JOURNAL OF
ENVIRONMENTAL HYDROLOGY

The Electronic Journal of the International Association for Environmental Hydrology
On the World Wide Web at http://www.hydroweb.com

VOLUME 8 2000

NUMERICAL SIMULATION OF FLOW AND CONTAMINANT MIGRATION AT A MUNICIPAL LANDFILL

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The flow and transport characteristics of the Ano Liosia Landfill site in Athens, Greece, were simulated by numerical methods using groundwater flow and contaminant transport models. A methodology for modeling the water flow and the transport of chlorides in steady state groundwater flow is presented. The Strack theory, which describes the extension of a simulation domain from two to three dimensions, is applied. This theory was used to determine the streamlines and path lines in three-dimensional space. For most waste disposal systems, interpretations and assumptions regarding undefined field properties will always have to be made in order to render the problem tractable. In this study, synthesis of field-measured and literature values of field properties provided a framework for the validation. The hydraulic heads derived from simulation fluctuated between 7 and 9 m in the area of the disposal site. The mean velocity (mean Darcy velocity / porosity) was found to be 6.5x10^{-2} m/day. The pollutant transport was simulated for 30 years. The simulation gave a plume of chlorides that extends 1843 m in length and 92 m in depth northwestward from the landfill. This study has demonstrated that an accurate and efficient computation of three-dimensional transport under advection - dispersion dominated conditions is feasible through extension of a two-dimensional to a three-dimensional flow model domain. The possibility of errors that may escape attention is eliminated when a coupled flow-transport three-dimensional model is used.

INTRODUCTION

Decomposition of municipal solid waste in landfill results in the formation of leachate, an aqueous liquid containing organic and inorganic constituents. The nature of landfill leachate varies widely. Leachate composition is a function of climatic conditions, hydrogeologic factors, waste type and quantity, and the age of the landfill. One problem is the possible contamination of soil, groundwater and surface water that may occur as leachate produced by water or liquid wastes moving into, through and out of the landfill migrates into adjacent areas.

In recent years there has been an increasing concern for groundwater contamination caused by the disposal of solid waste in landfills, (Gureghian et al., 1981). Numerical models for steady state groundwater flow in porous media have been frequently used to predict groundwater flow and contaminant migration in aquifers. In the mathematical simulation of aquifer contamination, an accurate definition of the flow system is of vital importance. A commonly used approach to obtain hydrogeologic parameters is by calibration of a flow model against field measurements of hydraulic heads, (Carrera and Neuman, 1986, Kauffmann and Kinzelbach, 1989). One weakness of this approach is that in most cases, head measurements are too limited in number to represent adequately the flow variations in the space-time domain that is relevant for transport problem. A flow model based on head measurements only may therefore not be fully compatible with the transport model. In addition, application of models to the analysis of steady flow in an aquifer requires knowledge of the spatial distributions of hydraulic conductivity, boundary conditions and recharge rates into the aquifer. Usually, measurements of hydraulic conductivity are too few to fully characterize the heterogeneity of the aquifer and often involve errors and uncertainties. Moreover, definitive information about boundary conditions and recharge rates rarely exists because of the complexity of the geology of the aquifer and the lack of reliable means to measure or estimate fluxes at boundaries, or recharge rates and their distribution. As a result, numerical models are often calibrated by adjusting values of hydraulic conductivity, boundary conditions and recharge rates so that simulated hydraulic head values are in agreement with hydraulic head measurements in spite of possible measurement errors (Yeh, 1986, Yeh and Mock, 1996). Some of the resulting errors may escape detection if a coupled flow-transport model is used without careful scrutiny of the flow field produced by the model (Frind et al., 1985).

A case study involving an unconfined aquifer in the vicinity of a landfill site is examined in this paper. The flow pattern was simulated by the MODFLOW model (McDonald and Harbaugh, 1988) in two dimensions. At a later stage, an extension of the model into a three-dimensional domain was accomplished taking into account the Strack theory (Strack, 1989), and finally the transport pattern was simulated by the MAD model (Syriopoulou and Koussis, 1991) in three dimensions.

