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This constitutes the final report for a WRRC project directed by Hans-Olaf Pfannkuch and Howard D. Mooers of the Department of Geology and Geophysics, University of Minnesota entitled: Groundwater mapping and the determination of the optimum level of information for the assessment of groundwater sensitivity to contamination in complex glacial-drift aquifers. This project was funded for the period July 1989 to June 1991.

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# ABSTRACT

Uncertainties in the hydrogeology of a study area and in the transport properties of potential contaminants challenge the designers of groundwater monitoring well networks. Numerical flow modeling is a useful tool for guiding the placement of wells, but it requires justified boundary conditions and sufficient knowledge of aquifer parameters. In a graciated terrain, additional problems arise because of complex spatial arrangements of aquifers and aquitards. The transport of conservation contaminant tracers is normally calculated as a simple function of average values of hydraulic conductivity, hydraulic gradient, and effective porosity. But are the results of this straightforward method always valid?

In order to address these topics, this investigation focused on a landfill on the Anoka sandplain of east-central Minnesota. The purpose of this study was to determine the proper placement of an initial group of monitoring wells at the landfill using only offsite public-domain data. The results may then be applied to the siting of other wells.

Finite-difference flow modeling was supported by an abundance of inexpensive public-domain information and by the construction of a detailed, sub-regional glacial geologic map. A two-dimensional kriging analysis refined the model by determining the cell-by-cell best estimates of the basal elevation of the surficial aquifer. Particle tracking results indicated the expected pathway of landfill leachate. Based on the results, one well upgradient of the landfill and several downgradient wells were selected from the database of actual monitoring wells, and the head data from these shallow wells were used to calibrate the model. The calibrated hydraulic conductivity of the sandplain aquifer agrees closely with values obtained through grain-size analyses and pump tests. Numerical analyses of boundary conditions support the validity of the flow model.

Other case studies of unconfined outwash aquifers suggest that predicted plumes of conservative tracers are often greater than the actual extents. Compared to the chloride data for monitoring wells at the Anoka site, particle tracking results have an accurate orientation but a length at least two times too long. Uncertainties, such as the effect of longitudinal dispersion and the transient nature of the leachate's initial concentration and source area, suggest an even greater difference. This conceptual understanding of plume migration provides guidance for the placement of additional downgradient wells. The described application of models and inexpensive offsite data to monitoring well network design is a methodology that may be effective for the monitoring of solutes from existing or proposed potential contamination sources.



## ACKNOWLEDGEMENTS

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# INTRODUCTION

### *Objective and approach*

A groundwater contamination investigation involves a large financial expenditure, much of which is spent in the installation of monitoring wells. By first addressing a site using available data and techniques, monitoring wells may be positioned at locations that optimize sampling objectives, namely the direction and extent of potential contaminant migration.

The purpose of this study is to develop a method for determining the placement of monitoring wells at an existing or potential source of groundwater contamination. The approach makes use of offsite public-domain information in a hydrogeologic investigation utilizing groundwater modeling and geostatistical techniques to address the migration of conservative, advectively transported contaminants. An extensive monitoring well network at a landfill near Anoka, Minnesota serves only as an unbiased means of calibrating the flow model and checking the dependability of the results. The availability of actual onsite data is conducive to a practical case study with an approach relying on a thorough hydrogeologic analysis rather than a hypothetical scenario. This study may serve as a means of guiding monitoring well placement at other sites, especially those in similar geologic settings.

### *The Anoka Regional Sanitary Landfill*

The Anoka Regional Sanitary Landfill (Anoka landfill) is located in the city of Ramsey in southwestern Anoka County, Minnesota (Figure 1), approximately 20 miles (32 km) to the north of Minneapolis. Housing developments are north and west of the landfill (Figure 2). Another subdivision, Hunters Hills, is one-half mile (800 m) to the southeast.

The landfill history is described in detail by Hickok and Foth & Van Dyke (1987). In late 1967 the city of Anoka began the landfilling operations in an abandoned sand and gravel pit. The 65 acre (27 hectare) waste disposal area has no liner or leachate collection system. The elevation of the base of the waste is approximately 870 ft (265 m). At the time of the 1993 closure, the maximum elevation including the cap will be 1,024 ft (312.1 m) (S. Kollodge, 1989, personal communication), which is about 150 ft (45 m) above grade.

The predominant waste type disposed of in the Anoka landfill is mixed municipal wastes with smaller amounts of foundry sands, fly ash, and demolition waste. During the years preceding its 1972 State Solid Waste Permit, however, the Anoka landfill also received hazardous wastes in the form of paint sludges and paint thinners. In addition, the landfill accepted liquid sludge from the city of Anoka wastewater treatment plant during this period. No liquid waste, materials in drums, or materials currently classified as hazardous are believed to have been disposed of at the landfill since the 1972 permit; only municipal and other non-hazardous wastes have been accepted.

In recent years a wealth of data has been compiled on site conditions at the Anoka landfill, including well logs, well construction details, aquifer pump tests, and temporal analyses of heads and water quality parameters. A groundwater monitoring network was begun in 1972 to evaluate the potential for groundwater contamination. As of 1987 the database included:

- 77 monitoring wells (17 single monitoring well locations and 22 locations of nests of 2 to 5 wells),
- 23 piezometers,
- 6 soil borings, and
- 2 aquifer pump-test wells

arranged around the site, with the majority downgradient of the waste disposal area. In 1990 the network had grown to include two more aquifer test wells and ten more associated piezometers (Foth & Van Dyke, 1989a), and two additional bedrock wells (Foth & Van Dyke, 1990).

Approximately quarterly sampling of wells began in the 1970s, providing a large database of spatial and temporal changes in heads and water quality parameters, including general parameters, major ions, heavy metals, and volatile organic compounds.

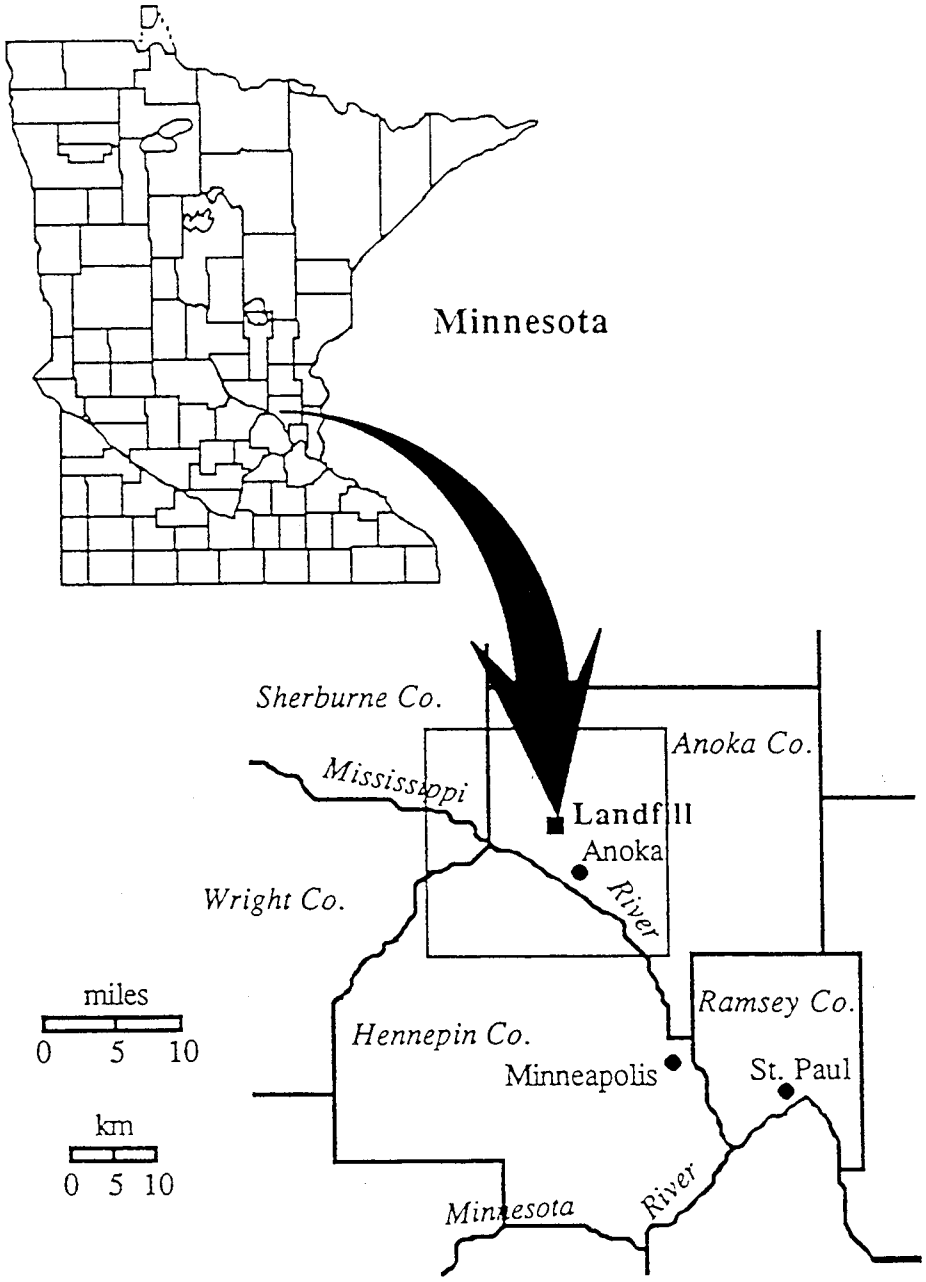



Figure 1 - Location of landfill. The square represents the border of the 300 $mi^2$ (780 $km^2$) glacial geology mapping area of Quinn (1992).

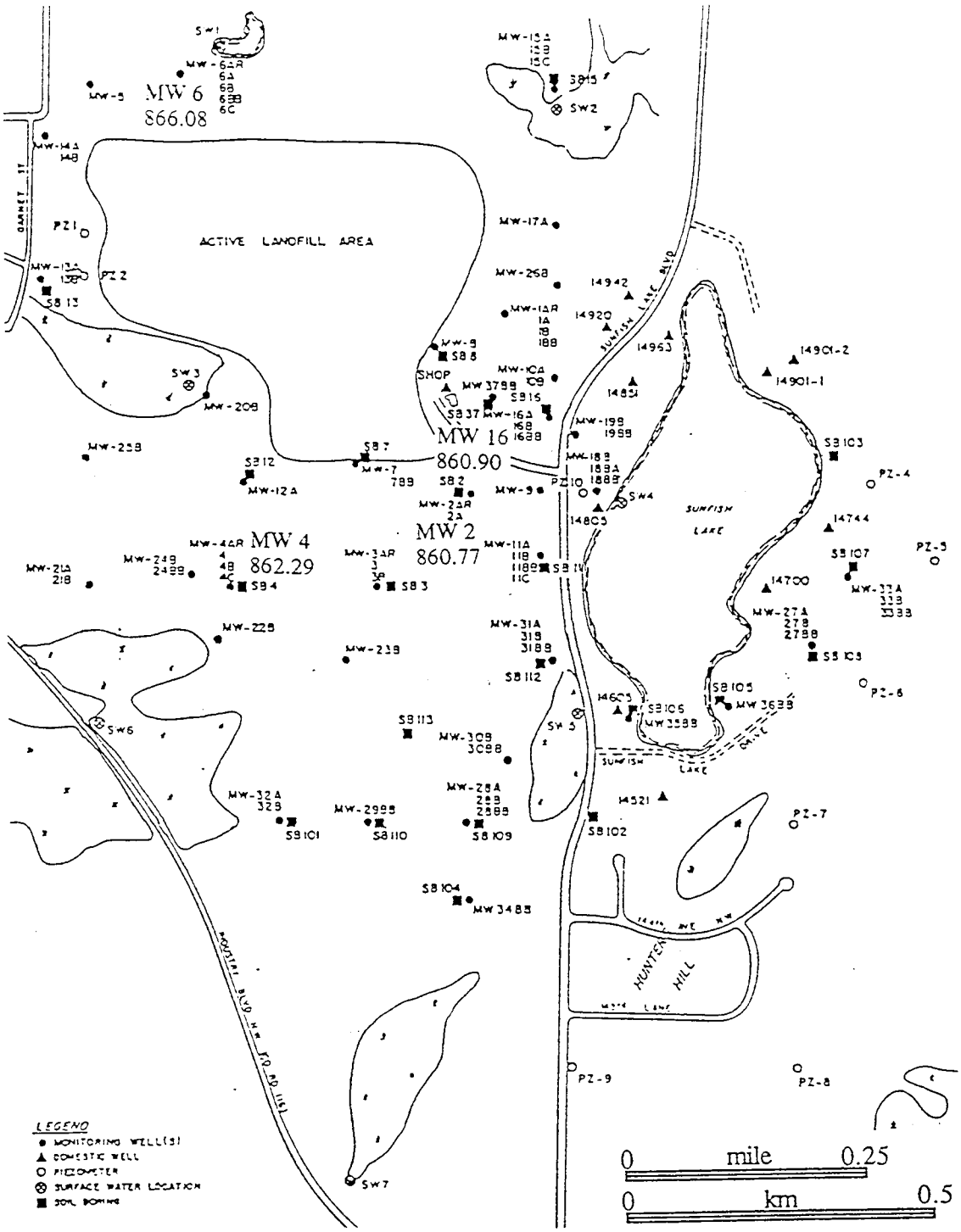



Figure 2 - The Anoka landfill's network of monitoring wells, piezometers, and soil borings, and the four water-table monitoring wells used in calibration of the flow model (modified from Hickok and Foth & Van Dyke, 1987).

### *Monitoring guidelines*

Federal guidelines dealing with groundwater monitoring are generally ambiguous and open to a great deal of interpretation. In the case of a waste disposal site, the monitoring scheme is to be comprised of four monitoring wells: one located upgradient from the site and three in the direction of groundwater flow (NWWA and U.S. EPA, 1986; Aller and others, 1989). These are requirements for an initial network only; once contamination is detected the groundwater monitoring effort is augmented to delineate the horizontal and vertical extents of the plume and to determine the rate of contaminant migration (Aller and others, 1989; U.S. EPA, 1986). The growth of the network is normally done in a step-wise fashion, so that information obtained from sample locations may guide the placement of future sampling. The guidelines, however, cannot supply specific information dealing with, for example, the proximity of wells to a possible source of pollution or the depth at which the wells should be screened. Of primary concern is assessing the impact to the uppermost aquifer (NWWA & U.S. EPA, 1986). But in a case of complex hydrogeology, pathways to deeper aquifers may exist, and they may be difficult to determine.

Individual states may choose to adopt regulations more stringent than those of the U.S. EPA (Aller and others, 1989). Normally the monitoring well network design necessary to define the hydrogeology of a site is ultimately determined by a regulatory agency (Bagchi, 1990).

The Minnesota Rules (State of Minnesota, 1989, chapter 7035.2815) provide general requirements for the monitoring of mixed municipal solid waste disposal facilities. These guidelines allow a subjective approach to designing a groundwater monitoring network. The rules state the basic purpose of monitoring wells: the early detection of the pollutants from a facility and the determination of the spatial extent of pollutant concentrations. In order to comply, the number, placement, types and depths of monitoring points around a site must be chosen according to an evaluation of the hydrogeologic setting, which includes an assessment of potential contaminant pathways. Additional monitoring points need to be constructed in the event of the detection of contaminants.

Because of the inherent variability of geologic deposits, interpretations of these guidelines may vary widely among investigators and result in different monitoring schemes. Case studies of monitored sites may provide the best resource for future network designs at other facilities in similar hydrogeologic environments.

### *Design of monitoring well networks*

An area of growing emphasis in the literature focuses on the various approaches of monitoring well networks. Specifically, many researchers are striving to develop a method for the optimization of well placement. Studies of this sort are often theoretical in nature.

Meyer and Brill (1988) proposed a computationally intensive technique for monitoring well network design that relies on an optimization model to choose locations that maximize the probability of plume detection. Loaiciga (1989) developed a method of optimization based on mass transport modeling, statistical accuracy, and economic constraints. Massmann and Freeze (1987a, 1987b) included groundwater monitoring aspects in an elaborate risk-cost-benefit analysis of potential pollution sources. Freeze and others (1990) and Massmann and others (1991) provide a thorough introduction to the importance of geology and engineering in decision analysis of hydrogeological studies using a risk-cost-benefit function. This approach may be applied to well network design.

Pfannkuch (1975) addressed the problem of determining the optimal amount of information needed for assessing hydrogeologic uncertainty. He proposed that the worth of additional data could only be quantified if the benefits of the extra effort could be economically calculated. Pfannkuch and Labno (1976) detailed a step-wise approach to optimum well placement that stresses the various objectives and constraints associated with the design strategy.

Other efforts to determine optimal monitoring well placement involve some form of spatial analysis such as kriging. Spruill and Candela (1990) demonstrated the use of kriging to gain a maximum of knowledge from a minimum of monitoring wells in a regional water quality network. They compared the data needs for this objective with the greater amount of information needed in the case of determining a mean or median parameter concentration value for the region.

McLaughlin and Graham (1986), in a cokriging analysis, showed how the spatial correlation and cross-correlation of hydraulic conductivity, head, and chemical concentration may aid in monitoring well network design.

***Leachate and tracer plumes in surficial sandy aquifers***

Several recent studies investigate the characteristics of contaminant plumes in unconfined glaciofluvial aquifers.

Kimmel and Braids (1980, 1974) examined the impact of two landfills in Long Island, New York. Extensive monitoring well networks allowed them to observe the three-dimensional nature of the plumes. They found a downward component of leachate movement beneath the landfills, attributed to a density contrast between contaminated and background waters. Slugs of the high-density leachate are believed to flow downward through the aquifer after recharge events.

Chloride is a common component of landfill leachate. Because chloride is generally non-reactive, it serves as conservative tracer of contaminant movement (Davis and others, 1985). A conservative tracer would be expected to travel at the same rate as the average linear groundwater velocity, according to Darcy's law:

$$v = \frac{K \cdot i}{n_e} \qquad \text{Eq. 1}$$

where $v$ = average linear groundwater velocity [$LT^{-1}$],
$K$ = hydraulic conductivity [$LT^{-1}$],
$i$ = hydraulic gradient [unitless], and
$n_e$ = effective porosity [unitless].

