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A Finite Element Model of Groundwater Contamination at Landfill Site

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The quantitative study of the groundwater contamination in a porous media is a difficult task. For complex problems, numerical solutions are the most effective means to study the movement of contaminants in the groundwater. The solute transport model used in this study has proved to be an efficient tool to model contaminant transport for complex problems. The model demonstrates its effectiveness in reproducing the contamination by chlorides of the groundwater at the landfill site due to leachate from the wastes. It describes the two dimentional solute transport and alteration of the water quality and forecasts the contamination for different management alternatives of the landfill. The model also indicates how the groundwater contamination can be contained within the Lowry site if a barrier is constructed downstream of the disposed wastes.

Key words : Groundwater contamination, Landfill, Leachate, Groundwater quality, Finite element model, Hazardous waste

1. INTRODUCTION

In parallel with the progress of industrial societies the amount and toxicity of associated wastes has been increasing. One of the major problems that has arises is the storage of these wastes in a safe place where they are not a threat to environmental quality. The disposal of hazardous wastes has been carried out in many places whose characteristics are not the optimal ones and groundwater pollution has been the result. Numerical models have been developed to evaluate the influence on the water quality of this kinds of practices, either to anticipate the effects of future disposal activities or to determine the extent of the contamination in an actual case and to find the best solutions to clean up the site. A finite element model with triangular elements and linear shape functions written and programmed by J. W. Warner (1991) was used to simulate two-dimensional solute transport in groundwater flow. The purpose of this study is to determine the effectiveness of the model in its application to a real case study involving groundwater contamination at the Lowry landfill site.

2. DESCRIPTION OF THE STUDY AREA

2.1 Geology and Aquifers

The Lowry landfill is located slightly east of the axis of a broad north south trending regional syncine known as the Denver Basin (Fig. 1). Ground water occurs in the alluvium deposited along the stream channels. Where saturated it is the uppermost aquifer.

2.2 Basic Data

The basic data on hydrology property and groundwater levels of the aquifers in the study area are from the reports of USGS and Golder Associates. (Robson, 1977 and Golder Associates, 1982). Also groundwater quality and other miscellaneous data gained by D. Fontane (1996) of Colorado State University from the Colorado Health Department.

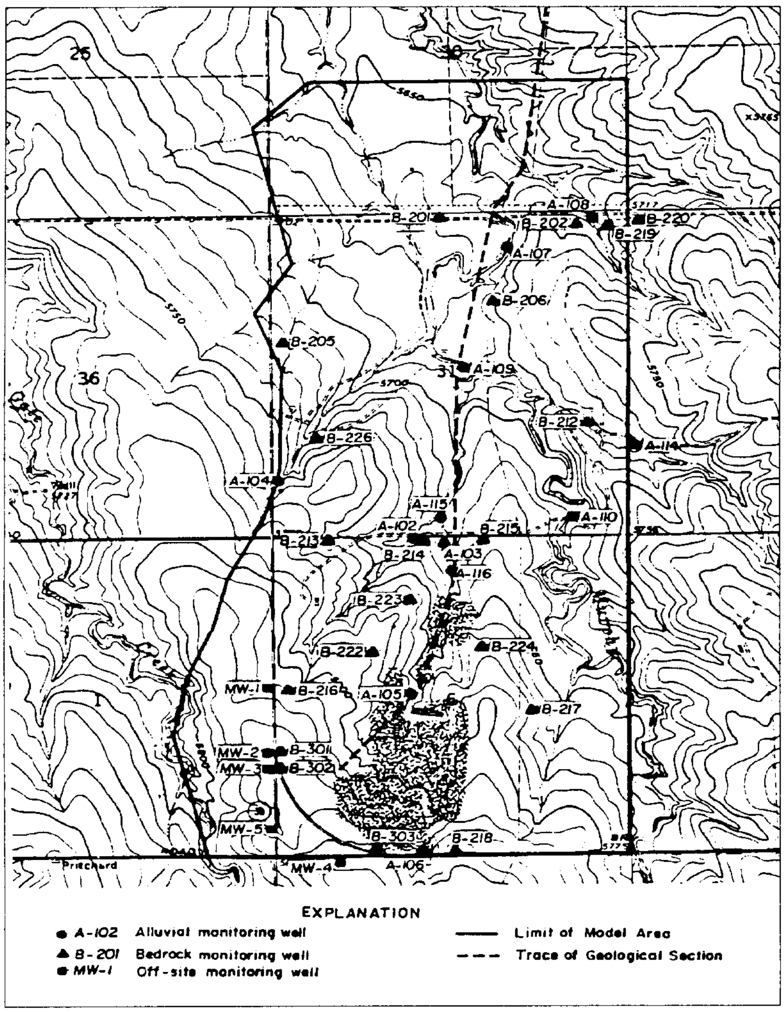

Fig. 1 Site map and monitoring well location.

2.2.1 Hydraulic Conductivity and Porosity

According to Robson the hydraulic conductivity for 10 samples from alluvium within 3.22 to 14.48 km from the Lowry landfill ranged from 2.4×10^{-3} to 39.62 m/day with a mean of 9.75 m/day. The hydraulic conductivity of six samples taken from the upper part of the bedrock formations within 1.61 to 12.87 km from the Lowry landfill ranged from 2.87×10^{-5} to 2.65 m/day with a mean of 0.85 m/day. In 1978 Daley and Aldrich carried out in-situ percolation tests and laboratory tests at a proposed salt-brine disposal pond situated in the northeast of Section 31. They found that the uppermost few feet of weathered rock had hydraulic conductivity in the 10^{-4} to 10^{-5} cm/sec range and the unweathered bedrock was expected to be in the range of 10^{-6} to 10^{-8} cm/sec. In 1982 Golder Associates performed a study to evaluate the feasibility of constructing a waste containment structure to intercept contaminated ground water flowing north in the alluvial channel from the center of Section 6. The containment site is situated in the south part of Section 31. Field permeability tests yielded a hydraulic conductivity for the surficial soils of from 5.3×10^{-5} to 1×10^{-4} cm/sec, for the highly weathered sandstone of the Dawson formation of from 3.2 to 9.9×10^{-4} cm/sec and for the weathered claystone and siltstone of the same Dawson formation of from 1.7×10^{-7} to 4.0×10^{-5} cm/sec.