MODELING PROCEDURE

Area Description

Landfilling of municipal solid waste has been the most common method of solid waste disposal in recent years. The Ano Liosia landfill is the major disposal site of Athens, Greece, and it has been in operation since 1973. It currently receives the wastes from an urban population of nearly 3,500,000 inhabitants and extends to an area of 173 ha, of which 45 ha are covered by deposited wastes. The average quantity of wastes deposited daily is about 4000 tons. The daily per capita production of wastes is estimated at approximately 0.875 kg/capita/day with an annual increase rate of the range of 1.5-2.5 percent (ACMAR, 1996). The municipal waste is characterized by the presence of high quantities of organics (52 percent) which includes 20 percent paper, 9 percent plastics, 4 percent metals, 4 percent glass, 3 percent textiles, wood and leather, 4 percent inert, and 4 percent other.

The bottom of the landfill is not lined. There is no provision for the collection of leachate apart from a system of surface trenches dug along the bottom contours of the deposits that collect leachates mixed with surface runoff and divert it to a pond, with a capacity of 1000 m^3. The wastes are compacted on site mechanically, deposited in layers of 150-350 cm and are covered by 50-200 cm of earth, sand, gravel and clay materials. The depth of deposition in the centre of landfill body exceeds 20 m, (Stournas et al., 1990).

The area in which the disposal site is located is moderately steep, semiarid, and is covered by native vegetation consisting of low plants and grass. The prevailing underlying geological formations are limestone (karst type) of the Triassic and Jurassic ages.

Model Description

The computer models used in this study are MODFLOW, the U.S. Geological Survey modular three dimensional finite difference flow model, (McDonald and Harbaugh, 1988) and MAD, the matched artificial dispersivity model (Syriopoulou and Koussis, 1991).

The MODFLOW Model

The three-dimensional groundwater flow model MODFLOW is one of the most widely used and accepted models in the world today (Osiensky and Williams, 1997). Three dimensional movement of groundwater through porous earth material may be described by a partial differential equation as follows:

$$\frac{\partial}{\partial x} * \left(K_{xx} \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y} * \left(K_{yy} \frac{\partial h}{\partial y} \right) + \frac{\partial}{\partial z} * \left(Ko_{zz} \frac{\partial h}{\partial z} \right) - W = S_s \frac{\partial h}{\partial t} \tag{1}$$

where K_{xx} K_{yy} and K_{zz} are the hydraulic conductivities along x, y and z coordinates (l/t); h is the hydraulic head (l); W is a volumetric flux per unit volume and unit time representing sources and/or sinks of water (t^{-1}); S_s is the specific storage of porous material (l^{-1}); and t is the time (t).

MODFLOW employs a finite difference method as a numerical approximation scheme. With the model MODFLOW, a continuous system is subdivided into a finite number of discrete points in terms of space and time. The partial derivatives in the groundwater flow equation are replaced by differences between functional values at these points. This process results in systems of simultaneous linear algebraic points and time. Flow from external stresses such as flow to wells and drains, areal recharge and flow through streams and riverbeds can be simulated.

The mathematical program of this model consists of a main program (Basic Package) and a large number of highly independent subroutines called modules that are grouped into “packages”. In this application, the following packages have been used: Basic Package, Block Centered Package, Recharge Package and Preconditioned Conjugate Gradient 2 Package.

Griding and Input Data

The model was applied in two dimensions due to lack of consistent data. The spatial discretization of the solution domain in MODFLOW is based on block-centered grid. The basic unit for both the definition of the medium properties and application of the water balance that leads to the formation of the finite difference equation is the cell. The unknown heads are defined at the central node of each cell.