A key finding of Kimmel and Braids (1980) is that a great difference between the expected chloride plume location based on the darcian approximation and the actual plume location supported by the field data. The theoretical extent of the plumes, equal to $v$ multiplied by the time since landfilling began, is 3 to 4 times longer than the actual extent. Even if leaching of contaminants did not occur until sometime after the opening of the landfills, this disagreement is difficult to rationalize.

MacFarlane and others (1983), using an elaborate, high-density monitoring well network, studied a landfill located on a glaciofluvial aquifer in Ontario. Their findings were similar to those of Kimmel and Braids (1980) in that the leachate plume has a downward component beneath the landfill and then travels more horizontally at depth in the aquifer. They attribute the downward movement to up to three possible factors: downward head gradients produced by a transient water-table mound beneath the landfill, greater density of leachate, and lower viscosity of leachate as a result of heat generated by microbial activity in the plume. At this site, a wide range of conductivities were estimated using grain-size analyses, permeameter tests, slug tests, and pump tests. Using the geometric mean hydraulic conductivity obtained with each method produced predicted conservative contaminant plumes of 0.5 to 1.5 times the extent of the actual chloride plume. The actual plume, however, exhibited pronounced longitudinal dispersion, with low chloride concentrations throughout the downgradient half of the plume. The theoretical extent of the plume therefore was much less than the extent predicted using most of the conductivity determination methods. The authors suggest two reasons for the disagreement of theoretical vs. actual plume length: small-scale heterogeneities in the aquifer, and errors in the various methods used to determine hydraulic conductivity values.

Robertson and others (1991) performed bromide traces in two sandy surficial aquifers and in one case found the groundwater velocity indicated by the tracer to be half of the velocity calculated using permeameter-estimated hydraulic conductivity and average effective porosity. In contrast to the landfill studies, this tracer experiment is free of uncertainties related to the source, such as the transient nature of contaminant release and the changing areal extent of landfilled waste.

In some cases the theoretical plume may match the actual plume. Mackay and others (1986) released chloride and bromide tracers, as well as several organic solutes, into an outwash aquifer. They actually found the conservative tracers to travel faster than the average linear

groundwater flow velocity predicted using conductivity values obtained through slug tests, grain size analyses, and and permeameter methods. They attribute the discrepancy to errors in the estimates of hydraulic conductivity, hydraulic gradient, and effective porosity.

Dispersion is a factor which could affect the distribution of conservative tracers. However, plumes in unconfined sand aquifers have been shown to have negligible transverse dispersion (Robertson and others, 1991; Kimmel and Braids, 1980, 1974; Mackay and others, 1986). MacFarlane and others (1983) attribute lateral spreading of the landfill's plume to slight seasonal changes in groundwater flow directions. Vertical dispersion was found to be negligible in all of the above studies. Because of the infiltration of recharge, clean groundwater is typically above a vertically narrow plume downgradient from a contamination source. The effect of longitudinal dispersion, however, may be very difficult to evaluate, especially in studies that do not involve a controlled tracer experiment.

## METHODS

### *Glacial geology mapping procedures*

Detailed reconnaissance-level glacial geologic mapping can be accomplished by a synthesis of information compiled from air photos, topographic maps, and soil surveys.

Low-altitude (1:20,000-scale) black-and-white air photos from the University of Minnesota's Borchert Map Library are a valuable mapping tool. Stereo-pairs are available from a wide range of years. In suburban areas, older series that were photographed prior to extensive development are most useful in discerning various glaciogenic features. Tunnel valleys, eskers, kettles, dunefields, and paleo-drainageways are distinguished effectively by working with the stereo-pairs in addition to topographic maps.

Soil surveys indicate the distribution of various soil types, from which the glacial parent materials may be inferred. The surveys serve as a cross-check of air-photo interpretations in the case of delineating the zones of alluvial soils or mucks found in former meltwater drainageways or in peaty kettles. They are especially useful in distinguishing the borders of outwash or till units.

Glaciogenic items of interest were delineated on 1:24,000-scale topographic quadrangles. This information was then transcribed onto a mylar overlay of a portion of a 1:100,000-scale USGS planimetric map.

### *Hydrogeological glacial facies modeling*

A facies model serves as a framework for interpreting and predicting the geology of a given sedimentary environment (Walker, 1984). Glacial geologic facies models are useful in determining large-scale trends and distributions of different glaciogenic units (Eyles and Miall, 1984; White, 1974). A glacial facies model is a generalized assemblage of sediments that are representative of a certain glaciological depositional environment (Eyles, 1983).

Recent studies have emphasized the hydrogeological importance of such models (Anderson, 1989, 1987a, 1987b; Fraser and Bleuer, 1987; Stephenson and others, 1988). By understanding the surficial and buried sedimentological units in terms of their glacial genesis, one develops the ability to predict the distribution of aquifers and aquitards. These studies are usually concerned with large-scale facies because of the difficulties in evaluating small-scale heterogeneities (Anderson, 1989).

Glacial process models have been proposed for terrains associated with portions of Minnesota (Mooers, 1990, 1988; Chernicoff, 1983, 1980). Models such as these may form the basis for initial hydrogeological facies models useful in assessments of particular areas.

### *Groundwater modeling*

*Finite difference modeling and MODFLOW*

A finite difference groundwater model is a numerical approximation of a flow system. Conceptually, this technique is simple. The area of interest is chosen and discretized by a finite difference grid. Boundary conditions are assigned to the model according to available or inferred hydrologic features. The choice of the type of boundary conditions and their locations may necessitate a change in the configuration of the modeled area. Flow equations are assigned to each cell in the model depending on aquifer type and stresses imposed on the system. The equations are solved by an iterative method. Discussions of equations and applications are found in Wang and Anderson (1982) and Anderson and Woessner (1992), respectively.

For a cell in a steady-state, two-dimensional model of a homogeneous, unconfined aquifer, flow is calculated by the Boussinesq equation:

$$\frac{\partial}{\partial x}\left(K_x h \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_y h \frac{\partial h}{\partial y}\right) = -R \qquad \text{Eq. 2}$$

where $h$ = saturated thickness (head - base of aquifer) [L]
$x,y$ = cartesian coordinates [L],
$K_{x,y}$ = hydraulic conductivity [$LT^{-1}$], and
$R$ = recharge [$LT^{-1}$].

In the finite difference method, these derivatives are replaced by differences, and the solution is found by iterating to a specified convergence criterion. The calculation of a cell's transmissivity incorporates the varying saturated thickness of an unconfined aquifer.

MODFLOW is a versatile finite difference model written by McDonald and Harbaugh (1988). The model handles a multitude of hydrogeologic and anthropogenic factors. MODFLOW was chosen for use in this study because it is well documented, well tested, and its output can be utilized in a particle tracking program. In addition, a finite difference model was desired because it can easily incorporate an input array of geostatistically determined values.

Recent hydrogeologic literature contains numerous examples of finite difference modeling, and MODFLOW is the code used in many of these papers.

*Advective transport particle tracking and MODPATH*

MODPATH (Pollock, 1989, 1988) is a semianalytical particle-tracking program for determining steady-state groundwater flow paths in block-centered finite difference models. One-, two-, or three-dimensional analyses may be performed.

Flow velocities are determined by the flow model, MODFLOW, for each face of each finite difference block. The particle-tracking method assumes that the velocity in each coordinate direction varies linearly between the velocities calculated at each face by the flow model. From this information, MODPATH derives an analytical expression for the flow velocity distribution of each cell. In a two-dimensional model, the path of any particle can be computed from the two-dimensional velocity field. The flow paths are calculated exactly by MODPATH based on the finite difference flow model output. The flow model, however, is a simplification of a system, and therefore the MODPATH results are only as accurate as the MODFLOW output. In a particle-tracking model, dispersion is ignored, and flow is by advective transport only. Particles may be tracked forward (for example, to estimate flow paths from a contaminant source) or backward (for example, to determine recharge areas).

***Geostatistics***

Geostatistics involves the application of special statistical tools to spatially correlated data. The basic premise of geostatistics is that samples that are close together will generally be more similar in some parameter of interest than samples that are far apart. This spatial correlation may be investigated in geostatistical studies of one, two, or three dimensions. Isaaks and Srivastava (1989), Knudsen (1988), and deMarsily (1986) offer introductions to the subject.

*Variograms*

The experimental variogram quantifies the spatial structure of a data set and is determined by

$$\gamma(h) = \frac{1}{2N_h} \sum_{i=1}^{N_h} [Z(x_i + h) - Z(x_i)]^2 \qquad \text{Eq. 3}$$

where $\gamma(h)$ = variogram value for separation distance h,
$h$ = separation distance between points (lag),
$N_h$ = number of pairs of separation distance h,
$Z(x_i)$ = value of sample at location x, and
$Z(x_i+h)$ = value of sample at location x+h.

Although the symbol $\gamma(h)$ is actually the semi-variogram, this study will use the common convention of simply referring to it as the variogram.

The variogram equation calculates the average squared difference for pairs of data. The graph of $\gamma(h)$ vs. h, using the data, is called the experimental variogram. Modeling involves choosing a variogram type (for example spherical, linear, exponential, etc.) and determining the parameters that produce the best fit of the experimental variogram. These parameters usually include the nugget effect ($C_0$), the range (a), and the sill (C). The nugget effect is the $\gamma(h)$ value when $h \approx 0$; it quantifies extremely short-range variability. Data are correlated up to a separation distance known as the range. Past the range, the data are uncorrelated and the variogram value is the ultimate sill, which is equal to the sill plus the nugget effect.

Typically, the maximum h allowed in a variogram calculation is equal to roughly one-third to one-half of the maximum extent of the data. Beyond this point the variogram may break down as a result of too few data pairs. Summaries of typical model variograms and their corresponding equations are presented in deMarsily (1986, p. 301-303), Knudsen (1988, p. 42-46), and Isaaks and Srivastava (1989, p. 372-375).

A data set may exhibit anisotropy with respect to its range; that is, the range may be different in different directions. Anisotropy is often not visually apparent in a data set and can be observed only in directional variograms. The variogram results can therefore discover and quantify geologic factors which are otherwise not visible. In a two-dimensional study, following the construction of a satisfactory omnidirectional variogram, a directional variogram analysis may be performed by specifying a search direction and an angular tolerance. For example, four directional variogram search directions may be oriented E-W, NE-SW, N-S, and NW-SE, each with an angular tolerance of at least 22.5°.

*Ordinary Kriging*

Kriging is a geostatistical tool for optimal interpolation of spatially correlated data. It is often described as a minimum-variance, linear, unbiased estimator. By weighting nearby samples according to the kriging algorithm when estimating the value of a parameter of a point or a block, the variance of the estimate is minimized. Ordinary kriging is the type of kriging most commonly performed in geological applications because it does not assume a particular value for the local mean; estimation therefore takes into account only the local samples.

The following equation relates the estimated value of a block with neighboring data:

$$Z^*(v) = \sum_{i=1}^{n} \lambda_i Z(x_i) \qquad \text{Eq. 4}$$

where $Z^*(v)$ = estimated value of block v,
$n$ = number of nearby samples,
$Z(x_i)$ = value of sample at $x_i$, and
$\lambda_i$ = kriging weight of sample i.

The kriging weights, $\lambda_i$, are found by the use of the Lagrange parameter. The kriging weights are used in the calculation of the kriging variance of each block, which represents the variance on the error of the estimation. This error is modeled by a normal distribution.

*VARIO and Geo-EAS*

VARIO is a variogram program developed by Dr. R.J. Barnes of the University of Minnesota Department of Civil and Mineral Engineering. Experimental variograms may be produced from one-, two-, or three-dimensional data, and anisotropy analyses may be performed. Most variogram programs use a lag increment to group classes of data when plotting the experimental variogram. VARIO, however, utilizes a smoothing parameter, which has units of length. The smoothing effect is produced by calculating a moving average of points within a specified window. Because the smoothing algorithm gives the user a large degree of control during refinement of the variogram, it produces more representative variograms than the lag increment method.

Geo-EAS (Geostatistical Environmental Assessment Software) is a geostatistical computer package developed for the U.S. EPA and documented by Englund and Sparks (1988). Geo-EAS handles experimental and model variograms, kriging, and several support functions. Data sets must be two-dimensional. Kriging options include choices between ordinary and simple kriging, and block and point kriging. Point kriging is the estimation of the value of a point based on the kriging of nearby samples. Block kriging involves the use of an array of points (2 by 2, for example) for each block in a grid, then kriging with these array values to determine the estimate of each block. Point and block kriging usually yield similar estimates; however, point kriging produces higher kriging standard deviations.

## STUDY AREA

### *Regional setting: glacial geology of east-central Minnesota*

*Introduction*

The surficial geology and landforms of east-central Minnesota are primarily the result of two Late Wisconsin ice advances: the Superior lobe and the Grantsburg sublobe (Wright, 1972b). Earlier Pleistocene advances had previously deposited drift throughout the region. Despite interglacial erosion and the reworking and redepositing of older deposits by the more recent ice lobes, some older drift presumably underlies the Late Wisconsin deposits throughout much of this region. However, due to a lack of deep exposures and only generalized sub-surface descriptions in well logs, characterization of the history, thickness, areal extent, and texture of the older deposits is difficult.

*Superior lobe*

The Superior lobe emanated from the Lake Superior basin and advanced to a position southwest of the Twin Cities area (Figure 3a) during its Late Wisconsin maximum, the St. Croix phase. The resulting St. Croix moraine generally has a breadth of about 6 miles (10 km) in central Minnesota (Wright, 1972a). The southwestern portion of the moraine has a northwest-southeast-trending segment bordered on much of its northeastern, inner side by the Mississippi River valley. Radiocarbon dating of lake sediments indicates that the Superior lobe reached its maximum extent prior to 20,500 ± 400 years before present (Wright and others, 1973), although Clayton and Moran (1982) suggest a somewhat younger date.

The Superior-lobe deposits reflect the bedrock geology along the lobe's path and are typically red in color. The till is normally sandy in texture. Characteristic lithologies from northeast Minnesota present in the drift include red sandstone, basalt, amygdaloidal basalt, and red felsite (Wright and others, 1973).

Tunnel valleys and eskers commonly occur together in the area glaciated by the Superior lobe. The tunnel valleys may have been eroded by geothermally or frictionally generated subglacial meltwater (Wright, 1973) or by supraglacial meltwater penetrating to the base of the ice via crevasses (Mooers, 1988, 1989). Eskers are often present within the tunnel valleys and were presumably deposited by waning meltwater flow. The eskers represent the position of the subglacial stream as the deforming ice constricted the flow. Eskers are much narrower than tunnel valleys, and two or more sub-parallel eskers may be present within a tunnel valley.

The tunnel valleys are normally straight or slightly curved and trend to the southwest. This direction is oblique to the regional slope, but it parallels the inferred ice gradient and therefore is in the direction of decreasing hydrostatic head (Wright,1973).

The tunnel valleys range in width from <590 ft (180 m) to >3,300 ft (1,000 m), with an average of about 980 ft (300 m) (Wright, 1973), and they are about 33 ft (10 m) deep. In length they vary greatly (Wright, 1973), although most segments are 6 to 12 mi (10 to 20 km) long (Mooers, 1989). The eskers are normally discontinuous features within or along the sides of the tunnel valleys, and their crests are usually equal in elevation to the adjoining till plain (Wright, 1973). Fairly continuous segments are generally less than about 2.5 mi (4 km) in length (Mooers, 1989).

*Grantsburg sublobe and Anoka sandplain*

The Des Moines lobe advanced from the Red River Valley of northwestern Minnesota down the Minnesota River Valley to the valley's bend at Mankato. From this point the ice flowed northeasterly into the Minneapolis Lowland as the Grantsburg sublobe (Figure 3b). After thickening sufficiently, the sublobe overtopped the St. Croix moraine along a length of 75 miles (120 km) from Minneapolis northwest to Albany (Wright and others, 1973). The glacier flowed in a northeasterly direction as far as Grantsburg, Wisconsin, over the area recently vacated by the Superior lobe. The Grantsburg sublobe reached its maximum position about 16,000 years ago (Wright and others, 1973) during its Pine City phase. Ice of the Des Moines lobe proper eventually thickened enough to pass a topographic barrier and flow south into Iowa, reaching its

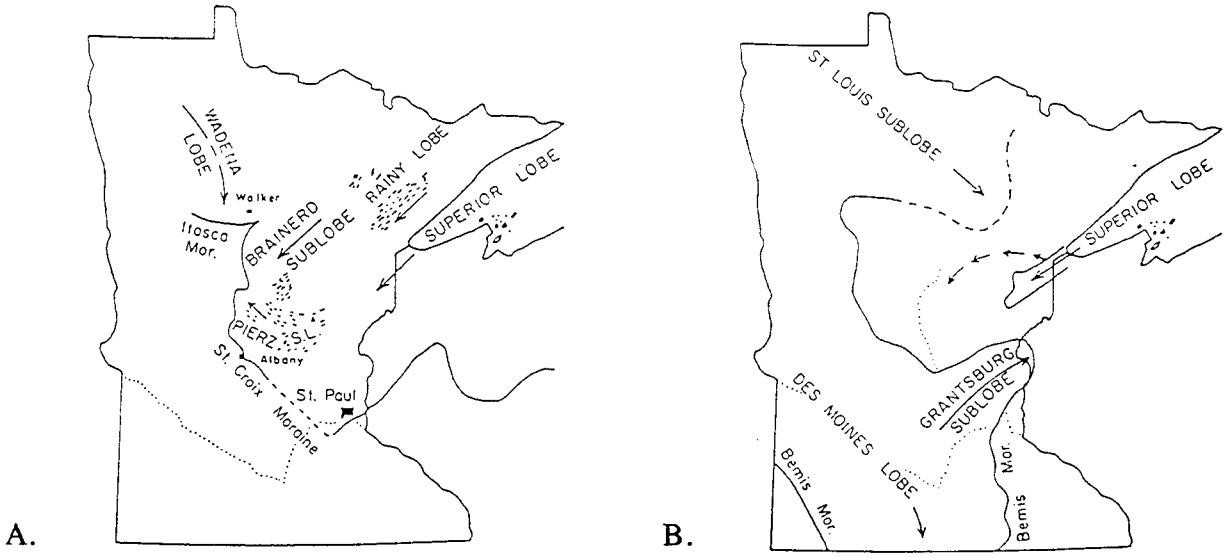



Figure 3 - Late Wisconsin glaciations affecting east-central Minnesota. A. Superior lobe and the St. Croix Moraine. B. Grantsburg sublobe (modified from Wright, 1972b).