2.2.2 Groundwater Levels

The water level altitude measured in the alluvial and shallow bedrock wells are very close. The deeper bedrock wells have lower water levels. In general, the deeper the well the lower the water level altitude. Indications are that the alluvium and the upper bedrock aquifer are hydrologically connected. The chemical analysis of the samples of up and down gradient wells taken at different depths seem to indicate that the water movement is rather lateral than vertical and that leachate remains in the upper aquifers. There are no wells within the disposal area but the presence of mounds can be asserted in local areas due to recharge from ponds. Gas monitoring wells, drilled only through the refuse with very small depth give evidence of the mounds in the southeast area of Section 6.

2.2.3 Water Quality

The chemical analyses indicated high concentrations of many of the substances analyzed for and very high concentrations for some of the contaminants. Chloride is an often used tracer to assess the spread of groundwater contamination. The chloride ion is conservative, that is it does not react with the solid aquifer material allowing it a high mobility. Other constituents, such as phenols and metals, have their movement inhibited by adsorption and are not as good of indicators as the chloride ion of the possible extent of the groundwater degradation. The analysis for chlorides carried out are shown in Table 1.

3. NUMERICAL MODEL FORMULATION

3.1 Description

The numerical model used to study groundwater contamination was a finite element two-dimensional solute transport model. The model was developed by J. Warner (1991) and it was applied in this present study with some modifications to fit the conditions in this study area. The program based on

Table 1. Chloride Concentration in Milligrams per Liter at Selected Wells

Well site	April 1981	January 1982	April 1991	January 1992
Alluvial Wells				
A-102	89.5	118		
A-103		127	823	835
A-105	8,975.0	9,200	9,650	9,870
A-107			42	48
A-109	18.9	26	96	98
A-110	16.9	24	32	36
A-115	142	168	860	905
A-116	346	436	1,020	1,085
B-210	27.8	31		
B-202A		33		
B-205	403	385		
B-206	140	177		
B-212	48.6	39		
B-213	31	30		
B-214	108	129	380	398
B-215	33	42		

the Galerkin method solves simultaneously the Boussinesq equation for a non-homogeneous isotropic medium describing groundwater flow and the equation describing two-dimensional mass transport for a reacting solute subject to adsorption in flowing groundwater.

3.2 Basic Assumptions

In the model, the hydraulic conductivity and thickness of the aquifer are a variable between elements but are a constant for each individual element. The porosity is assumed uniform in space and time for all elements. The variations in density on concentrations are not taken into account. The concentrations used in the program are the average vertical concentration in the aquifer. The model can simulate either transient flow and transient transport or steady state flow and transient solute transport. Steady state flow was used in the modeling. A review of the water level data indicated no major fluctuations in the water level elevations in any of the observation wells during the monitoring period. Therefore, recharge from infiltration of precipitation and discharge from evapotranspiration were assumed to be small and were neglected in the model. The waste disposal ponds are the sources of pollution. The available data indicate that migration of contaminants from the waste disposal ponds have affected the shallow and upper bedrock aquifer but not the deeper aquifers, that is, no leachate has occurred to those deeper aquifers. The program makes use of a mass lumping procedure.

3.3 Model Formulation by Galerkin Finite Element Method

3.3.1 Theoretical Equations

1) Flow Equation

The equation describing the transient two-dimensional flow of a homogeneous fluid through a non-homogeneous anisotropic saturated aquifer can be written as

$$\frac{\partial}{\partial x}\left(T_x \frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(T_y \frac{\partial h}{\partial y}\right)=S\frac{\partial h}{\partial t} +W+\sum_{p=1}^{m}(\delta(x-x_p)\cdot\delta(y-y_p)\cdot Q_p) \qquad (1)$$

where

$T_x=T_x(x, y)$ = transmissivity in the x-direction (L^2/T)

$T_y=Ty(x, y)$ = transmissivity in the y-direction (L^2/T)

$h=h(x, y, t)$ = potentiometric head (L)

$S=S(x, y)$ = storage coefficient (dimensionless)

$W=W(x, y, t)$ = distributed volumetric water flux per unit area, positive sign for discharge and negative sign for recharge (L/T)

$Q_p=Q_p(t)$ = volumetric water flux at a point located at (xp, yp) positive sign for withdrawal and negative sign for recharge (L^3/T)

$\delta(x-\xi)$ = Dirac delta function defined as

$\delta(x-\xi)=0$ if $x\neq\xi$ and

$\int_{\xi-\varepsilon}^{\xi+\varepsilon}\delta(x-\xi)\ f(x)\ d_x=f(\xi)$

t = time (T)

x, y = cartesian coordinates in the principal direction of transissivity.

2) Solute Transport Equation

The equation used to describe the two dimensional mass transport for a reacting solute subject to adsorption in flowing groundwater is derived from the principal of conservation of mass and can be expressed as (Warner, 1991)

$$-\frac{\partial C}{\partial t}-\frac{\partial \bar{C}}{\partial t}=\frac{\partial}{\partial x}(C\cdot V_x)+\frac{\partial}{\partial y}(C\cdot V_y) -\frac{\partial}{\partial x}\left(D_{xx}\frac{\partial C}{\partial x}\right)-\frac{\partial}{\partial y}\left(D_{yy}\frac{\partial C}{\partial y}\right) \qquad (2)$$

$$-\frac{\partial}{\partial x}\left(D_{xy}\frac{\partial C}{\partial y}\right)-\frac{\partial}{\partial y}\left(D_{yx}\frac{\partial C}{\partial x}\right)+\frac{WC'}{\varepsilon\cdot b} +\sum_{p=1}^{m}\left(\delta(x-x_p)\cdot\delta(y-y_p)\frac{Q_p\cdot C'}{\varepsilon\cdot b}\right)$$

where

$C=C(x, y, t)$ = dissolved concentration of the solute (M/L^3)

$\bar{C}=\bar{C}(x, y, t)$ = concentration adsorbed on the solid aquifer material per volume of solution (M/L^3)

$C'=C'(x, y, t)$ = dissolved concentration of the solute in the source or sink fluid (M/L^3)

$V_x=V_x(x, y, t)$ = average interstitial velocity on the groundwater in the x-direction (L/T)

$V_y=V_y(x, y, t)$ = average interstitial velocity on the groundwater in the y-direction (L/T)

$D_{ij}=D_{ij}(x, y, t)$ = coefficient of hydrodynamic dispersion (second-order tensor, L^2/T)

$\varepsilon=\varepsilon(x, y, t)$ = porosity of the aquifer (dimensionless)

$b=b(x, y, t)$ = saturated thickness (L)