The finite difference grid for MODFLOW application in this study consisted of one aquifer layer, which had 51 x 90 cells and covered an area of 45.9 km^2. The cell resolution was 100 x 100 m. The aquifer was considered to be unconfined, homogeneous and isotropic. The prevailing geological formation is bentonitic limestone.

In this simulation, the infiltration of precipitation into the landfill and the surrounding land was assumed to be the only source of recharge to the whole aquifer system. The recharge rate was estimated by using the Hydrologic Evaluation of Landfill Performance (HELP) model developed by the U.S. EPA, (Schroeder et al., 1994), taking into account climatic data from 1973 to 1995.

The characteristics of the aquifer under study which have been used as input data for the model application are shown in Table 1.

Table 1. Aquifer Characteristics.

Parameter	**Units**	
Hydraulic Conductivity	m/day	8.64 e^{-12}
Bottom Elevation	m	-450
Top Elevation	m	+150†
Recharge	m/day	4.73×10^{-4} (into the landfill)
		4.17×10^{-4} (into the surrounding land)

† Above Mean Sea Level

Due to the hydrogeological formations of the area, a no-flow boundary was set along the eastern part of the area under simulation (Neuman boundary). Since there were no convenient geological boundaries at either the northern and southern end of the section or the western boundary, constant-head boundaries (Dirichlet) were assigned to these parts. The heads were measured in wells located within this area.

Model Extension into Three Dimensions

The model was then extended into three dimensions. This was accomplished taking into account the Dupuit-Forchheimer assumption (Strack, 1989). This assumption consists of neglecting the resistance to flow in the vertical direction. Actually the resistance is not really neglected but it is rather not associated with any head loss in the vertical direction. In this work, this assumption was used to determine the streamlines and path lines in three-dimensional space. According to the theory of Strack (Strack, 1989), in the case of flow in an aquifer of constant thickness *H*, where infiltration occurs at a constant rate, *N*, along the horizontal upper boundary of the aquifer (Figure 1), the

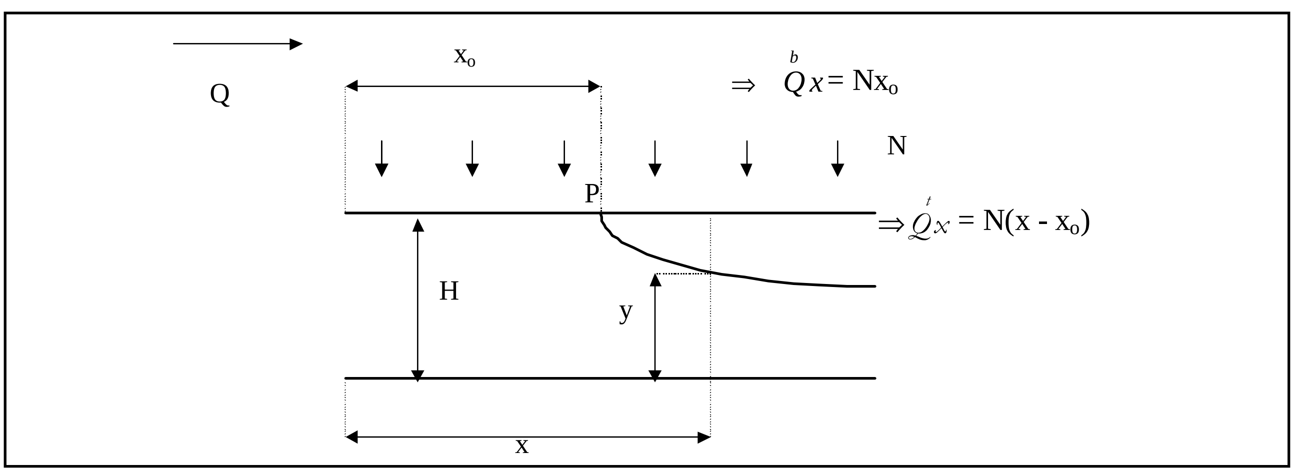

Figure 1. The path of a fluid particle.

