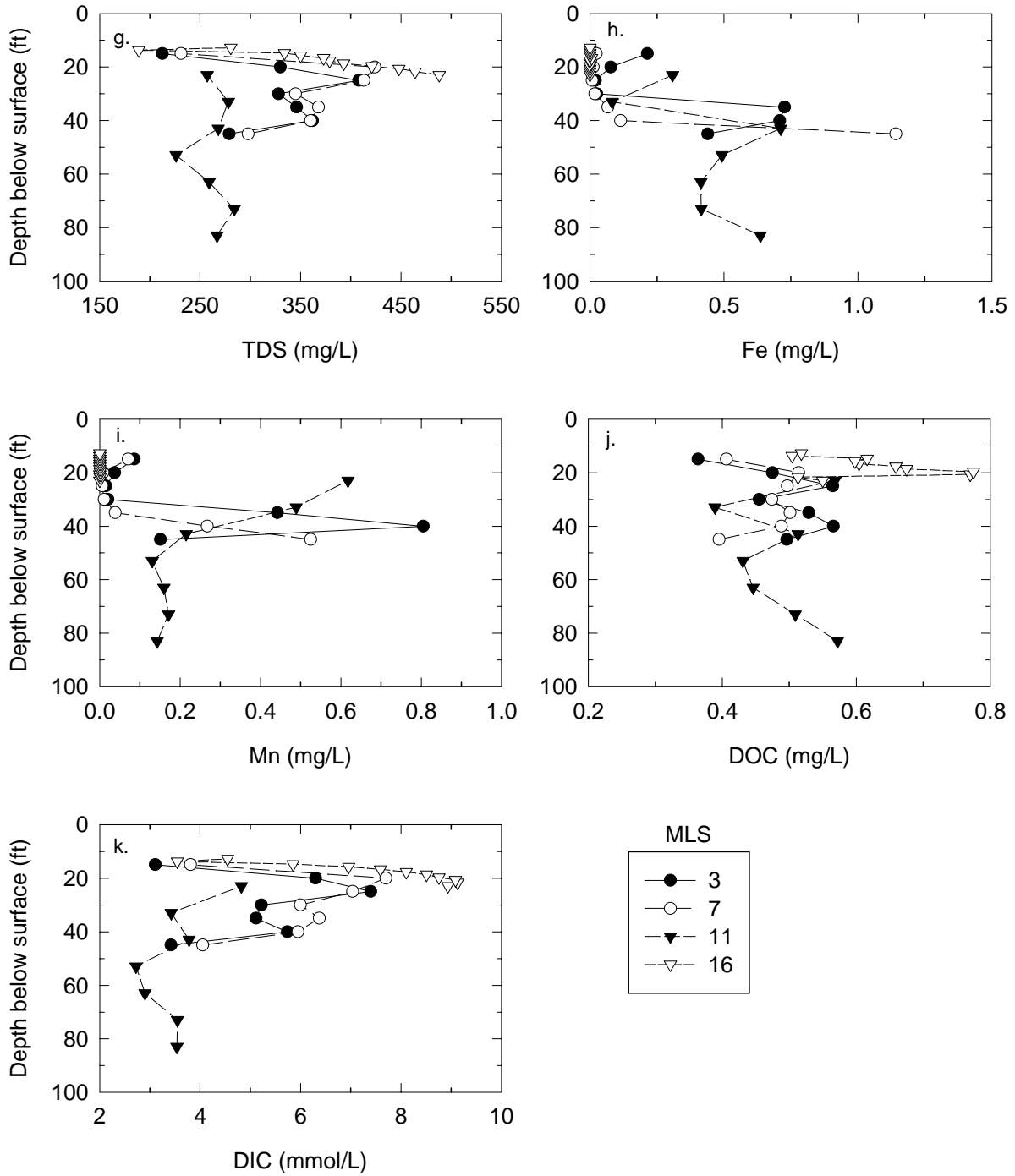


Figure C1. (concluded) Concentrations of parameters vs. depth for MLS-3, MLS-7, MLS-11, and MLS-16, July 1994: g. total dissolved solids; h. iron; i. manganese; j. dissolved organic carbon; k. dissolved inorganic carbon.