The adsorption process in the equation (2) is quantified using the Freundlich Isotherm empirical equation because of its mathematical simplicity. According to the equation, the adsorption of ions in solution by a solid is given by (Freeze and Cherry, 1979)

$$\bar{C}=K_d\cdot C^{\alpha} \qquad (3)$$

Where $\bar{C}$ is the adsorbed concentration, C is the dissolved concentration and K_d (distribution coefficient) and α are constants experimentally determined. The time derivative of $\bar{C}$ after (3) yields

$$\frac{\partial\bar{C}}{\partial t}=K_d\cdot\alpha C^{\alpha-1}\frac{\partial C}{\partial t} \qquad (4)$$

And the first term of (2) can be written as

$$-\frac{\partial C}{\partial t} - \frac{\partial \bar{C}}{\partial t} = -\frac{\partial C}{\partial t}(1+K_d \cdot \alpha \cdot C^{\alpha-1}) \quad (5)$$

With the substitution of (5) into (2) the latter becomes

$$-\frac{\partial C}{\partial t} = \frac{\partial}{\partial x}\left(C\frac{V_x}{R_d}\right) + \frac{\partial}{\partial y}\left(C\frac{V_y}{R_d}\right) - \frac{\partial}{\partial x}\left(\frac{D_{xx}}{R_d}\frac{\partial C}{\partial x}\right) - \frac{\partial}{\partial y}\left(\frac{D_{yy}}{R_d}\frac{\partial C}{\partial y}\right) - \frac{\partial}{\partial x}\left(\frac{D_{xy}}{R_d}\frac{\partial C}{\partial y}\right) - \frac{\partial}{\partial y}\left(\frac{D_{yx}}{R_d}\frac{\partial C}{\partial x}\right) + \frac{WC'}{\varepsilon \cdot b} + \sum_{p=1}^{m}(\delta(x-x_p)\,\delta(y-y_p)\,Q_p) \quad (6)$$

$$\text{where } R_d = 1 + K_d \cdot \alpha \cdot C^{\alpha-1} \quad (7)$$

R_d is called the retardation factor.

3.3.2 Galerkin Approximation

Consider an equation, $L(u)=0$ on the bounded domain D (8)

where L is a linear differential operator and u an unknown variable. Assume a trial solution

$\hat{U}(x, y, t)$ expressed as

$$\hat{U}(x, y, t) = \sum_{j=1}^{n}(G_j(t) \cdot \phi_j(x, y)) \quad (9)$$

ϕ is a set of independent shape functions, called basis functions, chosen to satisfy the boundary conditions and Gj are the unknown time dependent coefficients which represent solutions of the equation (8) at specified nodes within the solution domain.

Let a residual R be defined by

$$R(x, y, t) = L(\hat{U}) = L\left[\sum_{j=1}^{n} G_j(t) \cdot \phi_j(x, y)\right] \quad (10)$$

Local linear shape functions (V^e) are used because of the conjunction between computational effort and the reliability of the results. The approximating trial solution (9) is rewritten for the element e as

$$\hat{U}(x, y, t) = G_i(t) \cdot V_i^e(x, y) + G_j(t) \cdot V_j^e(x, y) + G_k(t) \cdot V_k^e(x, y) \quad (11)$$

The global shape function ϕ_i is the union of all the local shape functions that are non zero at node i. It is non zero only over elements which have node i as a vertice, equal to zero at all the other nodes. The value of the approximating trial solution at node i is

$$\hat{U}(x_i, y_i, t) = G_i(t) \cdot \phi_i(x_i, y_i) = G_i(t) \quad (12)$$

3.3.3 Solution to the Groundwater Flow Equation

1) Galerkin Approximation

The linear differential operator corresponding to equation (1) is defined as

$$L(h) = \frac{\partial}{\partial x}\left(T_x\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(T_y\frac{\partial h}{\partial y}\right) - S\frac{\partial h}{\partial t} - W - \sum_{p=1}^{m} Q_p \cdot \delta(x-x_p) \cdot \delta(y-y_p) = 0 \quad (13)$$

with the boundary conditions

$$\left.\frac{\partial h}{\partial x}\right|_B = \text{constant on boundary B}$$
$$\left.\frac{\partial h}{\partial y}\right|_B = \text{constant on boundary B} \quad (14)$$

These conditions represent constant gradient boundaries. Assuming a trial solution given as

$$h(x, y, t) \simeq \hat{h}(x, y, t) = \sum_{j=1}^{n}(G_j(t)\,\phi_j(x, y)) \quad (15)$$

and substituting it into the operator given by equation (13) the residual R (x, y, t) is obtained, and the approximating integral equations yield by setting the weighted integrals of the residual R(x, y, t) to be zero.

$$\iint_D \left\{ \frac{\partial}{\partial x} \left[T_x \frac{\partial}{\partial x} \left(\sum_{j=1}^{n} (G_j(t) \cdot \phi_j(x, y)) \right) \right] + \frac{\partial}{\partial y} \left[T_y \frac{\partial}{\partial y} \left(\sum_{j=1}^{n} G_j(t) \cdot \phi_j(x, y) \right) \right] - S \frac{\partial}{\partial t} \left[\sum_{j=1}^{n} G_j(t) \cdot \phi_j(x, y)) \right] - W - \sum_{p=1}^{m} (\delta(x-x_p) \cdot \delta(y-y_p) Q_p) \right\} \phi_i(x, y) dxdy = 0 \quad (16)$$

$$i = 1, 2, \cdots\cdots, n$$

Expanding and differentiating inside the summation brackets and applying Green's theorem and boundary conditions the n equations can be written in matrix form as

$$[A]\{G\} + [B]\left\{\frac{dG}{dt}\right\} + \{D\} + \{E\} + \{F\} = 0 \qquad (17)$$

where [A] and [B] are n x n dimensional matrices and {D}, {E}, {F}, {G} and $\left\{\frac{dG}{dt}\right\}$ are n dimensional vectors to solve the flow equation. The elements are

$$A_{ij} = \iint_D \left\{ T_x \frac{\partial \phi_i(x, y)}{\partial x} \cdot \frac{\partial \phi_j(x, y)}{\partial x} + T_y \frac{\partial \phi_i(x, y)}{\partial x} \cdot \frac{\partial \phi_j(x, y)}{\partial x} \right\} dA$$

$$B_{ij} = \iint_D \{S\phi_i(x, y)\, \phi_j(x, y)\}\, dA$$

$$D_i = \iint \{W\phi_i(x, y)\}\, dA$$

$$E_i = \iint_D \left\{ \phi_i(k, y) \left(\sum_{p=1}^{m} (\delta(x-x_p) \cdot \delta(y-y_p) Q_p) \right) \right\} dA$$

$$F_i = -\oint_B \left\{ T_x \phi_i(x, y) \left(\frac{\partial h}{\partial x} \Big|_B \right) l_x + T_y \phi_i(x, y) \left(\frac{\partial h}{\partial y} \Big|_B \right) l_y \right\} dL$$

2) Integrations

The integration of A_{ij}, B_{ij}, D_i, E_i and F_i is performed in a piecewise manner on an element basis. The global matrix for the entire domain is formed from these element matrices by summing for a given node the contribution of all the elements sharing that node.