following relation stands:

$$\frac{y}{H} = \frac{\overset{t}{Q}x}{Qx} = \frac{Nx_o + Qo}{Nx + Qo} \quad \Rightarrow \quad y = H\frac{Nx_o + Qo}{Nx + Qo} \tag{2}$$

where $\overset{t}{Q}x = N(x - x_o)$, the amount of water flowing between the streamline and the upper boundary. This amount equals the infiltration over the distance $x - x_o$, $\overset{b}{Q}x = Nx_o$, the amount of water flowing between the base boundary and the streamline, and finally $Qx = \overset{t}{Q}x + \overset{b}{Q}x = Nx$.

For the aquifer under examination the thickness was considered to be constant since it fluctuates between 600 and 610 m. The annual recharge equals 160 mm (= 4.45×10^{-4} m/day). The Darcy velocity in the cell which was defined by row 14 and column 34 (slice 9, Figure 2) equals 9.08×10^{-3} m/day. This was multiplied by the thickness of the aquifer and then the amount of flowing water, Q, was found to be 5.45 m^2/day. Therefore, for a known x_o, y was calculated for every $x \geq x_o$. In the present application, the streamlines (lines with a parabolic form, $y = f(x)$) were calculated for x_o= 25 m and x = 25 m.

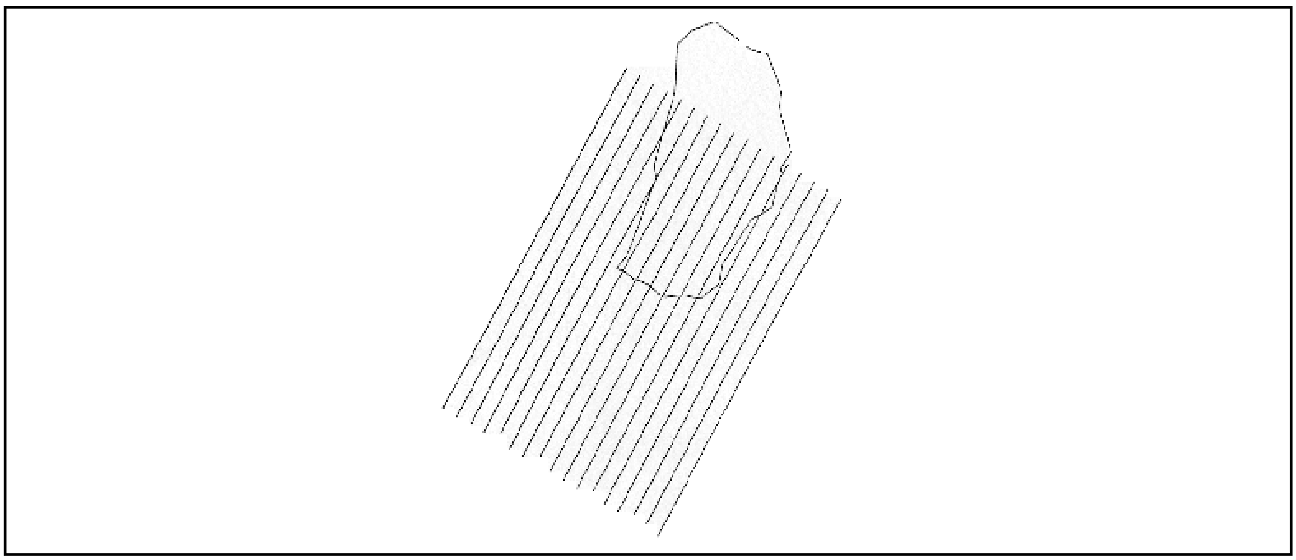

Figure 2. Vertical slices.