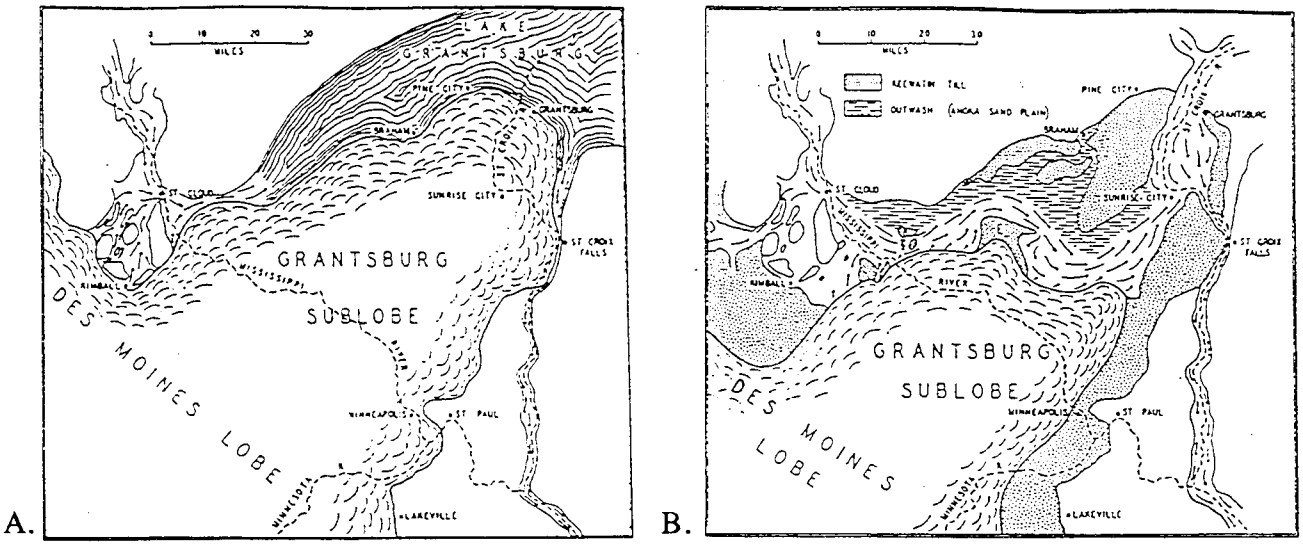



Figure 4 - Development of the Anoka sandplain. A. Maximum extent of Grantsburg sublobe. B. Flow around the wasting lobe as the sandplain evolves. Keewatin till is Grantsburg sublobe till, and "E" is Cooper's "Elk River Morainic Area" (from Cooper, 1935).

maximum extent at Des Moines 14,000 years ago (Wright and others, 1973; Clayton and Moran, 1982). Because the main lobe chose this new path, the flow of ice to the Grantsburg sublobe diminished. The sublobe stagnated, and the ice margin wasted back to the southwest as the glacial Mississippi River continuously shifted its course (Figure 4), distributing outwash over an area of 850 $mi^2$ (2,200 $km^2$) (Farnham, 1956). This outwash plain, together with the adjacent, slightly younger deposits of the Mississippi valley train, is known as the Anoka sandplain.

Unoxidized deposits of the Grantsburg sublobe are typically gray and calcareous. The till is generally silty and contains fragments of Cretaceous shale and lignite from western Minnesota and clasts of Paleozoic carbonate mainly from Manitoba (Wright and others, 1973). In the oxidized zone, however, the drift is light brown in color.

Tunnel valleys and eskers are not known to be associated with the Grantsburg sublobe (Chernicoff, 1980, 1983).

Numerous kettle lakes and other depressions abound in areas glaciated by the Superior and Grantsburg advances. The melting of buried pieces of ice produced these topographic lows, which may today be occupied by peatlands, swamps, or lakes. Stagnant Superior-lobe ice was present within the kettles and tunnel valleys during the advance of the Grantsburg sublobe. This dead ice preserved the morphology of these depressions during the deposition of Grantsburg till and/or the Anoka sandplain. Even the eskers within the tunnel valleys are easily discernable (Wright, 1972a, 1972b).

***Local study area***

The 23 $mi^2$ (59 $km^2$) local study area, for numerical modeling and geostatistical analysis, is bounded on the southwest by the Mississippi River, on the east by the Rum River, and on the north by Trott Brook (Figure 5). The western boundary is an inferred no-flow boundary which is discussed in the modeling section of this study.

*Hydrogeology of surficial sands*

The advances of the Grantsburg sublobe and the Superior lobe, in addition to pre-Late Wisconsin glaciers, have produced a complex assemblage of deposits of widely varying hydraulic conductivities. However, the surface of the entire local study area is covered with sands of the Mississippi valley train and the Anoka sandplain.

A detailed study by Quinn (1992) of an area covered by six 7.5-minute U.S. Geological Survey topographic quadrangles (Anoka, Nowthen, Elk River, Rogers, Coon Rapids, and Cedar) produced a map of the surficial geology of a 300 $mi^2$ (780 $km^2$) region centered around the Anoka landfill (Figure 1). The purpose of the mapping exercise was to closely examine the area's glacial geology in order to better understand the sedimentary associations closer to the landfill. Figure 6 shows a portion of this map representing approximately 100 $mi^2$ (260 $km^2$). The Anoka sandplain aquifer, comprised of Mississippi valley train sands and the Anoka sandplain proper, covers the entire modeling area.

Several shallow sandpits are located about 1 mile (1.6 km) south of the Anoka landfill (Quinn, 1992). These sandpits are the only known geologic exposures near the landfill. The sediments range from poorly sorted coarse sands and fine gravels to moderately sorted medium sand. Samples from texturally different crossbeds of the exposed Mississippi valley-train sands provide a means of estimating the hydraulic conductivity of the medium by two empirical techniques: the Hazen method (Holtz and Kovacs, 1981) and the Krumbein and Monk (1943) method. Porosities of the samples were also determined in the laboratory. Data and graphs are provided by Quinn (1992). A comparison of the hydraulic conductivity results shows quite close agreement between the values determined by the two methods. The conductivities obtained through these methods range between approximately $3.9 \times 10^{-4}$ to $2.4 \times 10^{-3}$ ft/s ($1.2 \times 10^{-2}$ to $7.3 \times 10^{-2}$ cm/s). Although these values do not provide an exact representation of the sandplain's range of permeability, they do support an average initial value of ~$1.2 \times 10^{-3}$ ft/s (~$3.5 \times 10^{-2}$ cm/s) for the modeling of the surficial sandy aquifer. This K value is typical for outwash sands (*e.g.*

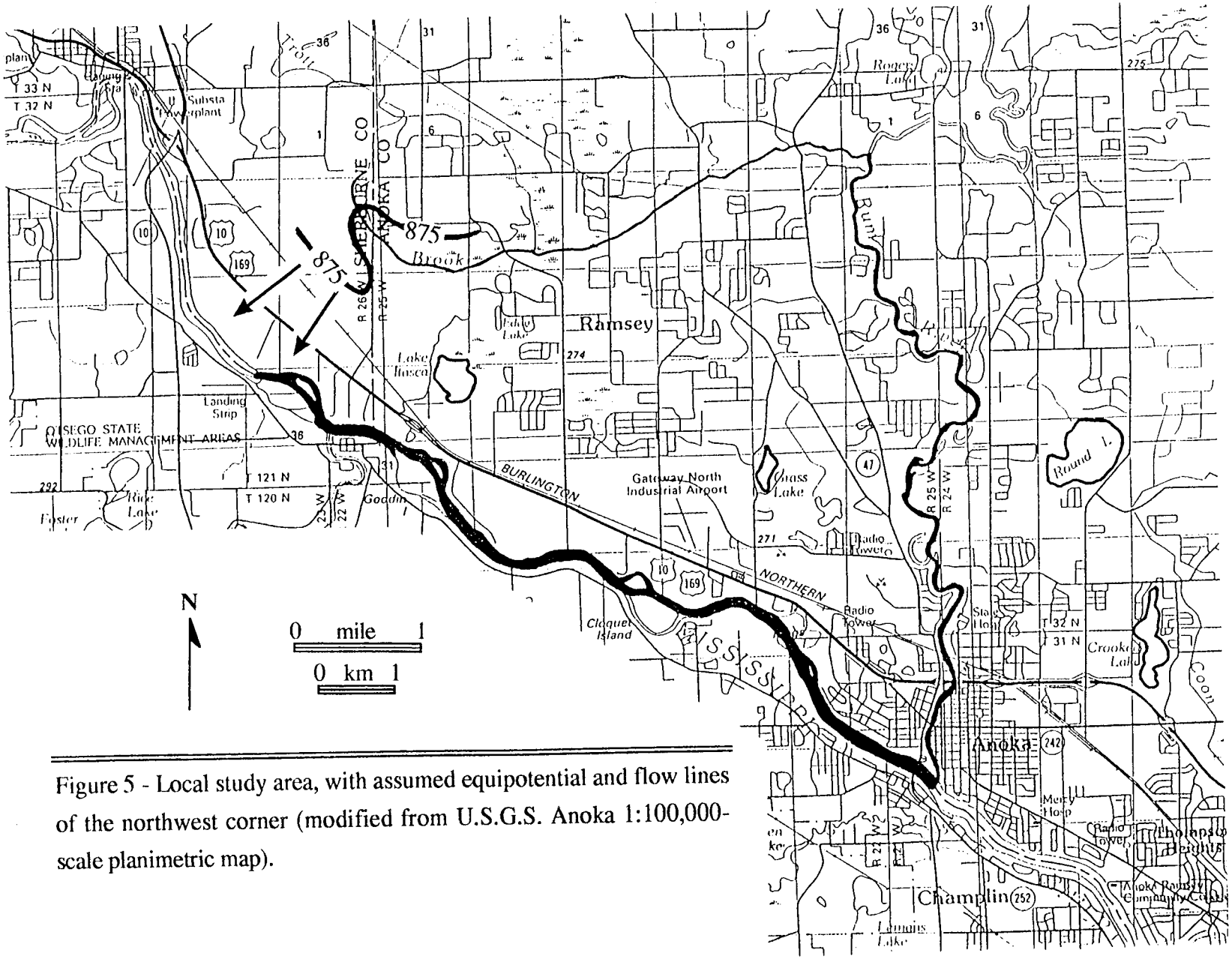
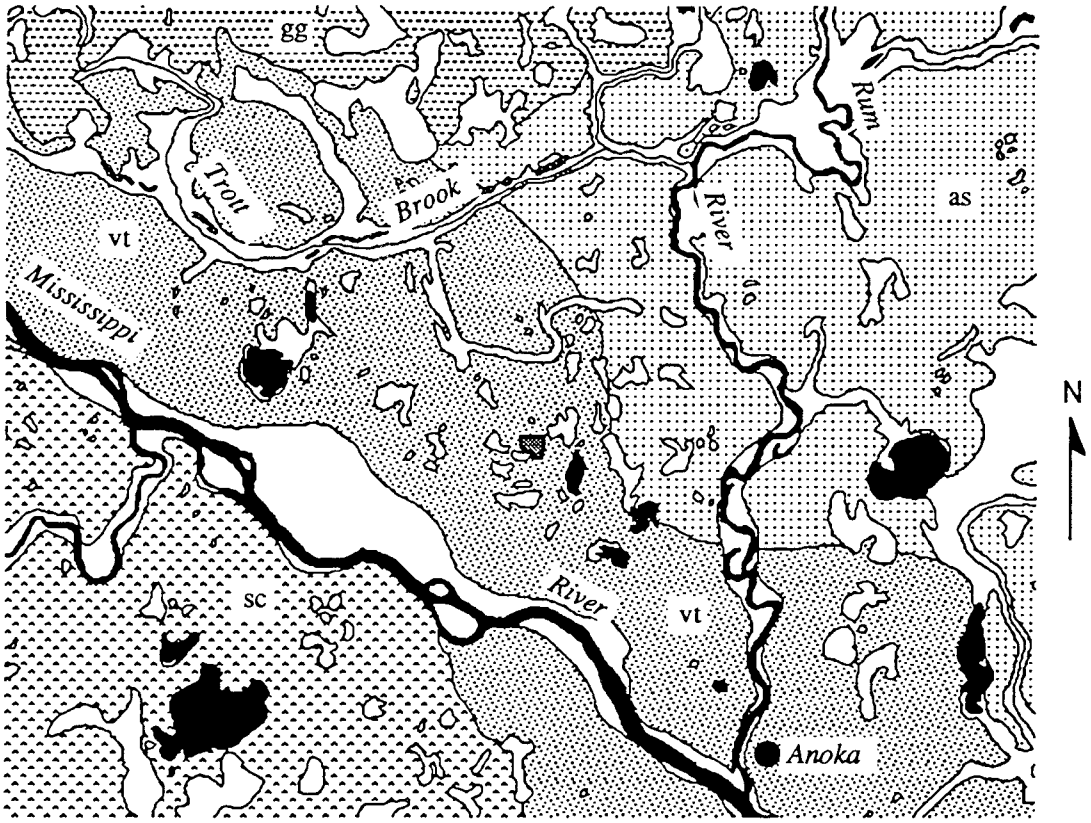



Figure 5 - Local study area, with assumed equipotential and flow lines of the northwest corner (modified from U.S.G.S. Anoka 1:100,000-scale planimetric map).




kilometers 0 1 2 3 4 5

miles 0 1 2 3

**Undifferentiated Lowlands.** Topographically low areas presently occupied by peat bogs, wetlands, lakes, and floodplains of creeks and rivers. Includes former tunnel valleys.

vt **Mississippi Valley Train.** Sands deposited by the glacial Mississippi River. The valley train is distinguished from the Anoka sandplain and from recent floodplains by elevation differences and by large braided channel patterns visible on air photos.

as **Anoka Sandplain.** Outwash sands and gravels distributed by the glacial Mississippi River as the Grantsburg sublobe wasted back to the southwest.

gg **Grantsburg Sublobe Ground Moraine.** Gray to buff fine-grained till.

**Eskers.** Sand and gravel deposited in the Superior lobe's sub-glacial tunnel valleys, which are characterized by long, southwest-trending topographic depressions that typically contain eskers.

sc **St. Croix Moraine.** A topographic high indicating the maximum extent of the Superior lobe. The moraine is blanketed with Grantsburg till in this study area.

Lakes, rivers, creeks, major ditches

the Anoka Regional Sanitary Landfill (near center of map)

Figure 6 - Surficial geology surrounding the Anoka landfill (modified from Quinn, 1992).

Freeze and Cherry, 1979, p. 29), and is within the range of $7x10^{-4}$ to $2x10^{-3}$ ft/s ($2x10^{-2}$ to $7x10^{-2}$ cm/s) estimated by Helgesen and Lindholm (1977) for the Anoka sandplain. It should be noted that these empirical K values are for particular samples of surficial Mississippi valley-train sands. They do not necessarily represent the average conductivity of sands of the valley train or the Anoka sandplain proper elsewhere in the study area, either areally or stratigraphically. However, reasonably close agreement is expected. This K value simply provides an appropriate first approximation for the hydraulic conductivity of the surficial sands; it is subject to change during the calibration of the numerical model.

A slight anisotropic component of permeability may be imparted to fluvial sand during deposition. Sand grains deposited by a uni-directional current exhibit imbrication; that is, elongate grains are aligned parallel to the current direction and they dip upstream at an angle of 10 to 25° from the horizontal (Potter and Mast, 1963). This imbrication produces a fabric in the porous medium which is aligned with cross-bed dip direction. The orientation of the direction of maximum permeability has been shown to be aligned parallel to cross-bed direction in sandstone (Mast and Potter, 1963) and in unconsolidated Holocene sands (Pryor, 1972).

In the local study area, large braided channels are visible on air-photos. Although the flow of the Mississippi valley train was generally to the southeast, the surficial channels are quite serpentine, without an apparent overall trend. Strike and dip measurements taken at the three small sandpits near the landfill indicate widely varying cross-bed orientations. Since only surficial information is available, general trends in the saturated zone are not known. Based on this information, the slight small-scale directional permeability imparted by depositional processes is considered negligible, and the surficial sand aquifer of this study is assumed to be isotropic.

*Hydrogeology of deeper drift*

Of the 152 logs of private wells inspected in the local study area, all describe a thickness of surficial sand and gravel (mean thickness 47 ft or 14 m). Most then encounter a unit described as "gray clay" by the drillers. The remainder of the logs show a "clay" or "brown clay" below the sandplain. This is interpreted to indicate that Grantsburg sublobe till forms the base of the surficial sandplain throughout the modeling area. This buried till unit is the same as the exposed Grantsburg sublobe ground moraine of the northern portion of Figure 6.

Hydraulic conductivity analyses of Grantsburg till yield a $K_H$ of $2.5x10^{-6}$ to $3.0x10^{-7}$ ft/s ($7.7x10^{-5}$ to $9.0x10^{-6}$ cm/s) obtained by slug/baildown tests and a $K_V$ of $1.5x10^{-8}$ to $7.5x10^{-11}$ ft/s ($4.5x10^{-7}$ to $2.3x10^{-9}$ cm/s) from laboratory tests (K. Keen, 1991, written communication).

Studies of fine-grained tills in North America have demonstrated that a surficial till is expected to have an upper, weathered zone that is hydrologically active, below which the till is an unweathered, unfractured aquitard (*e.g.* Ruland and others 1991; Hendry, 1982, 1988; Cravens and Ruedisili, 1987). However, Williams and Farvolden (1967) and Ruland and others note a minor amount of fracturing into the deeper, unweathered zone. No examples were found in the literature which address the possibility of fractures in deep, buried tills. No studies concerned with characterization of fractures in Minnesota tills are known.

Within the modeling area, the buried Grantsburg sublobe till appears to be continuous and of appreciable thickness. Therefore this unit is assumed to be an aquitard in the study area.