3) Time Derivative Approximation

Once the matrices [A] and [B] and the vectors {D}, {E}, and {D} are evaluated, the technique for solving the n ordinary differential equations (17) is the implicit finite difference scheme to approximate the time derivative. The finite difference approximation for the time derivative is

$$\frac{dG}{dt} = \frac{G_{t+\Delta t} - G_t}{\Delta t} \qquad (18)$$

{Gt} are the known values of G at time t and $\{G_{t+\Delta t}\}$ are the calculated values at the time $t+\Delta t$. The substitution of (18) into (17) with an implicit scheme yields after rearranging the terms

$$\left([A] + \frac{1}{\Delta t}[B] \right) \{Gt+\Delta t\} = \frac{1}{\Delta t}[B]\{G_t\} - \{D\} - \{E\} - \{F\} \qquad (19)$$

The solution of $\{G_{t+\Delta t}\}$ can be found by matrix-solving techniques as Gaussian elimination or point-iterative successive overrelaxation technigue, both are used optionally in the program.

3.3.4 Solution to the Solute Transport Equation

1) Galerkin Approximation

The linear differential operator corresponding to equation (6) is defined as

$$L(C) = \frac{\partial}{\partial x}\left(\frac{D_{xx}}{R_d} \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial y}\left(\frac{D_{yy}}{R_d} \frac{\partial C}{\partial y} \right) + \frac{\partial}{\partial x}\left(\frac{D_{xy}}{R_d} \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial y}\left(\frac{D_{yx}}{R_d} \frac{\partial C}{\partial y} \right)$$

$$-\frac{\partial}{\partial x}\left(C\,\frac{V_x}{R_d}\right)-\frac{\partial}{\partial y}\left(C\,\frac{V_y}{R_d}\right)-\frac{\partial C}{\partial t} \qquad (20)$$

$$-\frac{WC'}{\varepsilon\cdot b}-\sum_{p=1}^{m}(\delta(x-x_p)\ \delta(y-y_p))\ \frac{Q_p\cdot C'}{\varepsilon\cdot b}$$

The approximating integral equations are given by (10). Combining these equations along with expansion and differentiation inside the summation brackets and the application of Green's theorem and boundary conditions, leads to n equations written in matrix form as

$$[A_s]\{G_s\}+[B_s]\left\{\frac{dG_s}{dt}\right\}+\{D_s\}+\{E_s\}+\{F_s\}=0 \quad (21)$$

in which [As] and [Bs] are n×n dimensional matrices and $\{D_s\}$, $\{E_s\}$, $\{F_s\}$, Gs and $\{\frac{dGs}{dt}\}$ are n dimensional vectors to solve the solute transfort equation.

2) Integrations

The integration of A_{sij}, B_{sij}, D_{si}, E_{si} and F_{si} is performed in a piecewise manner on an element basis in an analogous way to the integration of the flow equation.

3) Time Derivative Approximation

The time derivative of equation (21) is approximated by using a first order implicit finite-difference scheme as shown in (18). The substitution of equation (18) into (21) and the rearrangement yield

$$\left([A_s]+\frac{1}{\Delta t}[B_s]\right)\{Gs_{t+\Delta t}\}=\frac{1}{\Delta t}[B_s]\ \{G_{st}\}-\{D_s\}-\{E_s\}-\{F_s\} \quad (22)$$

Then, the equation can be solved for the unknown vector $\{Gs_{t+\Delta t}\}$

4. Model Simulations

4.1 Model Input Data

4.1.1 Grid, Boundary Conditions and Recharge

The grid used contained 247 elements and 142 nodes (Fig. 2). The model area was about 1670

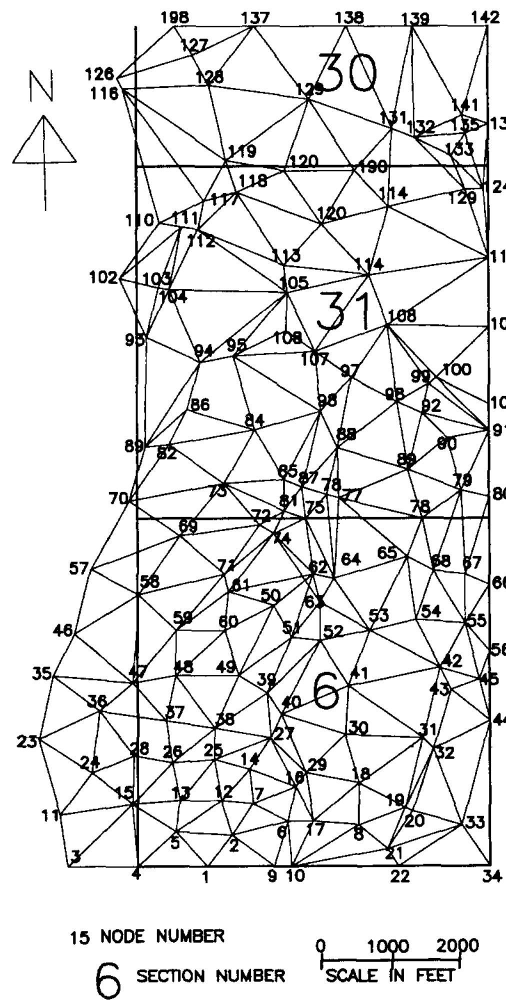

Fig. 2. Finite Element Grid for Study Area

acres situated in Section 6 and Section 31. The grid includes the area of waste disposal and the landfill area downgradient (to the north). The size of the elements are smallest near the waste disposal area. A larger element size is used for the surrounding model area. Constant head boundary nodes were specified all around the model area. No data were available on the flow rates to and from adjacent parts of the aquifer across the model boundaries. An infiltration of precipitation and evapotranspiration was thought to be small in the model area and was therefore neglected. The model is based on steady state flow. The amount of liquid wastes disposed of increased considerable over the period of operation of the landfill. The model is based on steady state flow. The average disposal rate of 6.7 million gallons per year was used in the groundwater flow part of the model. In the transport part of the model, mass slugs were introduced every year in proportion to the total amount of contaminant disposed of in that year. The location of the disposal ponds varied with time.