The equation of the line that is orthogonal to a parabolic line is given by Bronstein and Semendjajew, 1964. This equation is as follows:

$$Y - y = -\frac{1}{\frac{dy}{dx}}(X - x) \tag{3}$$

X and Y correspond to the vertical line while x and y are the streamlines. Therefore, it follows that

$$Y = y + \frac{Nx + Qo}{Ny}(X - x) \tag{4}$$

Considering that the initial points of the vertical line are known, since they are calculated from Equation (2), and that at the initial point where the vertical line crosses the next streamline, $Y = y$, it is found that

$$X = \sqrt{\frac{H(Nx_o + Qo)y + Qo\left(x^2 - y^2\right) + Qo^2 x / N}{Nx + Qo}\left(\frac{N^2\left(y^2 - x^2\right) + Qo^2}{2\left(N^2 x + QoN\right)}\right)^2} - \frac{N^2\left(y^2 - x^2\right) + Qo^2}{2\left(N^2 x + QoN\right)} \quad (5)$$

At a later stage, an algorithm in Microsoft Fortran was developed for the calculation of the nodes where the vertical lines cross the streamlines. In this way, the transverse grid was developed. This algorithm also creates the input file that is required by the MAD model. Seventeen vertical slices were created, in a way that they were parallel to the horizontal flow (Figure 2). The distance between every two slices was 125 m. Therefore, the region under simulation covered an area of 2 km x 3 km. A uniform flow regime was assumed (uniform flow direction and velocity distribution per slice). The pollution source distribution varied in every slice.

The MAD Model

MAD is based on the premise that the solution of the advection equation is a first approximation to the solution of the complete advection-dispersion equation in the flow direction. The model aims at the efficient confrontation of the difficulties associated with advection-dominated conditions (oscillations – free resolution of sharp fronts). Employing a spatial splitting algorithm within the framework of the (curvilinear) principal directions of transport formulation, it computes transport along streamlines by a variant of the pseudoviscosity method. The algorithm is subject to mild grid design constraints that afford efficiency in terms of CPU time and storage; its accuracy has been tested against analytical solutions elsewhere (Syriopoulou, 1987, Koussis et al., 1989, Syriopoulou and Koussis, 1991).

For the orthogonal, curvilinear principal directions (PDs) of transport coordinate system in Figure 3, in which S follows the streamlines, the mass transport equation in steady, saturated groundwater flow reads (Bear, 1972):

$$\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial S} = u\left\{\left(\frac{\partial}{\partial S}\right)\left[\left(a_L + D{*}/u\right)\left(\frac{\partial C}{\partial S}\right)\right] + \left(\frac{\partial}{\partial \mathrm{T}}\right)\left[\left(a_T + D{*}/u\right)\left(\frac{\partial C}{\partial \mathrm{T}}\right)\right]\right\} + R + \sigma \quad (6)$$

where C is the solute concentration, u is the linear pore velocity along S, a_L is the longitudinal and a_T the transverse dispersivity, σ is the source/sink term, R is the reaction term and t is the time.

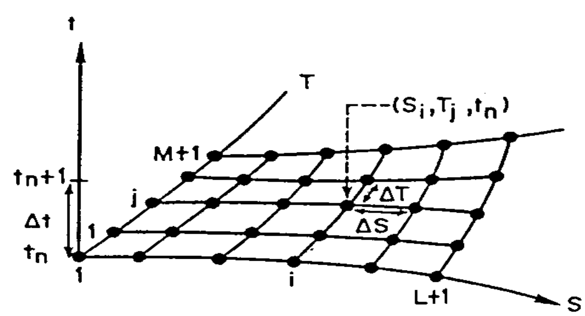

Figure 3. Principal directions of transport grid for the matched artificial dispersivity principal directions algorithm.

The dispersion coefficients are of the form $D = u\alpha + D*$ where $D*$ is the molecular diffusivity. As the above equation shows, a single advective and two dispersive terms remain. When used with simply constructed grids, the PD approach implies moderately curved flow lines.