Below the Grantsburg sublobe till the drift is believed to be a complex assemblage of Superior lobe till and outwash, and pre-Late Wisconsin drift. This arrangement is generally supported in the study area by inspection of drillers' logs.

The three-dimensional correlation structure of the glacial drift units in the study area has been investigated using geostatistical techniques by Quinn (1992) and Quinn and others (1992).

*Bedrock geology and hydrogeology*

A thick sequence of sedimentary rocks of Cambrian and Ordovician ages exists above a faulted Precambrian basement in southeastern Minnesota (Austin, 1972). Within the study area, the bedrock is predominantly the Cambrian St. Lawrence and Franconia Formations (Austin, 1972; Jirsa and others, 1986; Olsen and Bloomgren, 1989).

The St. Lawrence is comprised of silty, sandy, shaley, and dolomitic rocks (Austin, 1972). Its expected thickness is approximately 50 ft (15 m), however the upper surface of the formation is partially eroded in the vicinity of the landfill (Austin, 1972). In the Twin Cities basin the Franconia Formation is typically a fine-grained glauconitic sandstone and siltstone, with a thickness of about 155 ft (47 m) (Mossler, 1972). A few isolated knobs of Jordan Sandstone have been mapped, and buried bedrock valleys are cut into the Ironton-Galesville and the Eau Claire Formations in the region (Jirsa and others, 1986).

The St. Lawrence Formation is considered to be a confining unit (Hogberg, 1972; Kanivetsky, 1989). The underlying Franconia Formation has a permeable upper portion, but is predominantly an aquitard (Hogberg, 1972; Walton, 1986). The St. Lawrence and Franconia Formations have been modeled as a single confining unit in a Twin Cities regional aquifer study by Schoenberg (1990).

Below the Franconia are the Ironton and Galesville Sandstones, which are productive aquifers (Hogberg, 1972). The Franconia, despite the low permeability of its lower half, is sometimes considered part of the Franconia-Ironton-Galesville aquifer. An inspection of the potentiometric head of the Hennepin County portion of this system suggests that it discharges to the Mississippi River (Kanivetsky, 1989), presumably through the St. Lawrence/Franconia (Jirsa and others, 1986; Kanivetsky, 1989).

*Surface waters*

Because of the high infiltration rate of the sandy soils of the Anoka sandplain, the area near the Anoka landfill has few streams or ditches.

The Rum River flows to the south about 2 miles (3 km) east of the landfill (Figure 5). A USGS gaging station is located on the Rum about 5 miles (8 km) upstream from the confluence of the Rum River and Trott Brook. Discharge information from U.S. Geological Survey water resources data indicate that river stages at this station generally fluctuate between 862.7 ft (263.0 m) to 866.3 ft (264.0 m). On the 7.5-minute topographic maps, the contours that cross the Rum near this station correspond closely to the expected value of the river stage. The contours crossing the Rum River along the eastern border of the study area are therefore assumed to represent the average river stage at each location.

The Mississippi River flows southeasterly less than 2 miles (3.2 km) south of the Anoka Landfill. A USGS gaging station is located 6.5 miles (10.5 km) downstream from Anoka, near Coon Rapids. However, the station is on the downstream side of the Coon Rapids dam, which, according to the topographic map, causes a drop of about 19 ft (5.8 m). Upstream from Anoka, the next gaging station is at Elk River, but data collection there ceased decades ago. Consequently, no accurate temporal values are available for the stage of the Mississippi River near the landfill.

Trott Brook is a perennial stream located 3 miles (5 km) north of the landfill within a tunnel valley (Figure 6). No gaging station exists along the brook to aid in validating the contours crossing it on the topographic maps. The topographic contours crossing the Mississippi River and Trott Brook are assumed to represent average river stage conditions.

Within the local study area, all three drainageways are located on the Mississippi valley train and/or the Anoka sandplain. Because of the permeable nature of their setting, they are all expected to be in direct contact with the shallow groundwater flow system. Lindgren (1990) demonstrated the strong hydraulic connection between the Mississippi River stage and observation wells in the valley train aquifer. Ericson and others (1974), in a regional hydrology study, mapped the Mississippi, the Rum, and Trott Brook as gaining streams.

Whether Trott Brook is truly a gaining stream on both its northern and southern is somewhat questionable. It is even more uncertain as to whether the brook serves as a discharge

point for the entire saturated thickness of the Anoka sandplain aquifer. Ditches and streams are known to provide partial to complete capture of groundwater in unconfined flow systems (Chambers and Bahr, 1992; Zheng and others, 1988b). No field data are available to characterize vertical or horizontal hydraulic gradients in the vicinity of the brook. Zheng and others (1988a) developed an equation relating the depth of capture of ditches to several hydrogeological factors. This approach, however, calls for the ditch or stream to be located on a linearly sloping water table. Because the brook is believed to receive discharge from both banks, the method of Zheng and others is not applicable. In this study, Trott Brook, like the Mississippi and Rum Rivers, is assumed to function as a constant head boundary. That is, these waterways are assumed to receive discharge from the entire saturated thickness of the sandplain on both banks.

Lakes are common in the pitted sandplain and valley train. Because they are located in highly permeable basins, these lakes are also expected to be in direct connection with the surficial sand aquifer. The Minnesota Department of Natural Resources Division of Waters compiles data on lake levels of four Anoka County lakes situated on the sandplain near the city of Anoka. These include Crooked, Round, Sunfish (Grass), and Itasca (Figure 5). At Sunfish Lake, however, the monitoring program has been in effect only since 1989. The lake levels (Figure 7) reflect seasonal and year-to-year changes in the stresses on the shallow groundwater system, including recovery after the drought years of 1988 and 1989. Although only infrequent sampling occurred at Sunfish Lake after 1991, the data indicate an anomalous dip in lake level that is probably due to remediation at the landfill. Because the shallow groundwater is expected to interact closely with these sandplain lakes, general changes in the water table should closely match the lake level changes. The close similarities of lake level changes among the four lakes suggest that pre-1989 levels of Sunfish Lake may be approximately extrapolated to aid in estimating the water table elevation near the landfill for the period of 1985-1989.

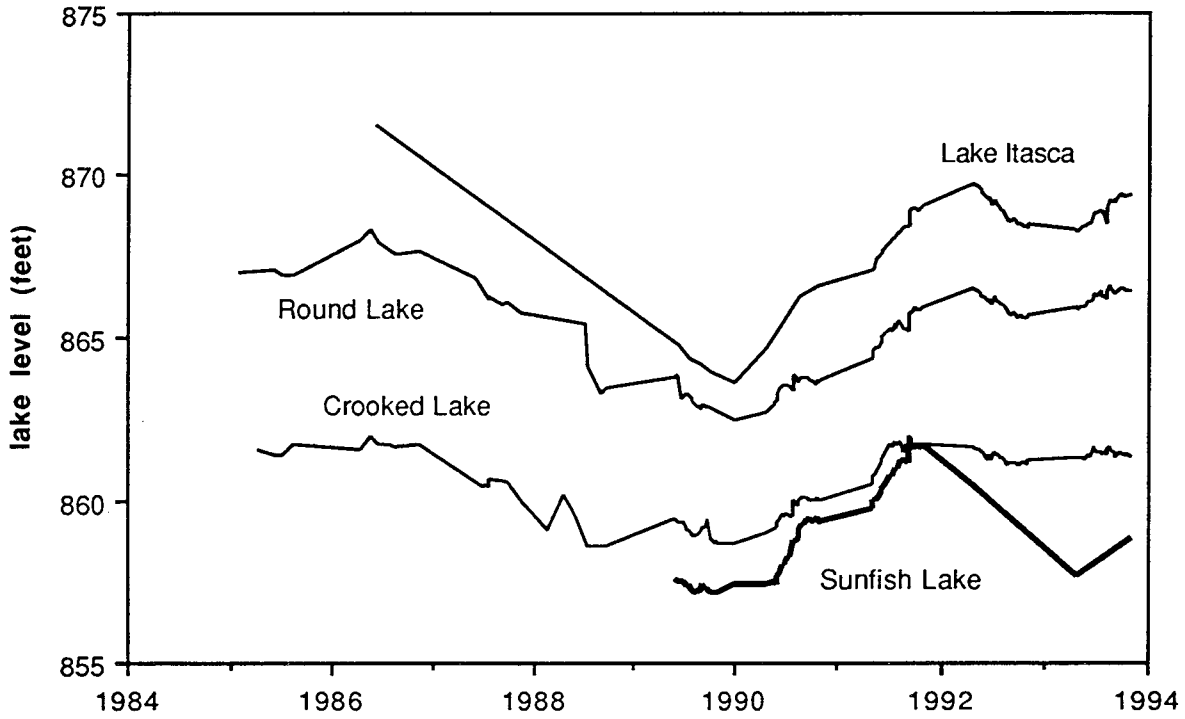



Figure 7 - Lake level changes of four lakes situated on the Anoka sandplain, within or near the local study area (data from Minnesota Department of Natural Resources).

*Recharge to the Anoka Sandplain*

In the local study area the mean annual precipitation is about 28 inches (71 cm), including the contribution from snowfall, which averages 45 inches (114 cm) annually (Kuehnast, 1978).

Expected characteristics of the permeable soils developed on a sandplain include high infiltration, high evapotranspiration, and low surface runoff. Precipitation and snowmelt, therefore, primarily either recharge the groundwater or are lost to evapotranspiration. Helgesen and Lindholm (1977) performed a two-year study of the hydrographs of 8 water table observation wells on the Anoka sandplain. They calculated an estimated average annual recharge to the surficial aquifer of 11 in (28 cm).

*Local Flow System*

An inspection of the average stages of the Mississippi and Rum Rivers and the Trott Brook, together with lake level data, indicate that groundwater near the landfill is expected to flow in a southeasterly direction. However, the configuration of streamlines in the vicinity of the landfill remains unknown because the landfill should be on or near a groundwater flow divide. The orientation of the streamlines will be addressed in this study by a combination of numerical flow modeling and advective transport modeling.

Because of the possibility of increased recharge in the waste disposal area, a water table mound may be present beneath a landfill. Slight mounding has been shown to occur at unlined landfills (MacFarlane and others, 1983; Kehew and Passero, 1990), although MacFarlane and others demonstrated that the mound was a transient feature that occurred in conjunction with snowmelt and spring rains. The possibility of mounding is unlikely at the Anoka landfill because of high infiltration capacities of both the sandplain and the landfill.

In Ramsey township, homeowners rely on private wells for their drinking water supply. An inspection of the logs of wells in the local study area indicated that nearly all of these wells are finished in either the permeable portion of the Franconia Formation or in the deeper Ironton Formation.

## MODELING GROUNDWATER FLOW AND LEACHATE MIGRATION

### *Hydrogeological facies model*

The private well log data support a fairly simple hydrogeological glacial facies model for the local study area. The uppermost unit of this model is comprised of the Anoka sandplain aquifer, which includes the Anoka sandplain and associated Mississippi valley-train sands. The thickness of this unit is approximately 47 feet (14 m). The sandplain is unconformably underlain by Grantsburg sublobe till, as interpreted from the drillers' descriptions and supported by glacial geological studies (Chernicoff, 1983, 1980; Wright and others, 1973). This till unit is expected to be continuous with the ground moraine in the north-central portion of Figure 6. The Grantsburg till is assumed to provide a suitable aquitard, thereby reducing the modeling effort to include only the surficial sand aquifer.

This model may seem simple, however, a facies model may not include all local hydrogeological features (Fraser and Bleuer, 1987). Even a relatively simple geologic framework does not preclude the possibility of surprising stratigraphic arrangements.

### *Kriging the basal elevation of the surficial aquifer*

The transmissivity of a surficial aquifer is dependent on two variables: the hydraulic conductivity and the saturated thickness, which is a function of the elevations of both the water table and the base of the aquifer. An investigation of the flow system of an unconfined aquifer must therefore consider the spatially varying basal elevation of the aquifer.

This phase of the study involves a geostatistical analysis of the areal distribution of the elevation of the contact between the Anoka sandplain aquifer and the underlying Grantsburg sublobe till aquitard. The study area is represented by cells with dimensions of 660 ft by 660 ft (201 m by 201 m). This is the same grid as that in the finite difference groundwater modeling portion of this study.

The database for this analysis is the drillers' logs of 152 private wells (Figure 8). A statewide inventory of this information is maintained at the Minnesota Geological Survey (MGS).

Several sources of error are inherent to the data set. First is the estimation of the elevation of the land at the wellhead. Addresses supplied on each well log facilitated the locating of each property on a county plat map. An approximate topographic value for the top elevation of each log was then determined by checking the corresponding location on a USGS 7.5-minute topographic map. This may result in an error of up to 5 feet (1.5 m) for the elevation of the top of the well log. Consequently, the elevation of the sand/till contact would be off by the same amount.

A problem affecting the quality of the data is that they were not recorded for use in a scientific analysis; rather, logging was a perfunctory task. For several reasons, the recorded depths to geologic contacts may be regarded with some degree of suspicion. The logging was carried out by numerous individuals from different contractors. MGS personnel who are familiar with the methods of local drillers describe that when drilling through drift, the stratigraphy is logged primarily by the speed at which the drilling is progressing: fast implies sand or gravel, slow implies a till or lacustrine clay. Seldom are samples taken or observations made of the materials which are brought up in drilling fluids. Some logs contain fairly detailed stratigraphy while others only describe units that are greater than 50 feet (15 m) thick, suggesting that some amount of information is often unrecorded. Also, the depth to contacts may be affected by boreholes that are not vertical or straight.

### *Variogram results*

The variogram investigation was carried out using the program VARIO. Directional variograms were calculated with an angular tolerance of 22.5° (Figure 9). This two-dimensional examination produced a linear model variogram with a nugget of 240 $ft^2$ (22 $m^2$) and fairly strong anisotropy (Figure 10a). The experimental omnidirectional variogram is linear through a distance of 25,000 ft (7,600 m), after which the variogram breaks down due to a lack of data at large separation distances. An additional directional variogram, oriented north-northwest to south-

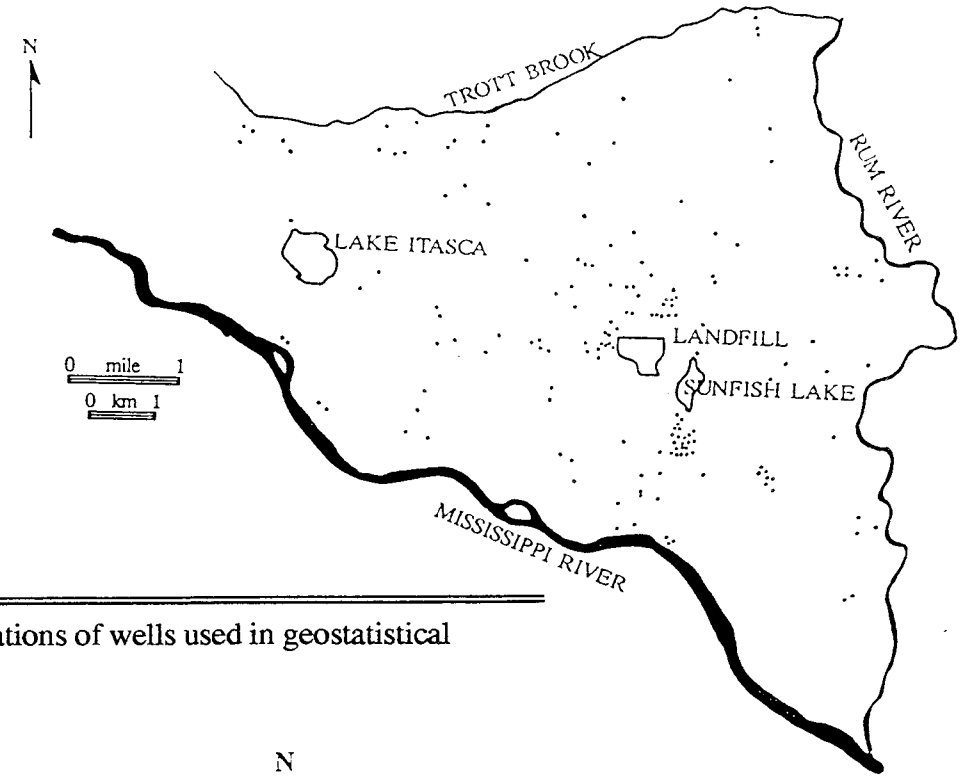



Figure 8 - Locations of wells used in geostatistical analyses.