4.1.2 Transmissivity, Porosity and Storage Coefficient

The hydraulic conductivity of the Dawson formation used in the model was 1.3×10^{-4} cm/sec. There were four values of hydraulic conductivity for the alluvium used in the model: 8.3×10^{-3} cm/sec, 0.011 cm/sec, 0.015 cm/sec and 0.021 cm/sec. The higher values correspond to the coarser soils near Coal Creek and the lower values correspond to less altered alluvium. These values of hydraulic conductivity are based on the field data and on the model calibration process and are somewhat higher than the reported field data. The saturated thickness of the alluvium ranged from almost absent to a maximum of 6.71 m. The saturated thickness of the Dawson and Denver formation ranged from 1.83 to 10.97 m. With these values of permeability and saturated thickness the transmissivities range from 0.20 m^2/day to 1.19 m^2/day in the bedrock and from 3.49 m^2/day to 66.89 m^2/day in the alluvium. The model uses a single value of porosity for the entire model area. A porosity of 0.37 was used in the model, Robson (1977) and Golder Associates (1982). Since only steady flow was considered a storage coefficient of zero was used in the model.

4.1.3 Groundwater Movement

The observed groundwater levels in the uppermost aquifer are shown on Fig. 3. The direction

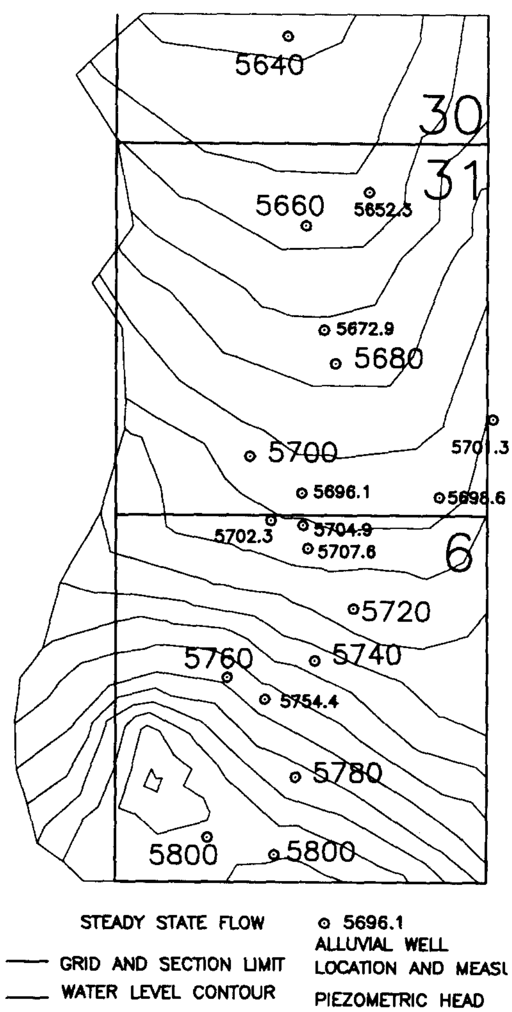

Fig. 3. Groundwater levels for Study Area.

of the flow is mainly to the north following the path of the alluvium channels. In the southwest part of Section 6 the presence of a mound is surmised. Its existence is due to the recharge from the waste disposal ponds and the low permeability of the bedrock. The groundwater velocity in the alluvium considering the permeability and porosity values assumed in above sections and the hydraulic gradient of the groundwater varies from nearly 0.76 m/day in Section 6 to about 0.30 m/day in Section 30 and upper part of Section 31.

4.1.4 Longitudinal, Transversal Dispersivities and Initial Concentrations

In the calibration process, a longitudinal dispersivity $\alpha_L = 30.48$ m gave the best fit to the data. The rate of transversal to longitudinal dispersivities is 0.30. These were within the range of values from dispersivity reported in the literature. Analysis of wells not affected by the waste disposal operations were used to determine the baseline concentration for chloride. Concentrations of chloride ranged from 15 to 35 mg/l and an average baseline concentration of 25 mg/l was used in the model.

4.2 Calibration of the Model

The calibration is a feed-back process that tries to match first the potentiometric heads followed by the fit of the concentration values. The model was initially run with a hydraulic conductivity value of 0.02 m/day for the bedrock and 0.46 m/day for the alluvium. These hydraulic conductivity values were similar to those reported in other studies. The final model calibration yielded perambulates of 0.11 m/day for the bedrock and from 7.16 to 18.29 m/day for the alluvium. No data were available on the chloride concentration of the liquid disposed of at

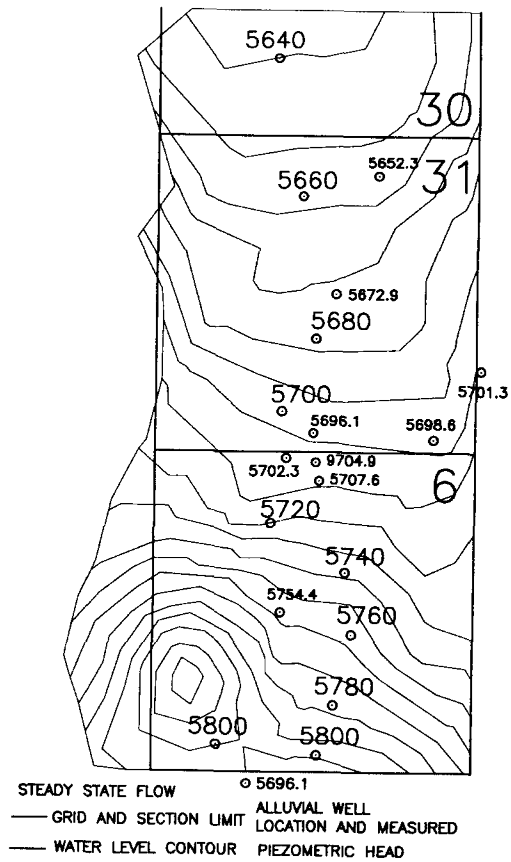

Fig. 4. Simulated groundwater levels after model calibration.

the Lowry landfill concentration was inferred from the present level of contamination in well 105 which is adjacent to the disposal area. The source concentration was then adjusted in the model to obtain the best fit of observed concentrations with model results. A source concentration for chloride of from 8000 to 10,000 mg/l gave the best fit in the model. The location of the disposal ponds and the amount of liquid wastes which were disposed of annually varied with time at Lowry. The capability of the model to match calculated potentiometric heads with observed heads and to match model calculated concentrations with observed concentrations is indicative of the goodness of fit of the calibration procedure. A comparison between calcula-