Following Cunge's (1969) analysis of the Muskingum scheme, grid size constraints for the MAD scheme are derived. The constraints define a 'feasible' area where oscillations are avoided. This area is defined from the following mathematical relations:

$$\begin{aligned} &1+2/P \geq C \geq 1-2/P \\ &2 \leq P \leq \infty \end{aligned} \tag{7}$$

where P is the Peclet ($P = ud/D_L$) and C the Courant (udt/dS) numbers.

Griding and Input Data

The PD grid construction is identical to that shown in Figure 3. The streamlines and the equipotentials are orthogonal to each other.

The leachate chloride concentrations at the landfill were measured by experimental analysis. By averaging the data at all sampling points, an average chloride concentration of 4057 mg/l was determined. For modeling purposes then, a leachate chloride concentration of 4000 mg/l was used. It was felt that the use of a spatially varying value was not warranted and would be difficult to determine. For a homogeneous landfill it can be assumed that a spatially constant chloride concentration can be used over the active extent of landfill. In addition, an assumption was made concerning the stability of the concentration over the course of time. The effect of unsaturated zone thickness on chloride concentrations was assumed to be negligible. To simplify the model consistent with the assumption of a steady state flow system, an average annual leachate concentration was assumed. Thus the effects of a changing infiltration rate, and consequently leachate volume throughout the year, were assumed to be negligible.

In view of the uncertainties associated with the other calibrated parameters, constant dispersivities were considered adequate. Optimal constant dispersivity values for the area under study have been reported by Chelmis et al., (1990). Because best fit values are model-dependent, they were used as guides. The dispersivities used in this simulation were as follows: a_L = 10 m, a_T (horizontal) = 3 m and a_T (vertical) = 0.01 m. The mean Darcy velocity was found to be 7.8×10^{-3} m/day through the MODFLOW application, and the aquifer porosity used was 12 percent.

Neuman conditions were applied at the free upper surface while Dirichlet conditions were applied at the upper surface where a pollution source existed (chloride concentration of 4000 mg/l). Dirichlet conditions were applied as base boundary (chloride concentration of 50 mg/l representing the background concentration). To meet the grid Peclet number constraint, an average step dS = 25 m was used. The transverse grid spacing was chosen dense enough to ensure a detailed plume description. The resulting space grid consisted of 130 streamlines, 117 equipotentials and 8882 nodes. The transport over 30 years was simulated in 60 cycles of duration dt = 182.5 days, which gave $0.2 \leq C \leq 1.8$, satisfying the constraint (Equation 7) everywhere.

RESULTS

Water Flow Simulation

The calibration of the groundwater flow model is crucial to its applicability and constituted the most time-consuming effort within the modeling procedure. The approach used to obtain the hydrological parameters was by calibration the flow model against field measurements of hydraulic head. Seven wells with known hydraulic heads were used for the calibration.

In Tables 2 and 3, the various wells with the known heads are presented along with the residuals resulted after the calibration. In addition the various calibration parameters are shown.

The hydraulic heads derived from the MODFLOW simulation fluctuated between 7 and 9 m in the area of the disposal site. Generally the thickness of the aquifer under study was found to vary between 600 and 610 m. The mean velocity (mean Darcy velocity / porosity) was found to be 6.5×10^{-2} m/day. The heads varied between 0 (at the area of aquifer discharge into the sea) and 10 (at the northern part of the disposal site). Generally, water was to found to move towards the northwestern part of the simulation domain, which corresponds to the real conditions of the area. The high hydraulic heads that have been observed at the disposal site (7 - 9 m) are attributed to the geological morphology of the area. A geological barrier that is located to the eastern part does not permit the water to move towards the west. Overall, the results are satisfactory and represent what may be considered as an average annual system.