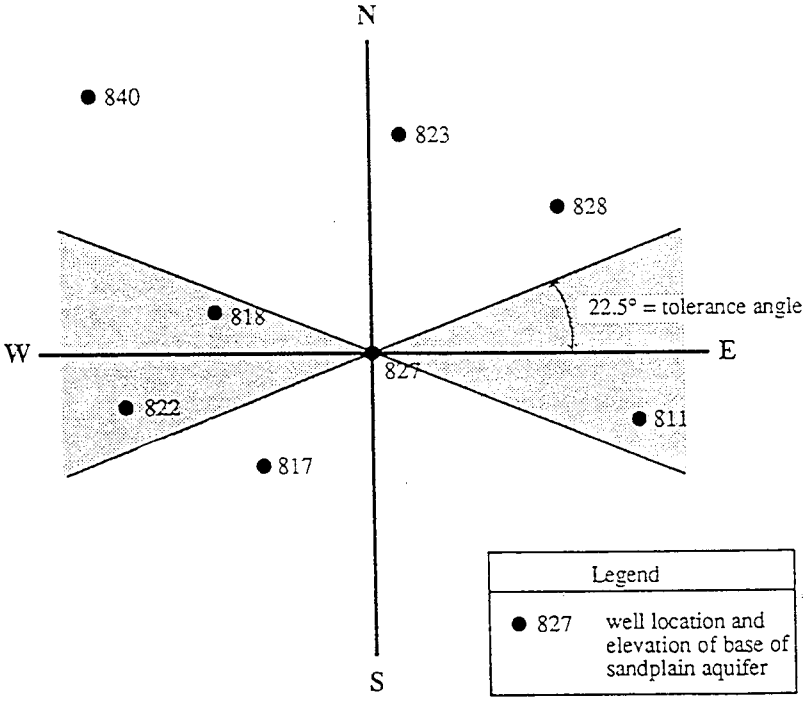
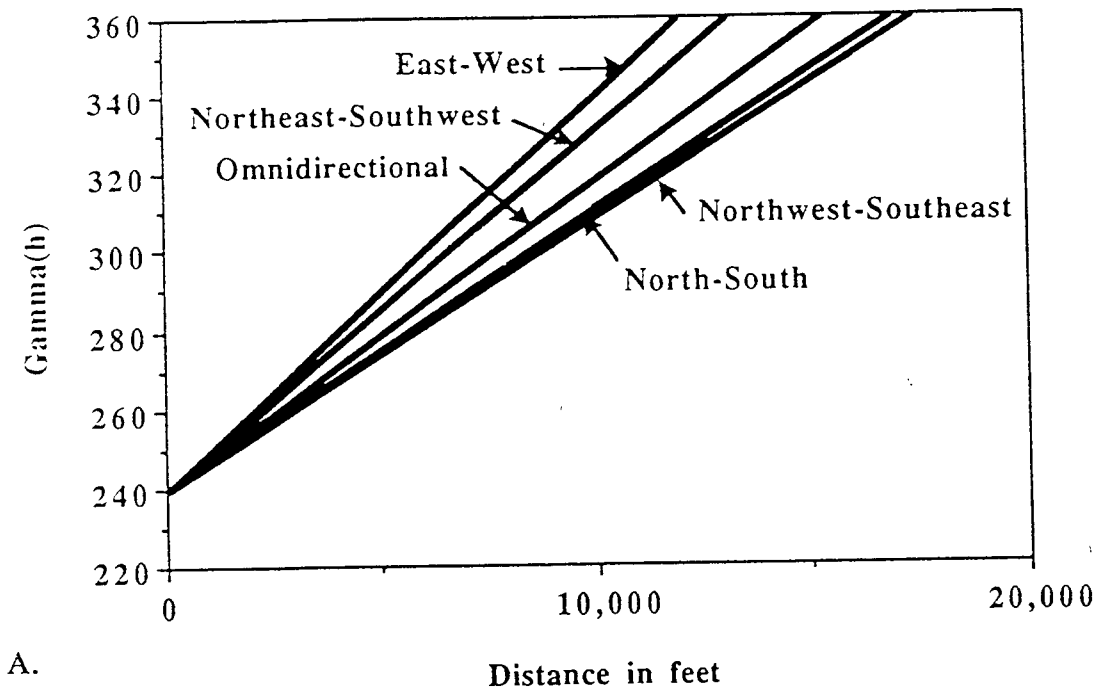
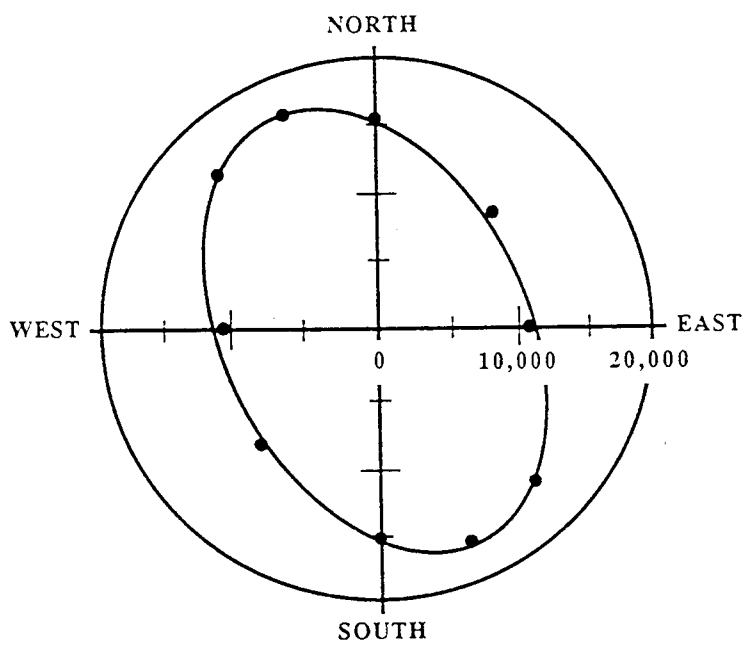



Figure 9 - Example of a two-dimensional directional variogram search area. The shaded area represents an east-west search for a given data location, with a tolerance angle of 22.5°.




A.




B.

Figure 10 - A. Omnidirectional and directional linear variograms for the basal elevation of the sandplain. B. Anisotropy ellipse for the variograms of the basal elevation of the sandplain. Plotted values are the ranges corresponding to an arbitrarily chosen $\gamma(h)$ value of 350 $ft^2$.

southeast, has the lowest slope and therefore represents the direction of maximum range. Geostatistical anisotropy is often illustrated by an ellipse, with major and minor axes corresponding to the axes of geometric anisotropy. In this investigation the anisotropy is oriented north-northwest to south-southeast with a ratio of approximately 3:2 (Figure 10b).

Because the nugget effect is equal to the variance of the data at zero separation, the nugget of 240 $ft^2$ (22 $m^2$) indicates a standard deviation of 15.5 ft (4.7 m) at h=0. This value may be attributed to geologic complexity, lack of careful logging, the error related to estimating the elevation of the land at the well location, or a combination of these factors. However, the relative impact of each cannot be determined.

In order to determine the sensitivity of the directional variogram to the angular tolerance, the tolerance was opened up to 30°, allowing the comparison of more sample pairs. The variograms, however, showed no noticeable differences when compared to the analyses with 22.5° tolerance angles.

Two possible geologic interpretations of the observed anisotropy are offered. The glacial Mississippi river, during the formation of both the Anoka sandplain and the Mississippi valley train, had a generally southeastern flow direction in this vicinity (Figure 4). Erosion by meltwater streams trending in this direction would have produced the spatial correlation illustrated by Figure 10. Another possibility is the influence of glacial depositional processes associated with the flow directions of ice lobes. As discussed previously, in the study area the Superior lobe flowed southwesterly, and the Grantsburg sublobe advanced northeasterly (Figure 3). These flow directions are both essentially perpendicular to the northwest-to-southeast trending anisotropy. The Superior lobe's St. Croix moraine, located along the southwest border of the study area, has the same trend as the spatial structure. This topographic influence on the depositional processes associated with these two most recent advances could have imparted some of the spatial correlation structure apparent in the variogram analysis.

*Kriging*

Ordinary kriging incorporates the anisotropic structure revealed by the variogram analyses to produce the best estimate of the sandplain's basal elevation at each grid cell in the study area. To determine the slope of a linear variogram, Geo-EAS requires specification of a $\gamma(h)$ and the corresponding h value (Table 1). The kriging estimations for the sand/till contact are contoured in Figure 11. The cell-by-cell values of this output are utilized as input in the MODFLOW numerical model of groundwater flow in the study area.

Table 1 - Geo-EAS input for kriging of elevation of aquifer base

| | |
|---|---|
| Linear variogram model | |
| Nugget Effect | = 240 $ft^2$ (22 $m^2$) |
| Major Range[1] | = 17,000 ft (5,200 m) |
| Minor Range[1] | = 11,000 ft (3,600 m) |
| Sill Value[1] | = 350 $ft^2$ (33 $m^2$) |
| Angle[2] | = 112.5° (NNW-SSE) |
| Maximum Points[3] | = 16 |
| Search Radius[4] | = 15,000 ft (4,600 m) |

[1] In a linear variogram model, the range and sill simply define the slope; they are not true ranges or sills.
[2] The angle is the direction (0 to 180°) of the maximum range. Geo-EAS measures this angle counterclockwise from east = 0°.
[3] The maximum number of data points included in the determination of the estimate of a block.
[4] The search radius is the maximum distance allowed in a data point search for the calculation of the value of a given block.

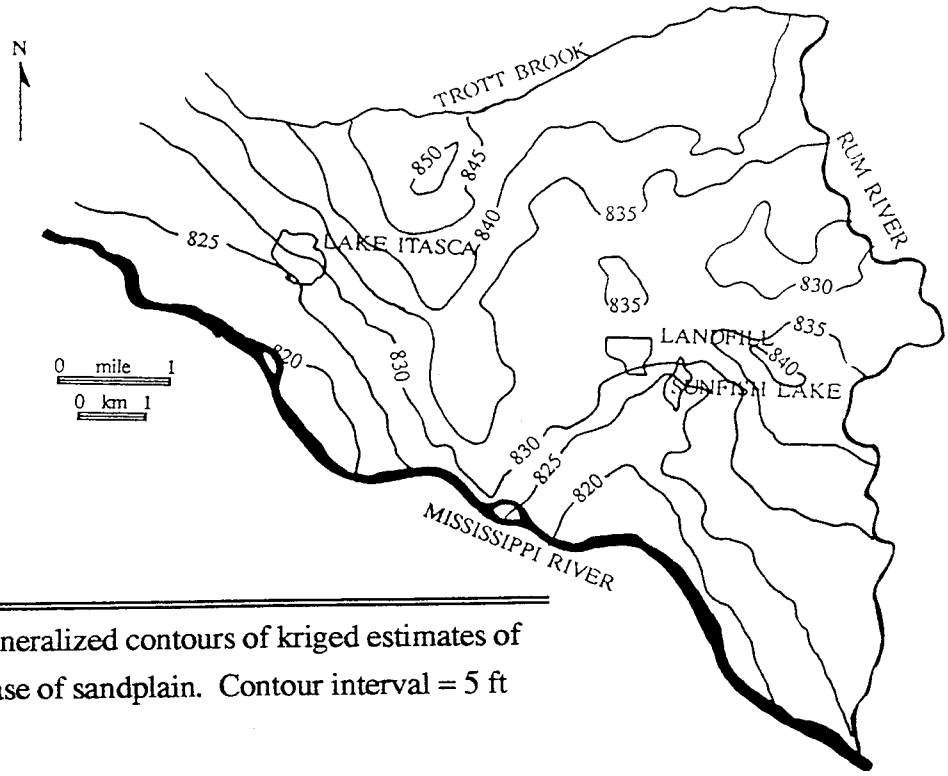



Figure 11 - Generalized contours of kriged estimates of elevation of base of sandplain. Contour interval = 5 ft (1.5 m).

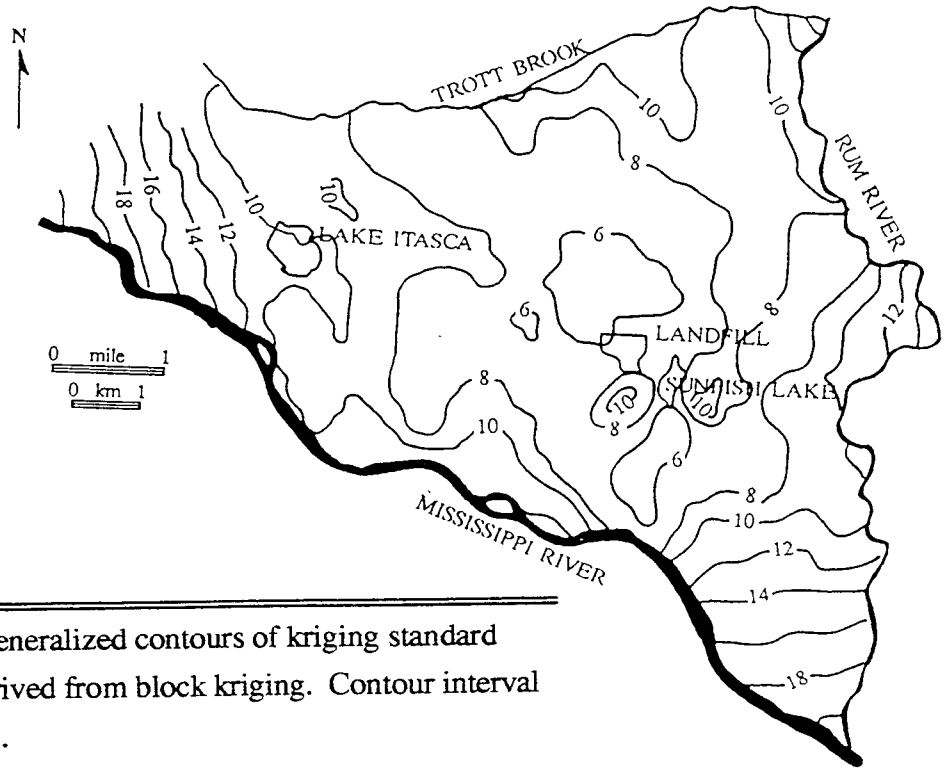



Figure 12 - Generalized contours of kriging standard deviations derived from block kriging. Contour interval = 2 ft (0.6 m).

Geo-EAS produced the same estimation output using both point and block kriging; however, the kriging standard deviations were markedly different. For block kriging, the kriging standard deviation was roughly 6 to 10 ft (2 to 3 m) throughout the center of the study area, with greater values around the edges (Figure 12). For point kriging, the value was about 16 to 18 ft (5.2 to 5.9 m) in the central portion. The discrepancy arises because the block kriging is describing the variance on the estimate for a large (660 by 660 ft, or 201 by 201 m) block, while the point kriging variance is for comparison of individual points (*i.e.* well locations). Because the finite difference model is concerned with the values of grid cells, the kriging standard deviation of 6 to 10 ft (2 to 3 m) represents a fairly low level of error on the estimate of cells in much of the modeling area. Kriging standard deviations, however, are sensitive to variogram parameters, and are more useful for representing relative error rather than accurately calculated error.

***Flow modeling***

A deterministic finite difference numerical model of groundwater flow in the local study area was developed in order to quantitatively assess the flow system in the local study area. The results from the calibrated version of the flow model are then utilized by a particle-tracking program in order to investigate the direction and rate of migration of conservative contaminants.

The boundaries of the local study area were chosen from inferred hydrogeologic boundaries, as described below. The area was discretized into a grid of cells measuring 660 ft by 660 ft (201 m by 201 m), using square mile sections as a base (Figure 13). With 64 cells per square mile (about 25 cells per square km), the grid is designed to represent the study area at a reasonably fine resolution. The effect of boundary condition discretization on heads near the landfill is expected to be minimized by the fine grid and the distance from the boundaries. The grid

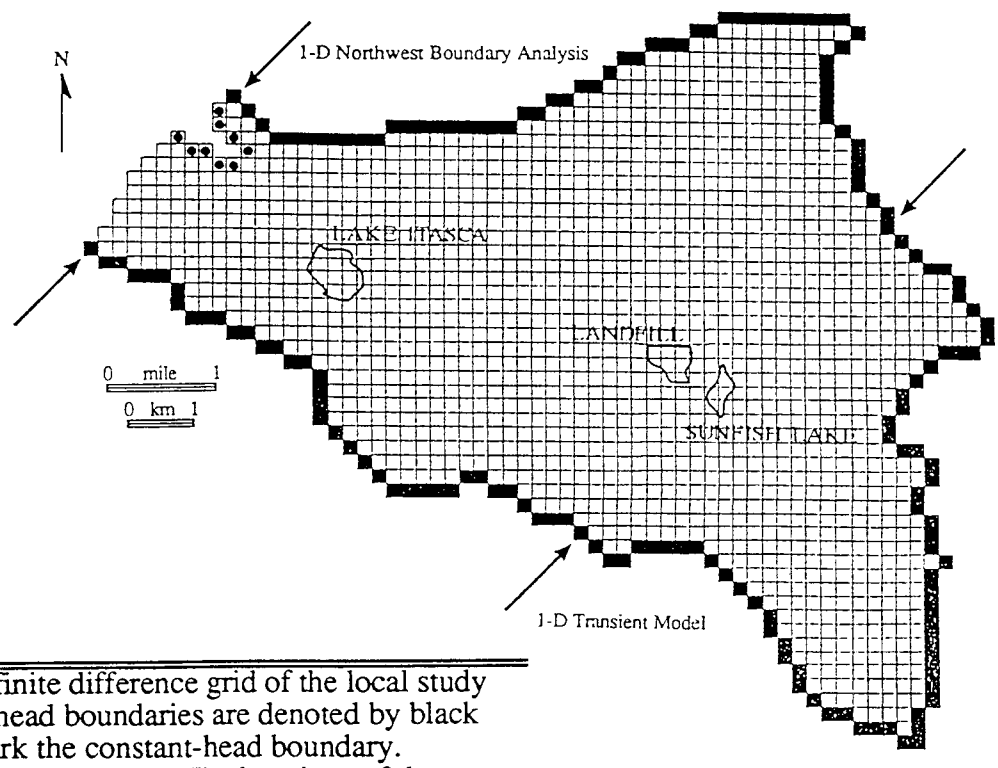



Figure 13 - The finite difference grid of the local study area. Specified-head boundaries are denoted by black squares; dots mark the constant-head boundary. Variable-head cells are white. The locations of the one-dimensional models are noted. Grid cells measure 660 ft (201 m) on each side.

is rectangular, 63 cells by 53 cells. Included are 1,428 variable-head cells, 163 constant-head cells along the three specified-head boundaries (rivers), 9 constant-head cells along an inferred constant-head boundary, and 1,739 inactive cells. The modeling is performed using MODFLOW (MacDonald and Harbaugh, 1988).

The model is a one-layer analysis. The suitability of the two-dimensional assumption will be discussed. In this study, engineering data on heads and chloride concentrations are taken from 1987, because precipitation was near normal for the year.

*Model assumptions, initial parameters, and boundary conditions*

Five main variables or controls affect the model: the values for hydraulic conductivity and recharge, the effect exerted by the varying base of the sandplain, the dimensionality, and the assigned boundary conditions.

The geologic environment modeled in this study is relatively simple and homogeneous in comparison to many hydrogeologic settings. However, heterogeneities exist even in a sandy medium. The hydraulic conductivity of the sand aquifer has been estimated by grain-size analyses to be about $1.2 \times 10^{-3}$ ft/s ($3.5 \times 10^{-2}$ cm/s), as discussed previously. A detailed three-dimensional distribution of conductivity values is not available to characterize in more detail the hydraulic conductivity of the sandplain. Therefore neither detailed three-dimensional nor stochastic modeling is possible. Instead, this deterministic model focuses on calibrating to determine the effective horizontal hydraulic conductivity for the Anoka sandplain in this area of interest.