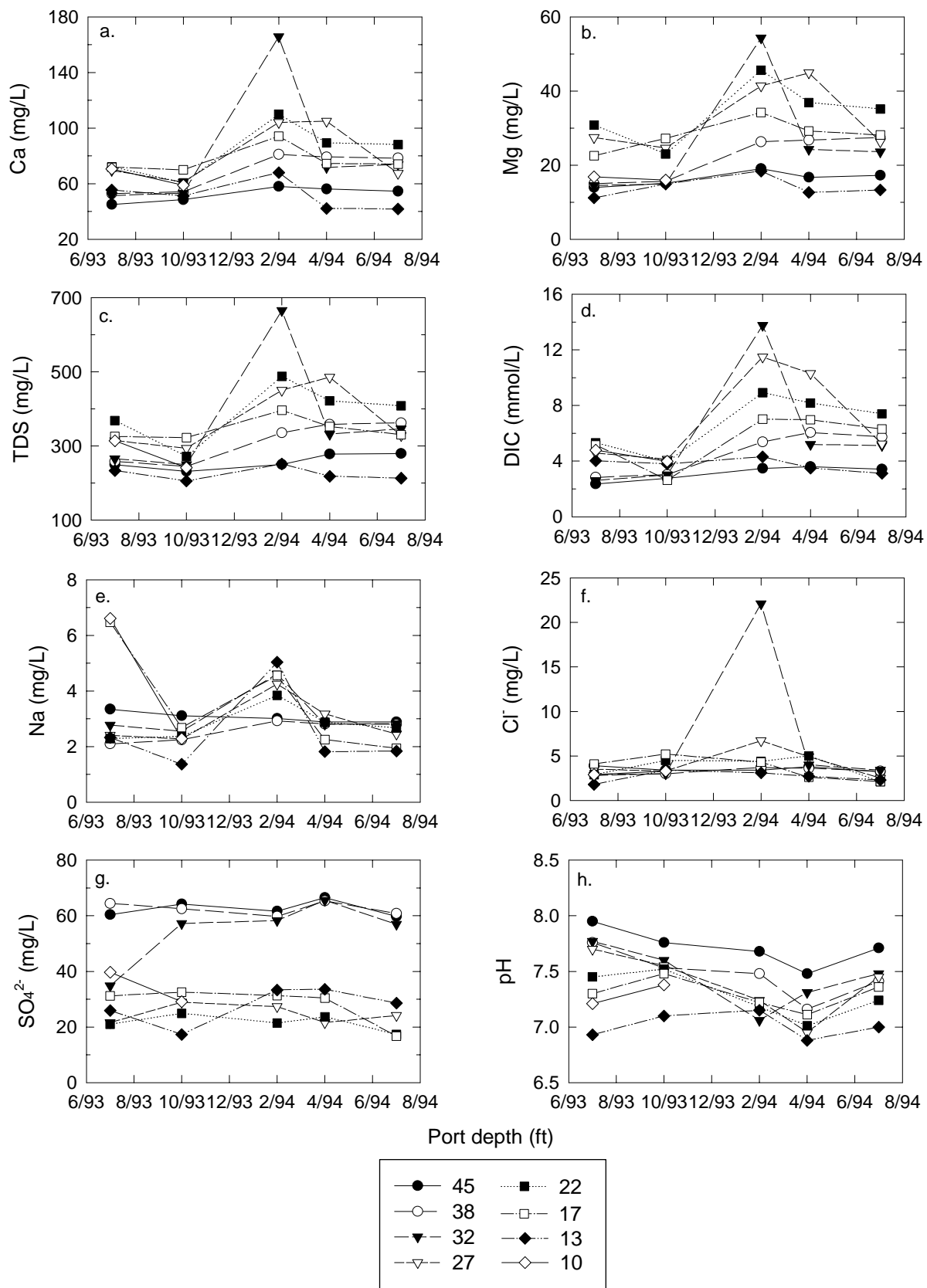


Figure C2. Temporal variation in concentrations of various parameters for all sampling ports of MLS-3: a. calcium; b. magnesium; c. total dissolved solids; d. dissolved inorganic carbon; e. sodium; f. chloride; g. sulfate; h. pH.