Table 2. Model Calibration

Well	Hydraulic heads (field measurements)	Residual
1	2.80	0.40
2	3.30	0.44
3	4.20	0.71
4	4.90	-0.12
5	6.50	-0.11
6	6.90	-0.06

Table 3. Calibration Parameters

Mean residual	0.21
Standard deviation of residual	0.32
Sum of square residuals	8.85 e^{-1}
Absolute mean residual value	0.31
Min residual value	-0.12
Max residual value	0.71
Real head range	4.10
Absolute mean residual value/real head range	0.078

Transport Simulation

The pollutant transport was simulated for 30 years. The model was calibrated against the chloride concentrations that had been measured by experimental analysis in various wells in the vicinity of the landfill site. Therefore the transport behavior was captured reasonably. The simulation satisfied the conditions that involve the Peclet and Courant number everywhere in the grid. Transport results are displayed as chloride concentration contours in milligrams per liter and are shown in Figure 4 (for vertical slices 5, 9, and 12). The figures represent the contaminant migration simulation assuming that the entire landfill is contributing chloride concentrations of 4000 mg/l from the initial development of the landfill.

As a result of over of 30 years of leachate input to the aquifer, the simulation gave a plume of contaminated groundwater that extends 1843 m in length and 92 m in depth northwestward from the landfill. Taking into account the fact that the distance between every 2 vertical slices in 125 m, it can be concluded that the width of the plume fluctuates between 1000 and 1250 m.