The average annual net recharge value of 11 in (28 cm) determined by Helgesen and Lindholm (1977) provides a reasonable recharge value to serve as input to the model. Because the recharge parameter is substantiated, its value is held constant, and the value for hydraulic conductivity is varied in order to calibrate the model. Surficial geologic mapping (Figure 6) and an inspection of the 152 well logs for the local study area suggest that, because of similar surficial materials, infiltration of precipitation may be expected to be uniform over the entire area of interest.

Because of the contrast in hydraulic conductivity of a minimum of four orders of magnitude between the sandplain and the underlying Grantsburg till, the till is considered to be an aquitard, and flow is modeled only in the surficial aquifer. Although the model consists of only one layer, including the varying base takes into account a three-dimensional aspect of the unconfined aquifer. Three-dimensional flow is ignored in the flow and particle-tracking analyses. Because this model is simplified into two dimensions, its head output is a function of an effective horizontal hydraulic conductivity for the entire saturated thickness of sand at each finite difference cell.

Proper boundary conditions are a crucial component of a groundwater model (Franke and Reilly, 1987; Franke and others, 1984). As described above, the contours on the topographic maps of the study area are assumed to represent average stages of the Rum River, Mississippi River, and Trott Brook. Geologic mapping (Figure 6) and an inspection of the well logs in the local study area suggest a highly permeable setting for the surface water bodies in the area of interest. The assumption is made, therefore, that the two major rivers, the brook, and the lakes in the sandplain are representations of the shallow flow system. Constant head values for each river cell are obtained by interpolating river stages between the locations where the contour lines cross the rivers on the topographic maps. Heads are also specified along an unnamed perennial tributary of the Rum River near the northeast corner of the study area.

The northwest edge of the study area, however, is not modeled by a specified-head boundary. Here, Trott Brook is assumed to be a gaining stream (Ericson and others, 1974) , even along its southern bank. An inferred 875 ft (267 m) constant-head boundary is therefore set from its position where it crosses the brook, through an area south of the brook (Figure 5). Groundwater in the northwestern portion of the study area is thereby forced to flow to both the brook and the Mississippi River. An upgradient flux is assumed to produce this configuration of equipotentials in this area.

The western edge of the modeling area is assumed to be a streamline, representing groundwater flowing straight from the 875 ft (267 m) equipotential to the Mississippi. This no-flow boundary is modeled as a linear border of active and inactive cells.

Several sources of error are inherent to the boundary condition and dimensionality assumptions. The specified-head boundaries are not completely justified in this two-dimensional model because the rivers are not fully penetrating. The two-dimensional modeling implies that all Anoka sandplain aquifer flow would be discharged along the stream boundaries. No information is available to determine whether the Rum River or Trott Brook may be considered regional boundaries relative to the entire saturated thickness of the Anoka sandplain.

Lake level data for several lakes in or near the local study area are available from the Minnesota Department of Natural Resources. Because the lakes are considered observation points of the water table, the lake level data (Figure 7) may be useful as a means of calibrating the input parameters of the flow model.

*Initial results*

Using a recharge of 11 in (28 cm) and the hydraulic conductivity suggested by the grain size analyses of 1.2 x $10^{-3}$ ft/s (3.5 x $10^{-2}$ cm/s), a somewhat reasonable representation of the groundwater flow system is determined, with groundwater near the landfill flowing in a southeast direction (Figure 14). However, a comparison of the water table contours with information on 1987 lake levels suggests that the water table elevation is too high in the vicinity of the landfill. An extrapolation of lake level data suggests that the level of Sunfish Lake was in the range of 858 to 860 ft (261.5 to 262.1 m) in 1987 (Figure 7).

*Calibration*

Based on the level of Sunfish Lake, the initial model run suggests that either the conductivity value used is too low, or the recharge value is too high, or a combination of these two factors. Because hydraulic conductivity within an outwash deposit typically ranges over several orders of magnitude, while the annual average recharge rate varies by much less, the hydraulic conductivity is the parameter that will be adjusted to calibrate the model. The recharge value, as determined by Helgesen and Lindholm (1977), is held constant.

Compared to Sunfish Lake, which covers several grid cells, calibration to water levels in monitoring wells provides greater accuracy. An initial monitoring network for any facility would

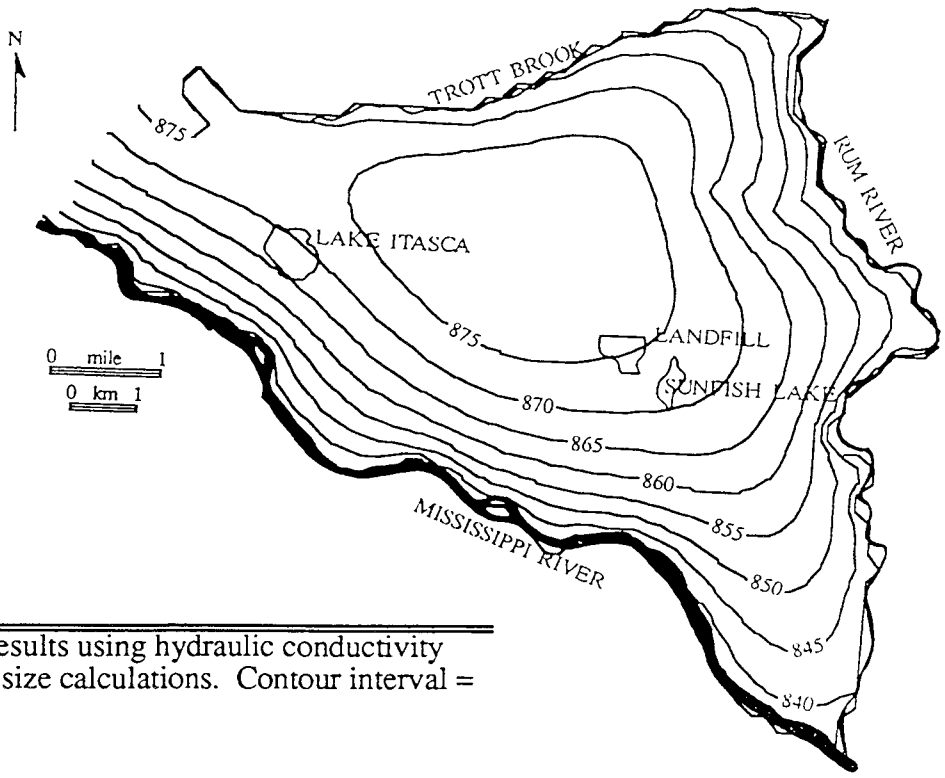



Figure 14 - Initial results using hydraulic conductivity derived from grain-size calculations. Contour interval = 5 ft (1.5 m).

include one upgradient well and three downgradient wells. Based solely on the contoured output, four water table monitoring wells from appropriate positions were chosen from the existing monitoring well network (Figure 2). The elevation of the saturated zone at each well location is found in the engineering database for a July 29, 1987 sampling event (Foth & Van Dyke, 1987). These values are used in calibrating the flow model more precisely than with lake level data alone. Through a series of trial-and-error model runs, an effective horizontal hydraulic conductivity of $2.7 \times 10^{-3}$ ft/s ($8.2 \times 10^{-2}$ cm/s) was determined for the local study area (Figure 15). The resulting equipotentials correspond well to the expected summer 1987 level of Sunfish Lake, and they match the water level data of the four monitoring wells adequately, within approximately 1 foot (0.3 m). The model's predicted water table elevation in the vicinity of Lake Itasca matches the historical data somewhat poorly, perhaps because the lake is a broad observation point of the water table located in an area of changing groundwater flow directions and hydraulic gradients. Lake Itasca is also more subject to boundary effects.

Modeling was performed using the strongly implicit procedure (McDonald and Harbaugh, 1988, chapter 12). Sixteen iterations were required to compute the results with a convergence criterion of 0.1 ft (3 cm).

The scale of the model ignores local effects, such as flow near lakes or variations in hydraulic conductivity distributions. A more detailed calibration of the flow model is not possible within the scope of this study because the only quantified water table observation points are the four engineering monitoring wells, Sunfish Lake, and Lake Itasca.

This groundwater model is believed to represent the unconfined flow system reasonably accurately. A volumetric budget analysis calculated by MODFLOW indicates model inputs of 4.3 $ft^3/s$ (0.12 $m^3/s$) from constant-head sources (the northwest constant-head boundary) and 18.8 $ft^3/s$ (0.53 $m^3/s$) from recharge by precipitation. Output is comprised of 23.2 $ft^3/s$ (0.66 $m^3/s$) to the specified-head boundaries along the Mississippi River, the Rum River, and Trott Brook. The percent discrepancy of input vs. output is negligible at 0.46%. The output file of this calibrated model is presented by Quinn (1992).

The model-determined hydraulic conductivity value of $2.7 \times 10^{-3}$ ft/s ($8.2 \times 10^{-2}$ cm/s) agrees well with values obtained through other methods. In two pump tests near the landfill, Foth & Van

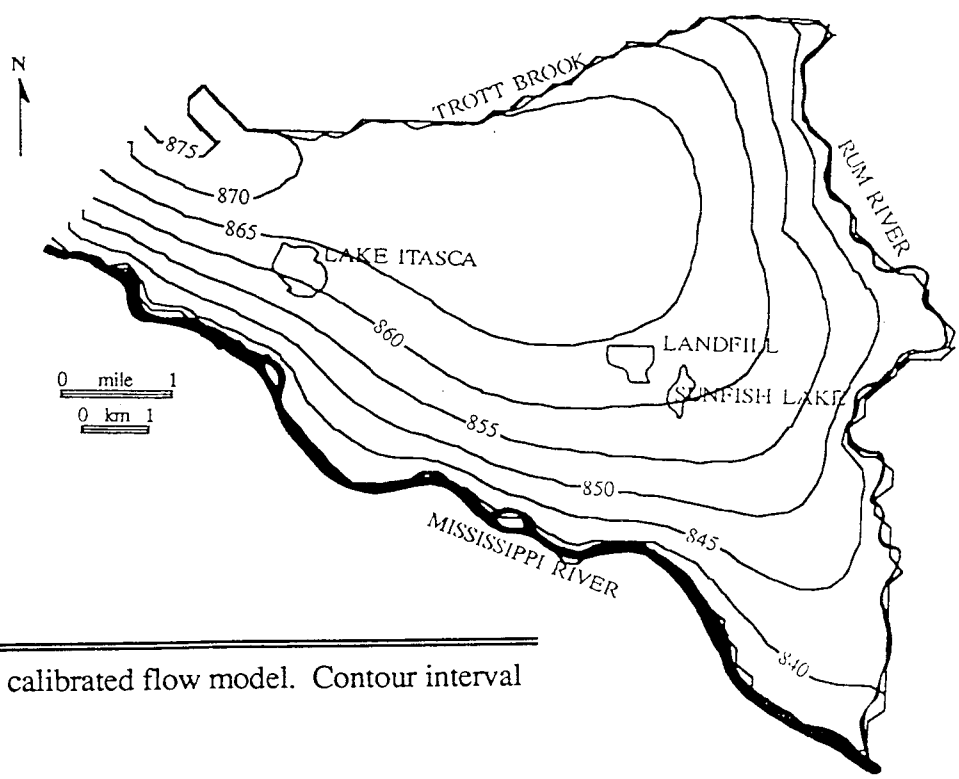



Figure 15 - The calibrated flow model. Contour interval = 5 ft (1.5 m).

Dyke (1989a) determined conductivity values ranging from $1.2 \times 10^{-3}$ to $4.6 \times 10^{-3}$ ft/s ($3.6 \times 10^{-2}$ to $1.4 \times 10^{-1}$ cm/s) in the sandplain aquifer. The mean pump test value was $2.1 \times 10^{-3}$ ft/s ($6.4 \times 10^{-2}$ cm/s). And, as discussed earlier, the mean conductivity value from grain-size analyses of surficial sand samples was approximately $1.2 \times 10^{-3}$ ft/s ($3.5 \times 10^{-2}$ cm/s).

*Sensitivity*

A series of sensitivity runs were performed in order to qualitatively assess the effect of varying the hydraulic conductivity (K) or recharge (R) while holding other parameters constant. The four trials included changing K and R each by -10% and +10%. With a decreased K or increased R, the calculated head at the location of the landfill increased roughly 1.5 ft (0.46 m) relative to the output of the calibrated model. In the case of increased K or decreased R, the head near the landfill decreased approximately the same amount.

Because the annual recharge rate may vary by more than 10%, the model is considered to be fairly sensitive to the recharge parameter. Recharge events no doubt have a great influence on heads in the sandplain aquifer. The model appears to be similarly affected by a slight change in hydraulic conductivity, and is considered sensitive to this parameter as well. Local variations in the actual horizontal hydraulic conductivity would therefore affect computed heads.

*Boundary condition analyses*

Two analyses were performed in order to test certain aspects of the boundary conditions of the calibrated groundwater flow model. Each of these models is one-dimensional and performed along diagonals of the two-dimensional grid. Square model cells measuring 933 ft (284 m) on each side are centered on cells of the two-dimensional grid. The best estimate of the basal elevation of the sandplain may then be assigned to each cell of the models. In both models, the parameters of recharge and calibrated hydraulic conductivity are the same as in the two-dimensional model. The finite difference codes written for these analyses use an explicit, iterative solution with a convergence criterion of 0.01 ft (0.3 cm).

The first model addresses the boundary condition of the northwest corner of the study area (Figure 5). An inspection of the hydrologic and geologic setting of the northwest area suggests two possibilities for groundwater flow directions. Trott Brook and the Mississippi are roughly parallel for a distance of about 4 mi (6 km). The surficial geologic materials in this region are primarily permeable Mississippi valley sands. Along this stretch, Trott Brook may be a gaining stream, receiving groundwater from both banks. Alternatively, the south side of the brook may be losing, resulting in southwesterly groundwater flow from the south side of the brook directly to the topographically lower Mississippi River.

These possibilities were explored by a one-dimensional finite difference model of the northwestern border (Figure 13). The modeled domain was 12 finite difference cells oriented along a northeast-southwest line.

Each variable-head cell contained the one-dimensional equation for unconfined steady-state flow (after Wang and Anderson, 1982, p. 53) modified to incorporate a non-zero basal aquifer elevation:

$$h_i = b_i + \left[ \frac{(h_{i-1} - b_{i-1})^2 + (h_{i+1} - b_{i+1})^2}{2} + \frac{R \cdot dx \cdot dy}{K} \right]^{0.5} \qquad \text{Eq. 5}$$

where
$h_i$ = head in cell i [L],
$b_i$ = basal elevation of aquifer in cell i [L],
$h_{i-1}$, $h_{i+1}$ = heads in adjacent cells [L],
$b_{i-1}$, $b_{i+1}$ = basal elevations in adjacent cells [L],
R = recharge [$LT^{-1}$],
dx, dy = cell dimensions [L], and
K = hydraulic conductivity [$LT^{-1}$].

A limitation of a one-dimensional model is that upgradient and downgradient components of flow are not taken into account. Also, the one-dimensional assumption ignores the fact that the rivers are not fully penetrating.

The results of this analysis are shown graphically in Figure 16. The simulation indicates continuously decreasing heads from Trott Brook to the Mississippi, with a low groundwater gradient near the brook. The presence or magnitude of an upgradient flux cannot easily be addressed. A small flux from upgradient would raise the heads just south of the brook to a level above the brook's stage, producing a situation in which Trott Brook would be gaining from both banks.

To compare the effect of different northwest boundaries, the MODFLOW model was run using a no-flow, streamline boundary straight from Trott Brook to the Mississippi. The brook in this case is a losing stream on the south bank. Only slight differences were noted in the arrangement of equipotentials away from the northwest corner (Figure 17). The two-dimensional model is therefore insensitive to any inaccuracy of the assigned northwest boundary.

A second boundary condition analysis examines the relationship between heads in the unconfined aquifer and changes in river stages. Clearly the river stages depicted on the topographic maps are not constants. Changes occur on time scales ranging from less than a day to periods of several years. However, these waterways, which surround most of the local study area, are believed to be sufficiently far from the landfill so that boundary effects will have minimal influence on flow in the central area of interest. A transient boundary condition analysis aids in testing this assumption.

The transient solution (after Wang and Anderson, 1982, p. 87), modified to incorporate the base of the aquifer, is:

$$h_i^{n+1} = b_i + \left\{ (h_i^n - b_i)^2 \cdot \left[ 1 - \frac{2 \cdot K \cdot (h_i^n - b_i) \cdot dt}{Sa^2} \right] + \frac{2 \cdot R \cdot (h_i^n - b_i) \cdot dt}{S} + \frac{K \cdot (h_i^n - b_i) \cdot dt}{Sa^2} \cdot \left[ (h_{i-1}^n - b_{i-1})^2 + (h_{i+1}^n - b_{i+1})^2 \right] \right\}^{0.5} \quad \text{Eq. 6}$$

where $h_i^{n+1}$ = head in cell i during present time step n + 1 [L],
$b_i$ = elevation of base of aquifer in cell i [L],
$h_i^n$ = head in cell i during previous time step n [L],
$h_{i-1}^n$ , $h_{i+1}^n$ = head in adjacent cells during previous time step n [L]
$b_{i-1}$ , $b_{i+1}$ = head in adjacent cells during previous time step n [L]
K = hydraulic conductivity [$LT^{-1}$],
dt = time step length [T],
S = specific yield [unitless],
a = length of side of square grid cell [L], and
R = recharge [$LT^{-1}$].

A sufficiently small dt value of five days prevents instability in the calculations (Wang and Anderson, 1982, p. 87).