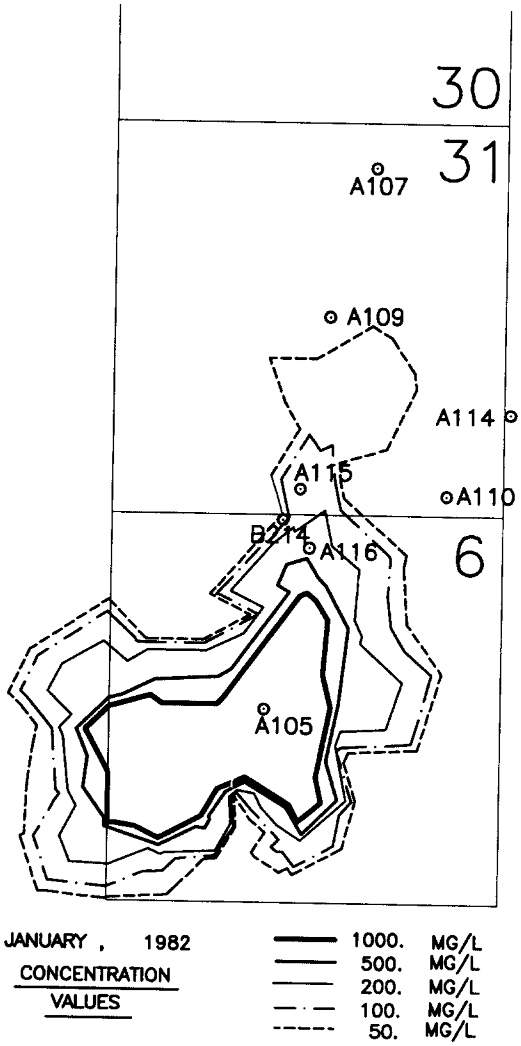

Fig. 5. Simulated chloride concentration distribution, April 1981

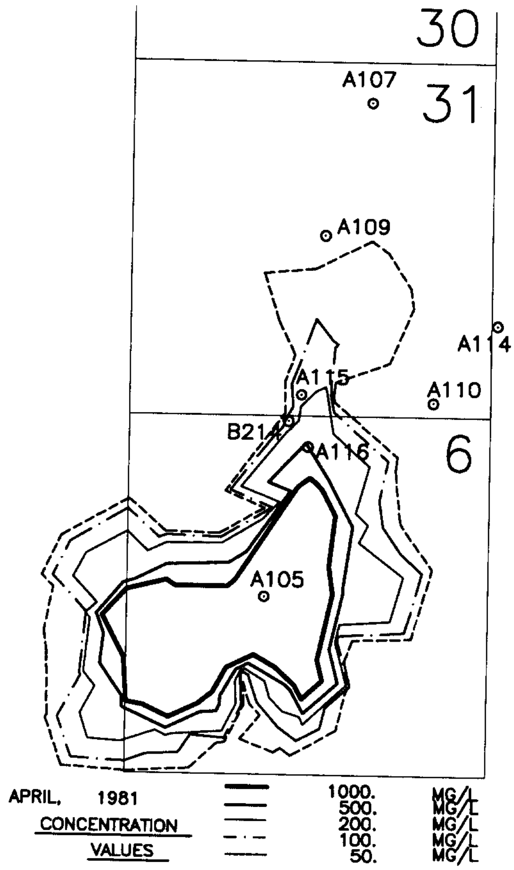

Fig. 6. Simulated chloride concentration distribution, January 1982

ted potentiometric heads and measured heads at the alluvium wells can be observed (Fig. 4). There is a lack of field data in the disposal area but otherwise the agreement is fairly good. The model calculated potentiometric heads agreed within 0.61 m of the water level altitude in the observation wells. A plot of the calculated concentrations for April 1981 is shown on Fig. 5, for January 1982 on Fig. 6 and for April 1991 on Fig. 7. Chloride concentrations were chosen to be modeled because the chloride ion is considered to be a conservative tracer and because the laboratory analysis for chloride is easy to do and yields consistent results. The migration of chloride should represent the maximum extent of the groundwater contamination at the Lowry landfil. Thus far, chloride concentration has been detected in only 14 observation wells and has migrated about 3/4 of 1.61 km to the north in the alluvium from the disposal areas. Tables 2a, 2b, 2c and 2d give a comparison between model calculated chloride concentrations with observed concentrations for times corresponding to Fig.s 5, 6 and 7 for selected monitoring wells. The differences bet-

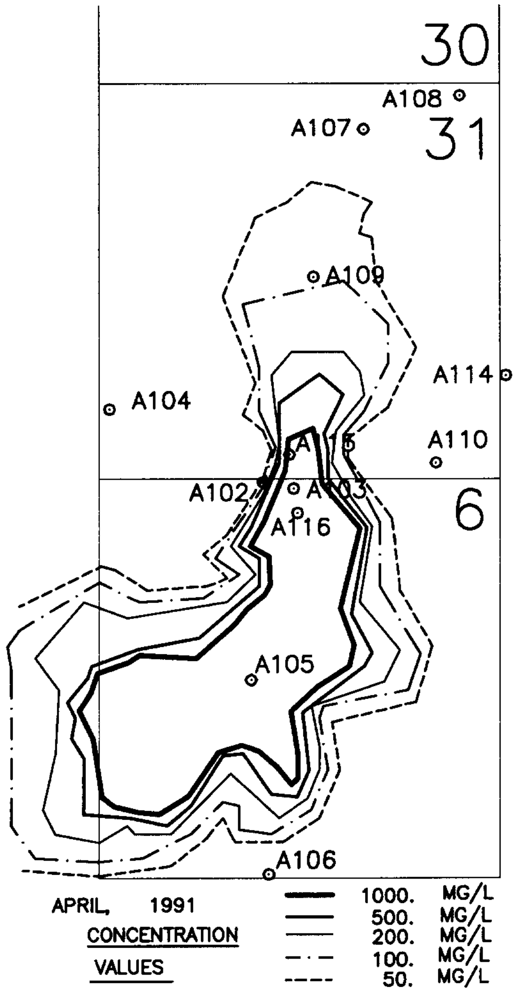

Fig. 7. Simulated chloride concentration distribution, April 1991.

ween calculated and observed concentrations are fairly small except for well A-109. The overall calibration of the model can be classified as good. Model calculated values agreed very closely with the observed field data but in general there was insufficient data to classify the model fit as excellent.