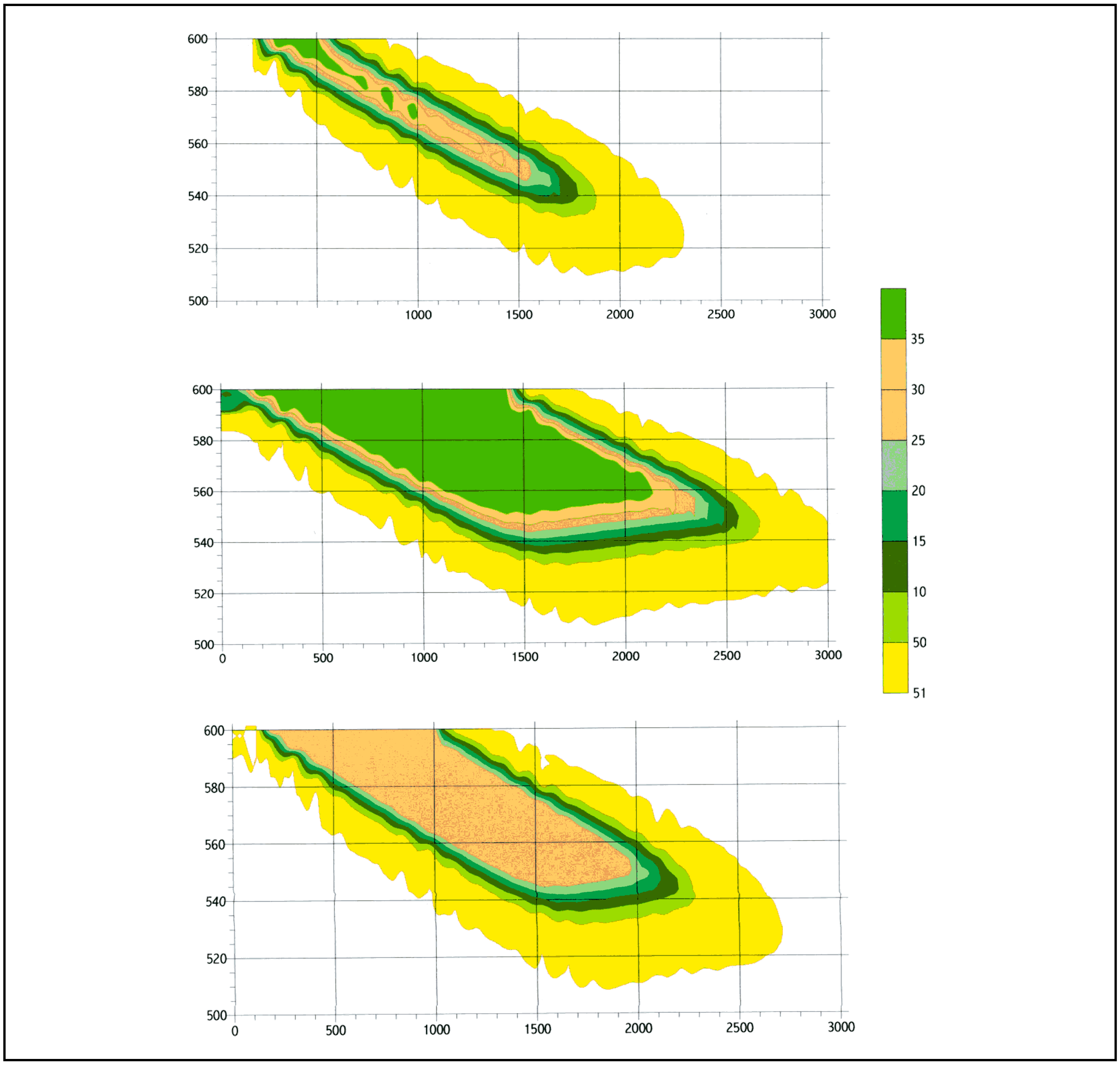

Figure 4. Chloride concentration contours for vertical slices 5, 9, and 12.

The behavior of every vertical slice as far as the extent and depth are concerned is associated with the characteristics of the pollution source (the way the leachate leaks, leakage duration, existing waste disposal pattern), the conditions around the source, the characteristics of the aquifer, such as flow direction, velocity, and gradient, and finally with the dispersion coefficients. The results are in agreement with the real measurements concerning the chloride concentrations. Nevertheless, the concentrations in two wells were found to be at lower levels than those calculated by the model. This can be attributed to the existence of discontinuities, fissures and fractures where the flow is quite different. The migration of pollutants is influenced by a large number of parameters. Some of them are not included in the simulation algorithms, but may influence the movement pattern.

The migration is associated with various physical mechanisms of advection and dispersion, with a large number of chemical and biochemical reactions, as well as with the various characteristics of the geological formations such as soil pH, leachate composition, etc. The pH of the soil and leachate influence the pollutant mobility. The forms in which the pollutants exist and their solubility depend on the pH. The same holds for the hydrolysis and ion exchange reactions. The most important factor that influences pollutant migration is the composition of the leachate. Complex leachates behave in a quite different way. The solubility, the density and the chemical structure have a major impact on the movement of the plume. The plume size may be reduced if the waste disposal rate or the water table level is minimized. The migration of the plume may be retarded due to sorption and other chemical reaction, and due to the minimization of the water flow velocity.

The groundwater flow rate depends highly on the type of the geological formations through which water moves. Higher rates are generally observed at large grain size formations. The physical and chemical characteristics such as hydraulic conductivity, ion exchange capacity and sorption also have an impact on the migration rate.

CONCLUSIONS

In the previous sections, groundwater flow and chlorides transport modeling at the Ano Liosia landfill site have been discussed. When available, known values of parameters determined from the field or laboratory study were used.

The necessity for simplifying and interpretative assumptions decreases the intended validation purpose of the study. However, for most waste disposal systems, interpretations and assumptions regarding undefined field properties will always have to be made in order to render the problem tractable. In this study synthesis of field-measured and literature values of field properties provided the framework for the validation.

An important factor in constructing a model of contaminant systems is a correct interpretation of the flow system and the development of an accurate representation. For conservative contaminants such as chloride, migration is governed by dispersion and advection.

Modeling provides a useful basis for contaminant transport studies, both in the evaluation of the hydrogeologic parameters relevant in the context of the transport problem and the screening out errors in the simulated flow field. This study has demonstrated that an accurate and efficient computation of three-dimensional transport under advection-dispersion dominated conditions is feasible through extension of a two-dimensional to a three-dimensional flow model domain.

ACKNOWLEDGMENTS

The authors wish to thank A.D. Koussis for providing the MAD model and for his contribution in the model calculations, helpful discussions and advice.

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