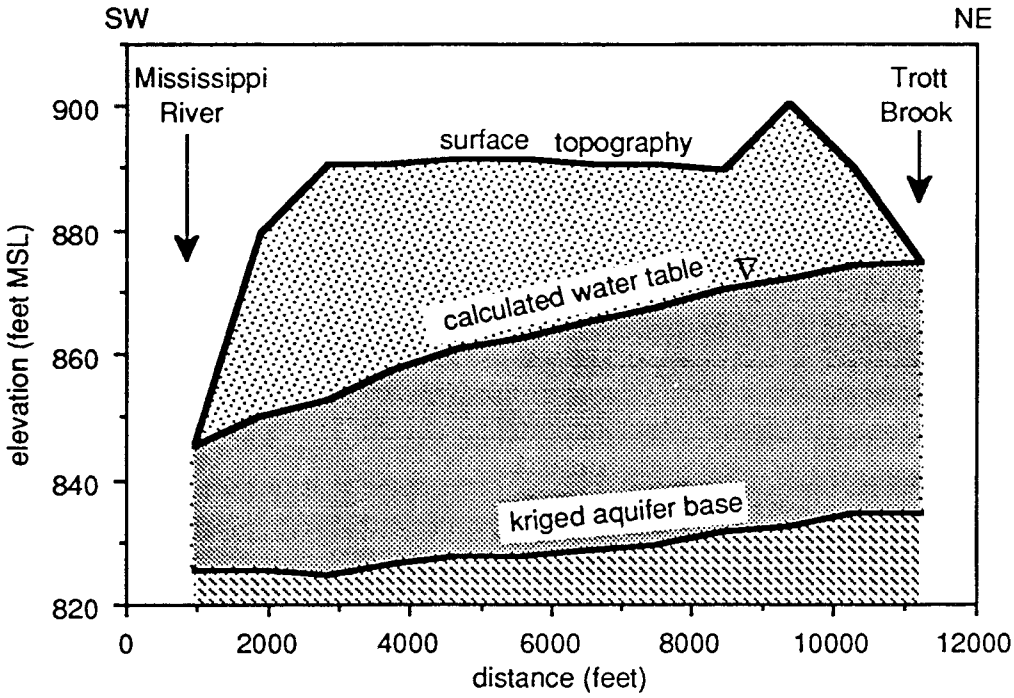



Figure 16 - Steady-state one-dimensional model results for northwest boundary condition analysis.

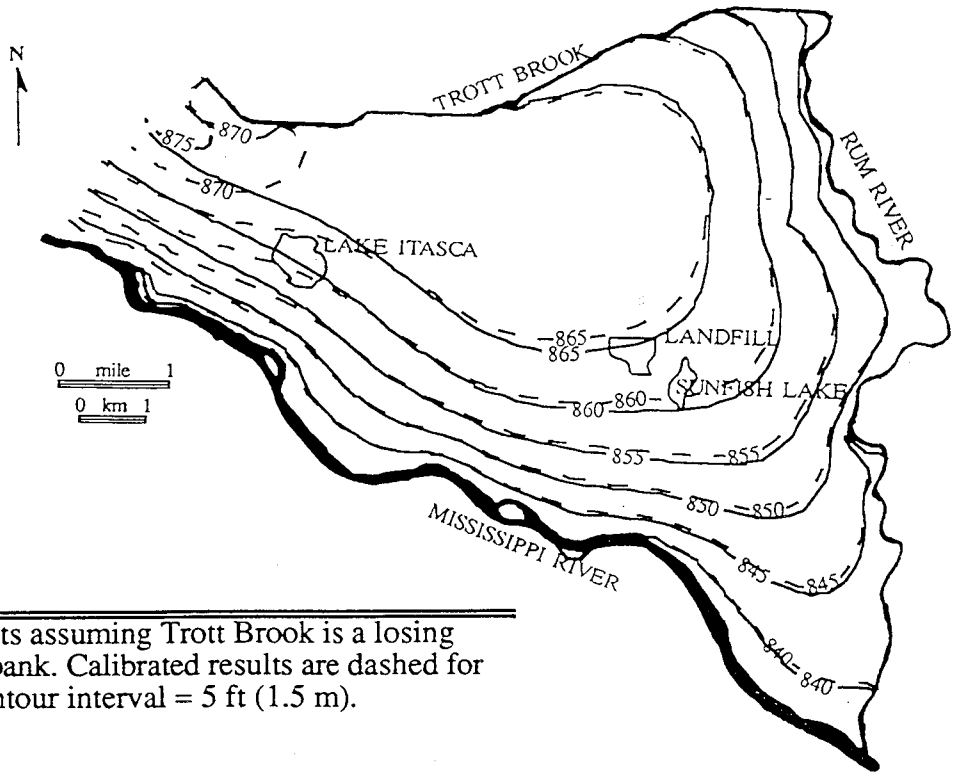



Figure 17 - Results assuming Trott Brook is a losing stream on south bank. Calibrated results are dashed for comparison. Contour interval = 5 ft (1.5 m).

The one-dimensional transient model specifically investigates the effect of instantaneous changes in river stage elevations of the Rum and Mississippi Rivers on the water table of the sandplain aquifer. The model is comprised of 22 cells that cross the southeast portion of the study area (Figure 13). Initial conditions were found by allowing a steady-state solution of the model (Equation 5) to converge using the calibrated conductivity and recharge and incorporating the varying aquifer base. Upgradient and downgradient flux are ignored, and recharge is held constant despite the instantaneous rise in the river stages. Rivers are assumed to be fully penetrating in the design of this model.

To simulate an instantaneous rise in river stages, the level of the Mississippi River was increased 2.0 ft (0.61 m), while the level of Trott Brook was increased 3.5 ft (1.1 m). The specific yield of the Anoka sandplain aquifer was set to 27%, an average percentage for coarse sand (Fetter, 1988). A transient solution is calculated for each time step using an explicit solution procedure. The results, shown graphically in Figure 18, suggest that a significant sudden change in the constant heads at the edges of the model has a negligible effect on the water table between the rivers, even over a period of 60 days. Because the effects of a storm or a snowmelt would end much sooner, any changes in the water table in the interfluvial region are expected to result from recharge events, and not from short-lived river stage variations. Based on this analysis, water table elevations predicted by the two-dimensional MODFLOW model are believed to be independent of transient river stages.

*Additional modeling efforts*

A test was performed to determine the effect of the kriged, varying elevation of the base of the aquifer. A simpler model would have utilized a rough average of the elevation of the sand/clay contact as the elevation of a horizontal planar aquifer base. The actual mean using all well logs of the database is 832 ft (254 m) above sea level. Using the same boundaries, recharge rate, and calibrated hydraulic conductivity, the contoured head values are markedly different when modeled using a planar aquifer base (Figure 19). Modeling with the kriged, irregular base is expected to produce a more accurate representation of the flow system.

The possibility of an effect on the surficial aquifer flow system by the pumping of residential wells was explored in an additional model run. A water usage of 100 gallons (380 L) per person per day, with three persons per household, was assumed for the 30 homes in the Hunters Hills subdivision, just south of Sunfish Lake. Although these wells are all finished in the permeable portion of the Franconia formation or in the Ironton formation, the pumpage of these wells was modeled as if they were screened in the surficial aquifer. The water table showed negligible effect from this worst-case scenario. Because essentially all private wells in the study area are deep bedrock wells, the effect of well pumpage is ignored. The city of Anoka has several municipal wells finished in bedrock. Only one is located in the study area, in the southeast tip. The others are located east of the Rum River. The effect these municipal wells may have on the surficial flow system is also assumed negligible.

***Advective transport modeling***

If any of the four initial monitoring wells were to detect contamination, then the monitoring well network would be expanded in order to assess the extent of the contamination. The purpose of this section is to quantitatively determine the configuration of the plume of a conservative contaminant tracer. Engineering database information on chloride is used as a means of checking the predicted results. The chloride distribution is examined in this study because it is a conservative tracer and therefore represents the furthest lateral extent expected for landfill leachate.

*Results*

MODPATH was used to determine flow paths and rates of flow of advective transport based on the calibrated MODFLOW output and an effective porosity. The average porosity of the surficial sand samples was determined in the lab to be 41% (Quinn, 1992). An effective porosity ($n_e$) of 35% is assumed in the modeling study. This value is consistent with other studies of

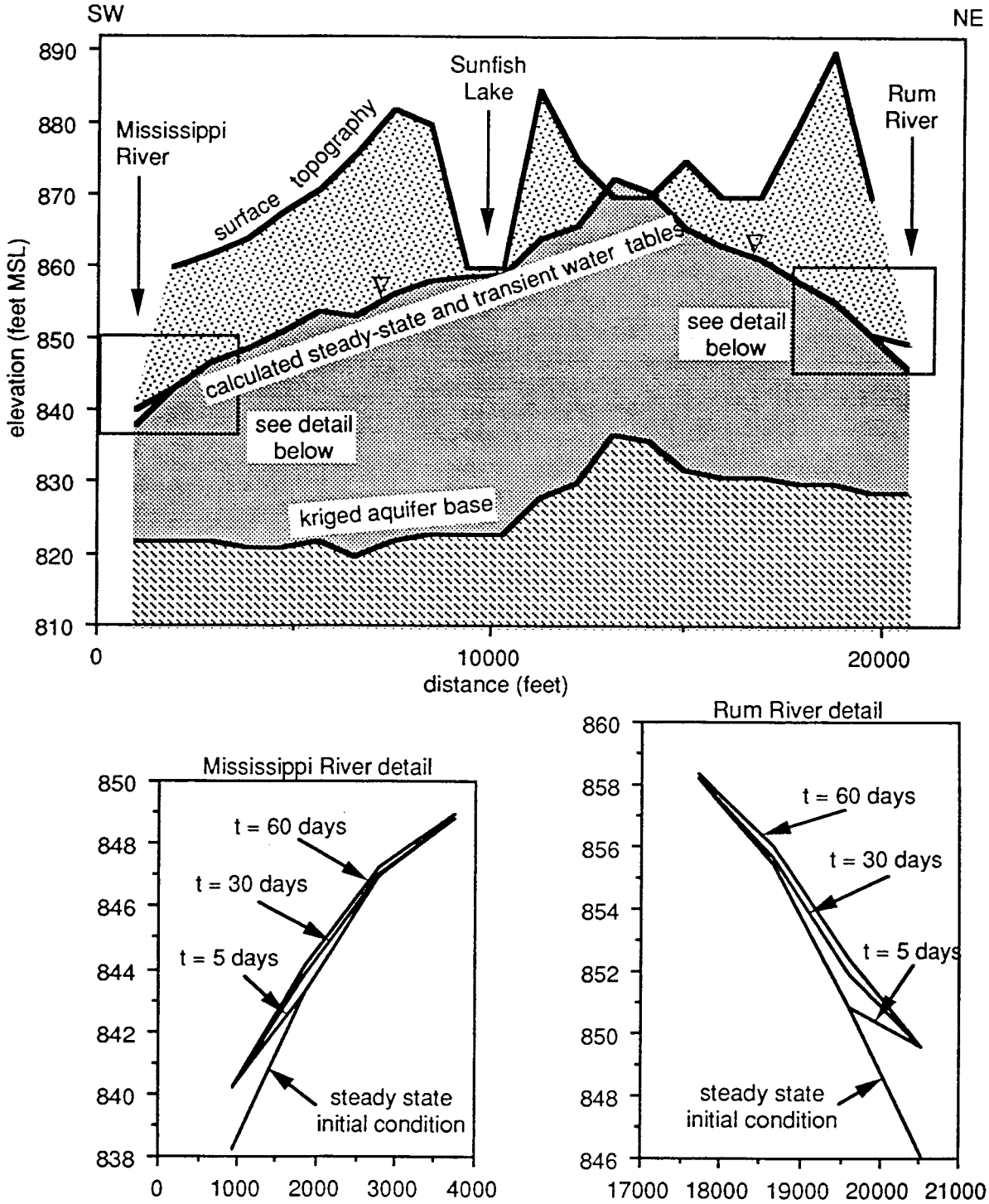



Figure 18 - Model results for transient simulation of river boundaries, including detailed views of calculated heads near Mississippi and Rum Rivers. Heads above land surface are due to simplicity of one-dimensional approximation.

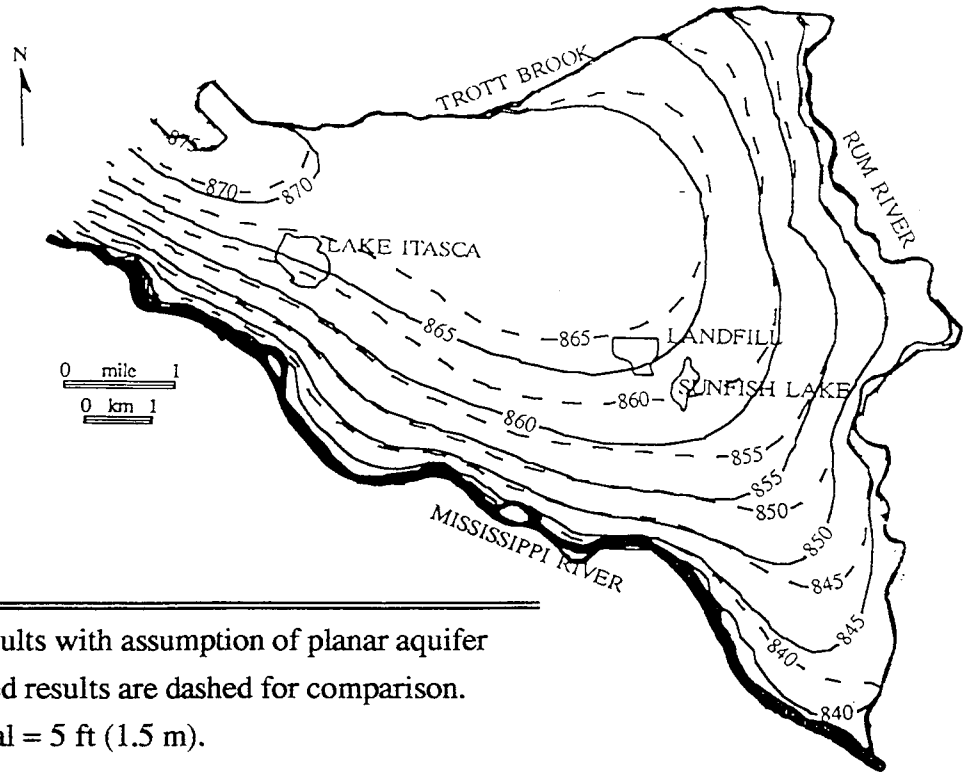



Figure 19 - Results with assumption of planar aquifer base. Calibrated results are dashed for comparison. Contour interval = 5 ft (1.5 m).

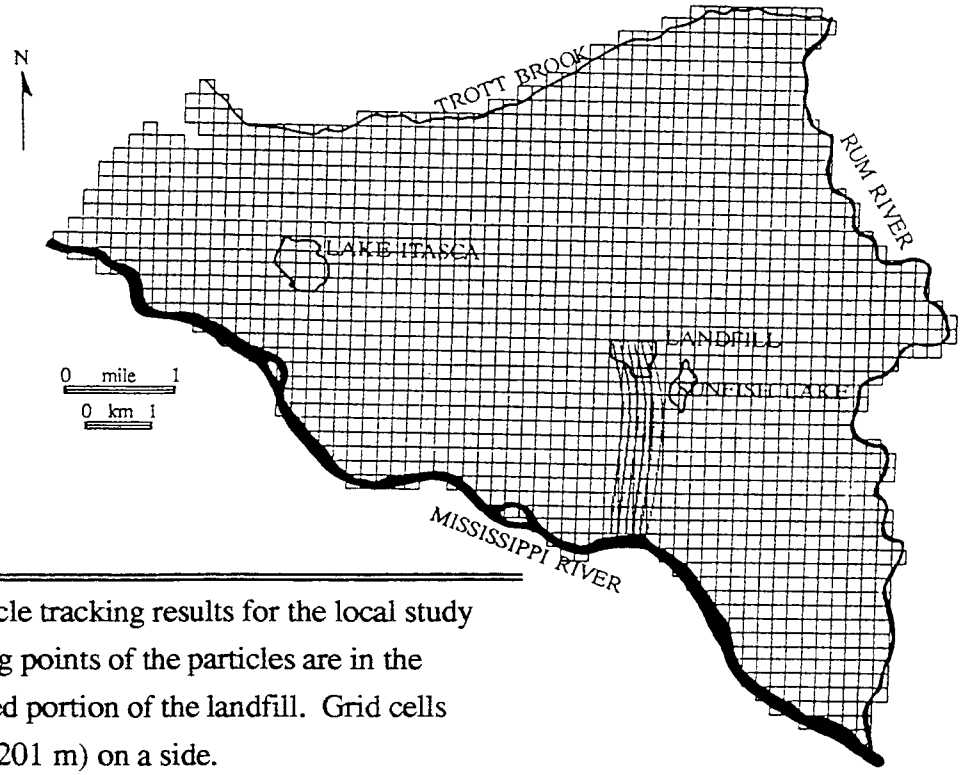



Figure 20 - Particle tracking results for the local study area. The starting points of the particles are in the earliest-developed portion of the landfill. Grid cells measure 660 ft (201 m) on a side.

outwash (*e.g.* Robertson and others, 1991; Mackay and others, 1986). The horizontal hydraulic gradient in the vicinity of the landfill is on the order of 0.0015.

In this study, the areal extent of the waste disposal area has changed a great deal over the years. However, sketches of the waste disposal area contained in Hickok and Foth & Van Dyke (1987) indicate that landfilling through April of 1969 was limited to the northern boundary of the present landfill, and the areal extent of the landfill changed little until after 1973. Eleven particles covering this area were therefore started near the northern border and tracked by MODPATH to their Mississippi River discharge points (Figure 20). The particles begin with a south-southeast trend and curve as they remain perpendicular to equipotentials. MODPATH allows a closer view of an area of interest (Figure 21).

*Sensitivity*

The sensitivity of the particle-tracking analysis was examined with respect to recharge, hydraulic conductivity, and effective porosity.

Changing R by ±10% produced negligible differences in the 5-year positions of tracked particles when compared to the calibrated model results. In the cases of varying K and $n_e$, the particle position changes were directly proportional to the percent change of either input parameter. This is expected simply because it is a simple application of Equation 1.

***Analysis of results***

As described previously, the extent of conservative chemicals from a waste source or tracer release may be expected to be a fraction of the distance predicted using various methods of hydraulic conductivity measurements. This section compares the MODPATH-predicted plume location with the actual chloride plume after approximately 20 years.