The calibrated model was used to calculate the quantity of groundwater flow and the velocity of the groundwater. From the model, the total recharge to the groundwater is 60.8 gallons per minute of which 12.8 gallons per minute is from the recharge by liquid wastes and the remainder as groundwater under flow across the model boundaries. The Darcy velocity at the central channel of the alluvium between Sections 6 and 31 is about 0.18~0.27 m/day.

Table 2a. Comparison Between Calculated and Observed Chloride Concentrations for April, 1981.

Well	Observed	Calculated
A-105	8,975	3,273
A-109	18.9	29
A-115	142	141.8
A-116	346	341.8
B-214	108	90.4

Table 2b. Comparison between Calculated and Observed Chloride Concentrations for January, 1982.

Well	Observed	Calculated
A-105	9,200	3,202
A-109	26	31
A-115	168	186
A-116	436	440
B-214	129	114

Table 2c. Comparison between Calculated and Observed Chloride Concentrations for April, 1991.

Well	Observed	Calculated
A-105	9,650	4,253
A-109	96	84
A-115	860	940
A-116	1,020	1,080
B-214	380	540

Table 2d. Comparison between Calculated and Observed Chloride Concentrations for January, 1992.

Well	Observed	Calculated
A-105	9,870	4,635
A-109	98	86
A-115	905	952
A-116	1,085	1,098
B-214	398	425

4.3 Simulations for prediction

The model can be used to predict future groundwater concentrations. In this way the model is a valuable tool that shows the response of the aquifer to conditions that are different from the present field situation. Successful application of the model for prediction purposes depends on how well the model simplifications and assumptions represent actual future field conditions and on how accurate the model parameters have been determined. Simulations were run to predict the chloride concentrations in the year 1996 and 2000. A groundwater barrier is planned to be constructed at Lowry to prevent the northward migration of the contaminants in the year 2000. The effect of this barrier on future groundwater concentrations were simulated. All of the simulations assume steady-state flow. The geohydrological conditions assumed in the calibration of the model are now assumed to be valid during the future simulation period.

4.3.1 Model Simulation for the 1996 year, Run 1

This simulation predicts the future groundwater chloride concentration in the year 1996 if the disposal operations are terminated but no remedial measures are taken to prevent the migration of the constaminats already in the groundwater. The results of this simulation are shown on Fig. 8. This simulation assumed that the disposal activities continue at a constant rate of 6.7 million gallons of liquid waste per year and no remedial action is taken. Fig. 8 shows the results of this simulation in the year 1996. the contamination plume extends past the northern boundary. The irregularities in the shape of the isoconcentration lines are due to the numerical approximations in the model as the result of dividing the model area into finite elements.

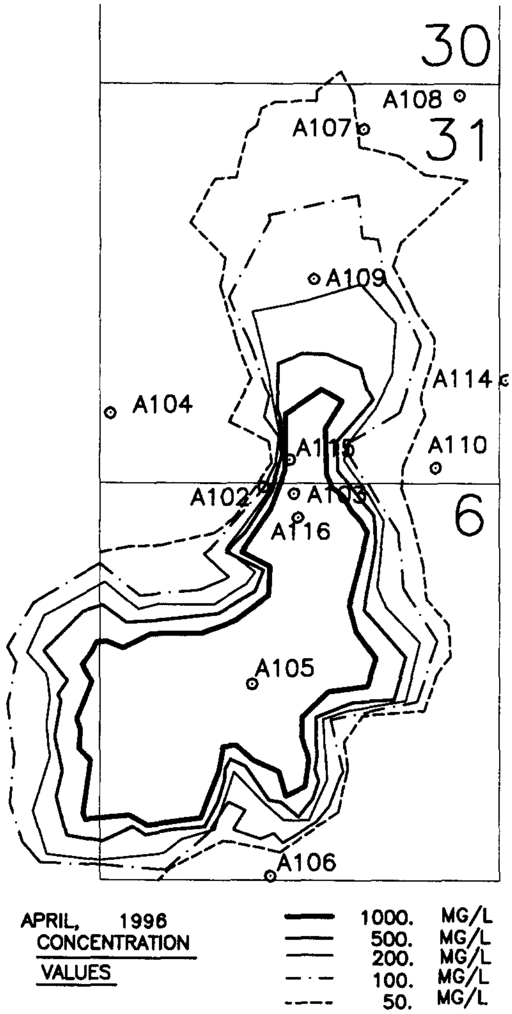

Fig. 8. Predicted chloride concentration distribution for the 1996 year, Run 1.

4.3.2 Model Simulation for the 2000 year, Run 2

This simulation assumed that the disposal activities continue at a constant rate of 6.7 million gallons of liquid waste per year and no remedial action is taken. Fig. 9 shows the results of this simulation in the year 2000. The concentration of the

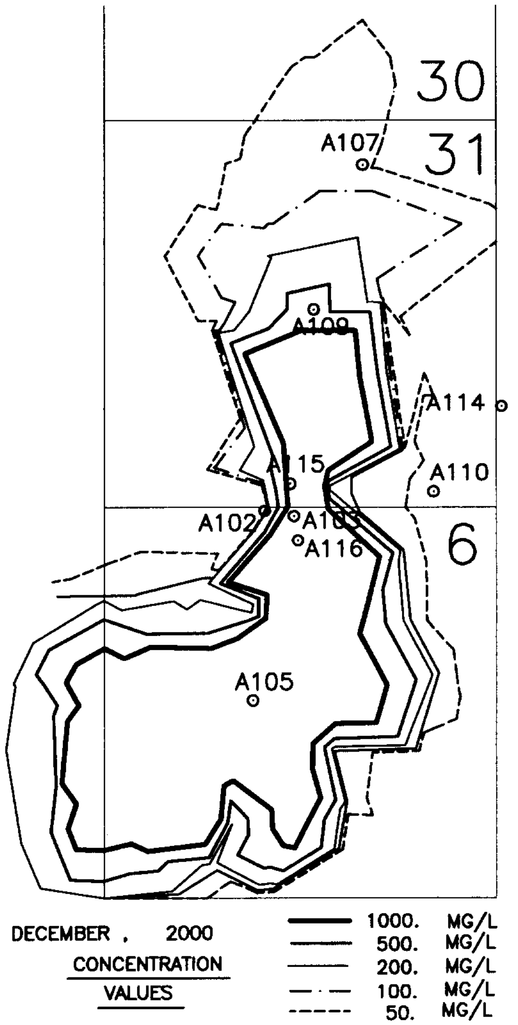

Fig. 9. Predicted chloride concentration distribution for the 2000 year, Run 2.

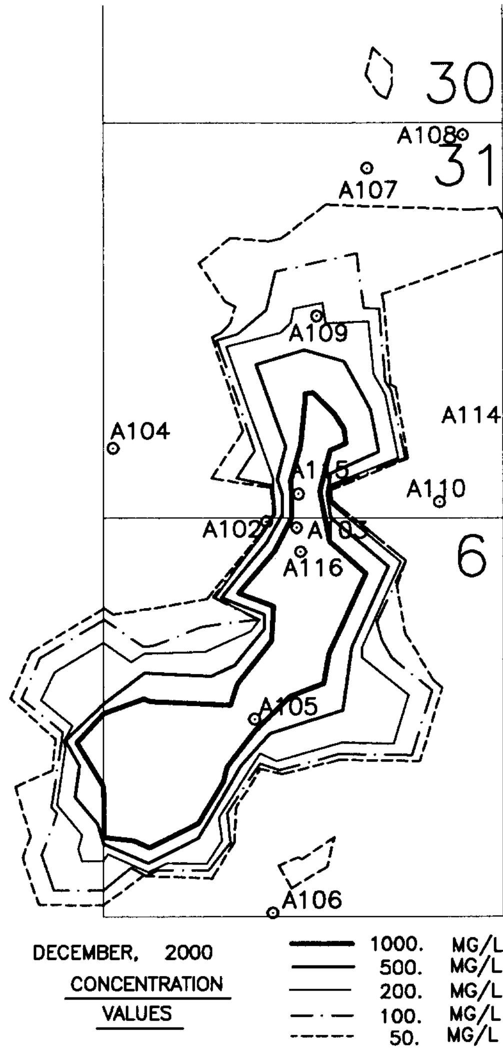

Fig. 10. Simulation result after disposal termination, Run 3.

contaminated groundwater that has migrated off of the landfill is not high but does represent a potential threat to water users in this area. Additionally the contamination has spread to the wider sections of the alluvium channel making it more difficult to implement remedial measures to contain the groundwater the landfill. The disposal simulation is rather moderate considering the increasing trend in the volume of wastes disposed at the site.

4.3.3 Model Simulation after disposal termination, Run 3

This simulation predicts the future groundwater chloride concentration in the year 2000 if the disposal operations are terminated but no remedial measures are taken to prevent the migration of the constaminats already in the groundwater. As seen in the Fig. 10, the contaminants move mainly th-

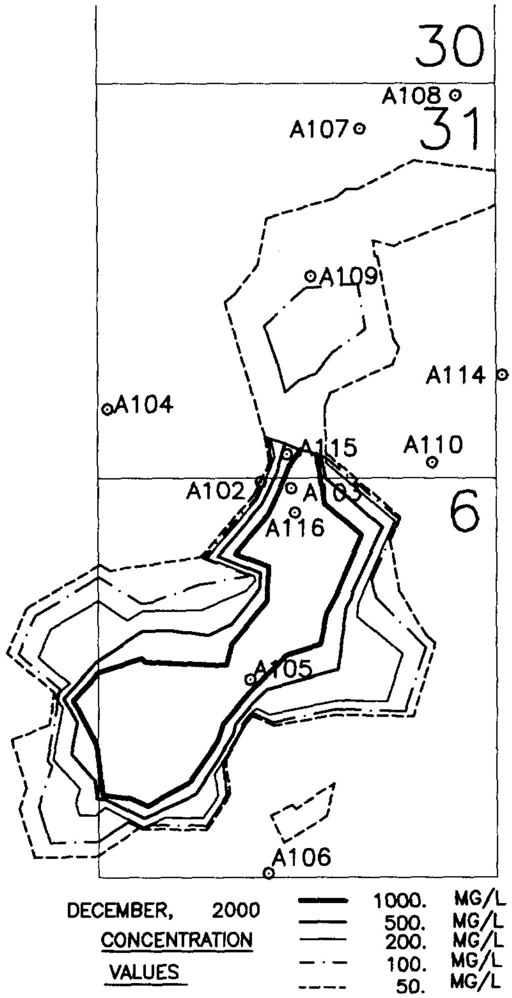

Fig. 11. Simulated chloride concentration distribution after barrier, Run 4.

rough the alluvium to the north. The plume has reached well A-109 and is close to well A-107. In the year 2000 the plume has still migrated past section 31 and off of the Lowry landfill property. Eventually, the plume is expected to migrate as far as north side of Section 31. Most of the northward contamination is confined to the alluvium and little has spread to the bedrock. The discontinuity between the permeability of the alluvium prevents significant migration of contaminants in the downgradient bedrock.

4.3.4 Model Simulation after the intercepting by barrier, Run 4

The barrier as about 250 meter long and about 122 meter downstream from well A-115 is simulated. The barrier is assumed to be in place by the end of 2000. In this simulation, the contaminated groundwater intercepted by the barrier is collected, treated and returned to the aquifer downstream of the barrier. The disposal operations are considered to be terminated. The rate of pumping at the barrier is such that a steady state constant head is maintained at the barrier. With this operation of the barrier, 5.6 gallons per minute is intercepted by the barrier. The chloride concentration in the model of the treaded water returned to the aquifer was 25 mg/l. The results of this simulation are shown on Fig. 11. In this model simulation the chloride concentrations in the aquifer downstream of the barrier are significantly reduced as compared to the previous simulations. The areal extent of the contaminated groundwater downstream from the barrier has also been significantly reduced.

5. CONCLUSIONS

In this study a finite element contaminant transport model was applied to the Lowry landfill where hazardous liquid waste disposal has taken place in the past. The model was successfully applied to an actual field problem at the Lowry landfill. The elements have to be smaller in these places of striking different peculiarities to reproduce better the sharp front in the concentration distribution between alluvium and bedrock. The injection or discharge of a great amount of mass produces instability if small time steps are not taken at the beginning of these periods. In spite of these difficulties, the model has simulated in a satisfactory way the behavior of the solute in the aquifer. Several simulations were shown the migration of the groundwater contamination to the north at the Lowry landfill. The treatment of the water pumped by the barrier and returning the treated water to the aquifer downstream of the barrier is the most effective way to clean the aquifer downstream of the barrier. However, it is simpler to pump the water from the aquifer and to dispose of the contaminated groundwater using evaporation ponds or returning it to the aquifer in the disposal area. The correct study of the bedrock would be by means of an exact definition of the different strata and the rate of percolation and solute movement through each of the layers.

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