*The Anoka landfill chloride plume*

The chloride concentrations from April 1986 (Hickok and Foth & Van Dyke, 1987), March 1987 (Foth & Van Dyke, 1987), and December 1988 (Foth & Van Dyke, 1989b) show a fair amount of variation in comparisons of data from each monitoring well. However, the data do support generalized contours of chloride concentrations representing the conditions about 20 years after landfilling began. These data provide the basis for a comparison with the particle-tracking results.

The monitoring well network at the landfill utilizes a naming scheme to denote the relative depth at which the wells are screened. Water table wells are the A series or the AR series. Wells screened at the base of the surficial sand aquifer are the B series. Wells labeled with a BB are screened in a deeper sand which has at least some degree of hydraulic connection with the sandplain. These logs indicate an absence of Grantsburg till at some locations. C-series wells are finished in bedrock.

Background chloride concentrations to the north and northeast of the landfill are approximately 1 mg/l.

The three-dimensional character of the chloride plume is apparent in an inspection of data from various sampling depths. A-series wells show chloride impact in the range of 200 to 600 mg/l within about 200 ft (60 m) of the southern and eastern limits of the waste. Past this range, chloride is present only at background concentrations. This finding is consistent with other studies that found clean water above plumes because of the infiltration of recharge and/or downward flow of dense leachate (Kimmel and Braids, 1980; MacFarlane and others, 1983; Robertson and others, 1991; Mackay and others, 1986). B-series wells show chloride at more than 500 mg/l within 200 ft (60 m) of the waste (Figure 21), suggesting that leachate is driven downward to the base of the aquifer below the landfill itself. Within less than 1,500 ft (460 m) downgradient from the edge of the waste, the chloride at this level is at background concentration. The three-dimensional character of the chloride plume is shown by data from the BB level, where concentrations are in the 20 to 100 mg/l range beneath the southern end of Sunfish Lake because of stratigraphic control and continued downward flow of leachate. C-series wells have background chloride concentrations.

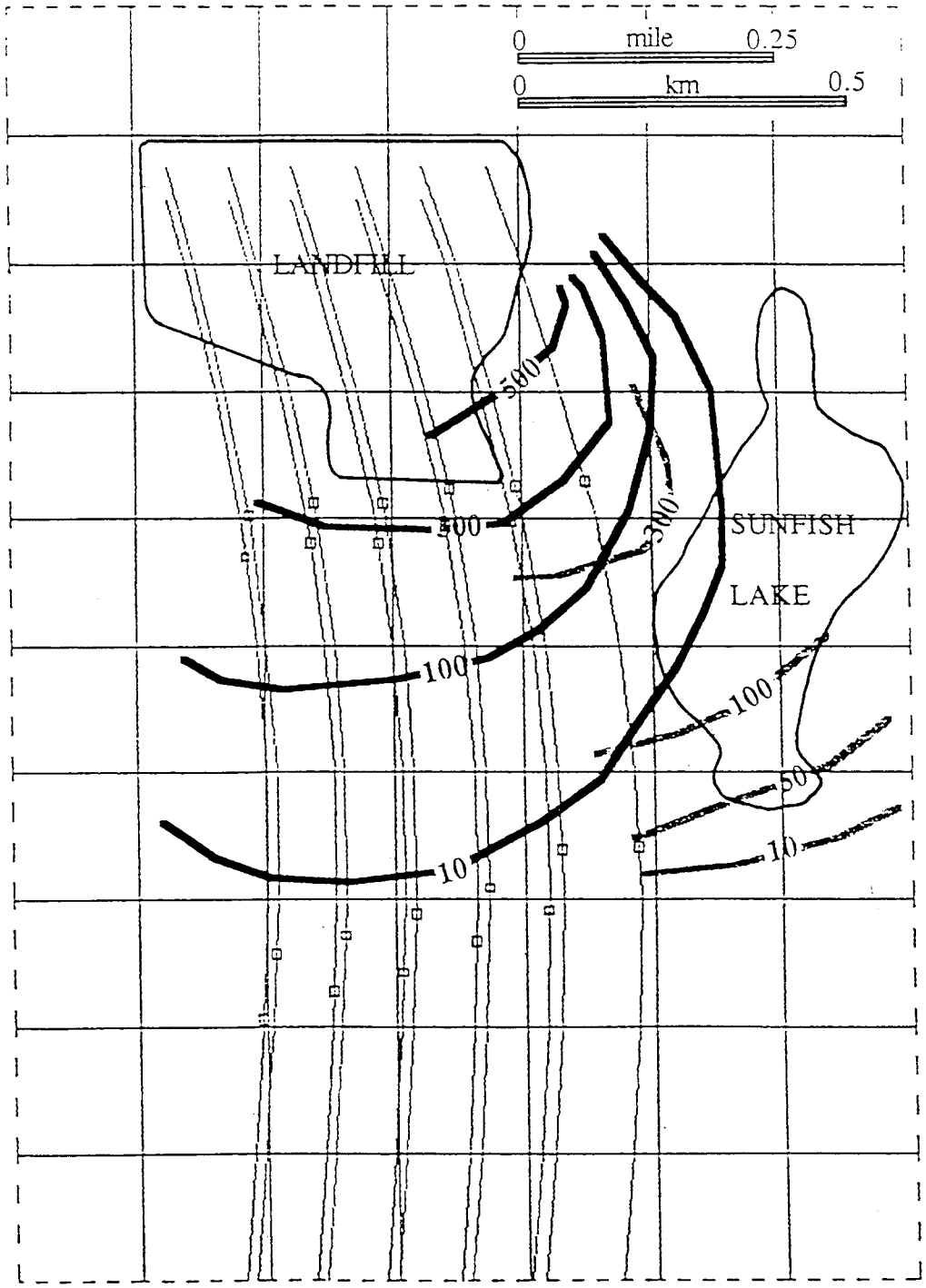



Figure 21 - Particle tracking results and actual 20-year chloride plume. Particles were started in the earliest waste disposal area. Boxes represent 5-year travel time locations. Black contours are chloride concentrations approximately at base of sandplain; gray contours are for a deeper, locally connected sand. Contour interval varies; units are mg/l. Grid cells measure 660 ft (201 m) per side.

*Discussion*

A comparison of the plume's extent with the temporal particle locations indicates a discrepancy of at least a factor of two, as the 20 year old plume coincides with the 10-year predicted plume (Figure 21). Several possibilities may explain the shorter-than-expected plume. The amount of time required for the initial generation of leachate is uncertain but probably not instantaneous. Also unknown is the time of travel in the unsaturated zone, which is composed of natural material and waste. Kimmel and Braids (1980) suggest that four years are required for the formation of leachate and its passage through the unsaturated zone; however, this estimate would certainly be subject to climate effects, waste characteristics, and the thickness and properties of the unsaturated media. In the saturated zone, the downward component of leachate flow increases the length of the flowpath and therefore decreases the extent of a plume shown in plan view. At the Anoka landfill, however, this is a fairly insignificant factor when the aquifer thickness is compared to the plume's extent. For this reason, the two-dimensional (one layer) modeling assumption is believed to be reasonable. Another possibility is that heterogeneities within the outwash may slow the flow of leachate (MacFarlane and others, 1983).

The history of the contamination source of this study creates additional complicating factors related to the distribution of chloride concentrations 20 years after landfilling commenced. The two main aspects are the longitudinal dispersion of the aquifer and the changing contaminant source. Certainly the chloride along the leading edge of the plume is affected by dispersion; that is, it has traveled faster than the average linear groundwater flow velocity. In late 1967, landfill operations began, but the areal extent, thickness, and composition of the waste has changed a great deal, especially in the operation's earlier years. The source concentration therefore has continually increased, as did the surface area of the aquifer receiving percolating leachate. Because the landfilled waste has grown in both area and thickness since waste disposal began, the concentrations and release areas of contaminant tracers have changed significantly, and the value of longitudinal dispersion cannot be estimated. However, the fact that the waste disposal area spread in the direction of groundwater flow, together with the likelihood of the effect of longitudinal dispersion, suggest that the plume discrepancy may be as great as a factor of three or four. If a source is continuous and has a constant area, then the longitudinal dispersion of the aquifer may be more easily addressed. Because of its history, the Anoka landfill does not lend itself to such a study, and because of these complicating factors, the times required for leachate generation and unsaturated zone transport cannot be easily or accurately quantified at the Anoka landfill.

Evidence of minimal lateral dispersion is present along the northeast edge of the plume. This lack of lateral dispersion is in agreement with numerous other studies of contaminants or other tracers in sandy aquifers (*e.g.* Robertson and others, 1991; Kimmel and Braids, 1980, 1974; Mackay and others, 1986; MacFarlane and others, 1983). Probable septic influence from nearby homes muddles the contours along the western edge of the plume (Hickok and Foth & Van Dyke, 1987). Although the monitoring well network at the Anoka landfill is not as dense as at the other study areas mentioned above, the effect of dispersion in the vertical direction appears to be negligible.

The orientation of the actual chloride plume differs only slightly from the path predicted by MODPATH (Figure 21). The reason is attributed to a combination of localized effects on the flow field caused by groundwater interaction with Sunfish Lake with a minor influence from values assigned along specified-head boundaries. However, given the position of the landfill on an assumed flow divide near the confluence of two major rivers, the flow of leachate could have been thought to have a flow direction ranging anywhere between due south and due east. This application of MODPATH is therefore believed to have achieved a fairly accurate prediction of flow direction.

*Selection of additional monitoring well locations*

If the four selected monitoring wells were the only available data for the Anoka landfill, then no statements could be made regarding the three-dimensional extents of various contaminant plumes. The placement of additional wells at this site, however, could have been guided by an application of the methodology presented in this paper.

When locating wells designed to determine the downgradient extent of a contaminant plume, the numerical results for the predicted plume should be assumed to be roughly two to four times the actual extent. Literature examples of plume studies describe negligible lateral dispersion, as is the case at the Anoka landfill. The lateral distribution of wells may therefore be appropriately constrained.

An understanding of expected plume movement can aid in augmenting a monitoring well network. But at the Anoka landfill, should the network have grown to approximately one hundred sampling points? A primary reason for increasing the network is to address those contaminants that behave different than a conservative solute such as chloride. The chloride plume is important because contained within its areal extent are the plumes of chemicals influenced by sorption, precipitation, biodegradation, oxidation-reduction, and other processes. Detection of light or dense non-aqueous phase liquids would require wells finished at the water table or at the base of the aquifer, respectively, in order to monitor the movement of those types of contaminants.

In the end, the size of a monitoring network may be strongly influenced by public or regulatory pressure, with a goal of providing a factor of safety on the estimation of the distribution of released chemicals. Even in this situation, a scientific approach has merit in preventing haphazard placement of monitoring wells.

The hydrogeologic environment is typically characterized by a high degree of uncertainty, which makes engineering decisions difficult (Freeze and others, 1990; Massmann and others, 1991). In this study, the extent of conservative contaminants was estimated without using any of the abundant amount of site-specific data, except for the heads of four water-table wells. Stratigraphic data from the monitoring network suggest a deep sand channel with roughly the same trend and location as Sunfish Lake. This feature has been termed a tunnel valley / esker complex by Hickok and Foth & Van Dyke (1987); however, this explanation is not consistent with the morphology or locations of tunnel valleys in the region, as presented in Figure 6 or by Quinn (1992). Also, the stratigraphic data from the private wells of the Hunters Hills subdivision south of Sunfish Lake do not indicate a deep surficial sand. How an apparent sand-filled channel fits into the scheme of the glacial history of the area is open to some debate; it may be a paleodrainageway associated with the Grantsburg sublobe. In a geologic framework, where the unexpected is usually expected, such a feature can change the design of a monitoring well network, and it exemplifies how a facies model may suddenly require revision.

## SUMMARY

### *Summary, conclusions, and applications*

This study illustrates the value of detailed surficial geologic mapping in a regional framework. The mapping of the glacial deposits was performed primarily with low-altitude stereo air-photos and soil surveys. Although the mapping is reconnaissance level information, valuable information was obtained for the local study area. Hydrogeologic boundaries and observation points were justified because rivers and lakes in the entire local study area were determined to be situated in permeable basins. Uniform recharge in the modeling area was assumed because of the similar geologic units covering the surface.

The two-dimensional geostatistical analysis of the basal elevation of the surficial aquifer exemplifies how abundant, inexpensive information in the form of well logs may refine both a geologic conceptual model and a numerical groundwater flow model. In this case, the discovered anisotropy of the surficial aquifer is attributed to south-southeasterly flowing meltwater that deposited the sandplain.

The spatially-varying sand/till contact was shown to exert significant control on the flow system when compared to the case of an assumed planar aquifer base. This two-dimensional technique has other applications in numerical modeling. For example, the upper and lower elevations of a confined aquifer could both be kriged in order to determine the geostatistical best estimates for constraining the thickness of the aquifer in each grid cell. Or, the likelihood of existence of a discontinuous confining unit may be analyzed with two-dimensional indicator geostatistics (Ritzi, 1991).

The hydrogeologic environment of the local study area is believed to allow fairly accurate modeling in two dimensions with a minimum of errors resulting from assumptions related to boundary conditions or improper geologic interpretation. Initial hydraulic conductivity estimates for the sandplain were determined from grain size analyses to equal approximately $1.2 \times 10^{-3}$ ft/s ($3.5 \times 10^{-2}$ cm/s). The effective hydraulic conductivity of the steady-state two-dimensional model was calibrated to $2.7 \times 10^{-3}$ ft/s ($8.2 \times 10^{-2}$ cm/s) using a literature-based recharge value of 11 in/yr (28 cm/yr). The calibrated conductivity agrees well with values derived from pump tests at the landfill. The model was found to be sensitive to changes in both hydraulic conductivity and recharge. One-dimensional analyses of the boundary conditions support the validation of the model by indicating that 1) the assumed flow conditions in the northwest corner of the study area are possible, and 2) the model is insensitive to short-lived changes in river stages.

The groundwater flow model could have application in modeling the effects of pumping, if such remediation took place at the landfill. Because the present grid discretization would not provide enough resolution for near-site pumping effects, a variable grid would have to be designed, with smaller finite difference cells near the landfill. The boundary conditions of the present model should suffice, although the effects of boundary conditions are tested best when the system is stressed (Franke and Reilly, 1987). If the cone of depression were to approach a specified-head river boundary, the river leakage package of MODFLOW would need to be implemented.

The actual chloride plume was found to be at least two times shorter than the predicted, particle-tracking plume. This discrepancy is consistent with other examples in the literature (*e.g.* Kimmel and Braids, 1980; MacFarlane and others, 1983; Robertson and others, 1991). Reasons for the difference may include delayed leachate generation and the leachate's time of travel through the unsaturated zone. The data at this site are complicated because the composition and thickness of waste has changed through the years and the area of landfilled waste has spread in the direction of groundwater flow. The effect of longitudinal dispersion on the flow of contaminants cannot be estimated because of the areally and temporally changing source.

The comparison of the actual chloride plume geometry versus its theoretical extent as determined by a particle-tracking analysis is applicable to many hydrogeological studies. This discrepancy is not mentioned in most texts or papers. In the investigations of Kimmel and Braids (1980), MacFarlane and others (1983), and Robertson and others (1991), hydraulic conductivity

values were obtained using various methods. These case studies suggest the inadequacy of simply using "average" hydraulic conductivity values for determining the transport of conservative tracers. Knowledge of such discrepancies has application in remediation investigations, wellhead protection studies, and other hydrogeological projects.

In this investigation, inexpensive public-domain geologic and hydrologic information produced a flow model that was calibrated with only four water table monitoring wells. The particle-tracking results matched the actual orientation of the plume quite closely. If only these four monitoring wells had existed at this landfill, the locations of additional wells could have been based on the predictive output of this modeling, with modification based on the common discrepancy between actual and predicted plume extents.

The findings of this study can serve as a general basis for the design of a three-dimensional monitoring well network for an existing or proposed potential contamination source. Because the predicted plume of a conservative contaminant tracer is typically two to four times greater than its actual extent, downgradient wells may be appropriately located. The depth at which the wells are screened may be guided by the concept of downward-driven solutes below and near the waste source, and an absence of solute contaminants at or near the water-table past a relatively short distance downgradient. The planning of other monitoring well networks, especially those in similar hydrogeologic settings, could benefit from the findings of case studies such as this.

Because this study is not a detailed analysis of the fate and transport characteristics of landfill leachate, the plumes of other contaminants have not been discussed. By examining the chloride data after twenty years of landfill operations, the inconsistency of predicted vs. actual plumes was addressed for a conservative pollutant tracer. The chloride plume should represent the furthest areal extent of contaminants from such a source. Although the spread of contaminants with different physical properties should be contained within the plan view of the chloride plume, relatively deeper or shallower monitoring wells would be necessary to monitor non-aqueous phase liquids. Ultimately, a monitoring well network may be augmented as a result of regulatory agencies and citizens groups desiring greater confidence on the understanding of contaminant distributions. In this case the placement of additional wells should be based on a scientific methodology that incorporates all available geological and hydrological information.

### *Suggestions for further research*

Other case studies of the relation of monitoring well networks with predicted and actual plumes would serve to refine the modeling approach used in this paper. Studies in different hydrogeological settings would provide the best comparisons. A highly detailed three-dimensional groundwater modeling exercise could rely on hydraulic conductivity values assigned to cells according to a three-dimensional geostatistical characterization, as performed by Johnson and Dreiss (1989) for alluvial sediments and proposed by Quinn and others (1989) for glacial-drift aquifers. In the present study, however, the hydrogeologic environment and boundary conditions were conducive to modeling only a surficial aquifer in two dimensions. Other study areas with reasonable three-dimensional boundary conditions may allow the opportunity to incorporate such detail, although in a glacial geologic framework both the model construction and the computational effort may be prohibitive.

This study benefitted from a literature-derived recharge rate which could be applied uniformly over the modeling area because of the similar outwash-derived soil throughout. Modeling in other glaciated locations in Minnesota and elsewhere could require an estimate of the recharge rate through surficial till. Daniels and others (1991) performed tritium analyses to determine a recharge rate through an Indiana till, but Helgesen and Lindholm (1977) is one of few recharge studies in Minnesota. Investigations of the recharge and fracture characteristics of Minnesota tills would aid in the construction of groundwater models ranging in scale from site-specific to regional, so that recharge input would be justified by a detailed independent analysis, rather than estimated by mere model calibration exercises.

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