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Numerical simulation of leachate transport into the groundwater at landfill sites

E. Rouholahnejad[1], S.A. Sadrnejad[2],

[1] *M.Sc Student of Dep of Civil Engineering, K.N.Toosi University of Technology, Tehran*
Email: e_rouholahnejad@yahoo.com
[2] *Professor of Dep of Civil Engineering, K.N.Toosi University of Technology, Tehran*

Abstract: This paper presents the development of a two-dimensional numerical model that can be used for quantifying groundwater inputs and associated contaminant discharge from a landfill facility with capacity of 2000 ton/day into the nearest aquifer. The model can be used for the simulation of contaminant transport in aquifers in any scale. It is established on finite difference-finite volume solution of two-dimensional advection-diffusion-linear sorption with first order decay equation. The goal is to assess leachate migration from the landfill in order to control its associated environmental impacts, particularly on groundwaters. Leachate discharge from landfills is the main route for release of the organic and inorganic contaminants through subsurface, commonly encountered in the refuse. Leachate quantity and potential percolation into the subsurface in this project was estimated using the Hydrologic Evaluation of Landfill Performance (HELP) model before. Contaminant transport simulation after leachate receives to the groundwater surface is done in this project. A comprehensive sensitivity analysis to leachate transport control parameters was also conducted. Sensitivity analysis suggest that changes in source strength, aquifer hydraulic conductivity, and dispersivities have the most significant impact on model output indicating that these parameters should be carefully selected when similar modeling studies are performed. The sensitivity of the model to variations in input parameters results in two opposing patterns of contaminant concentration. While higher groundwater velocities increase the speed of plume spread, they also increase the dilution ratio and hence decrease the concentration. Increasing gradient by factor of 2 leads to larger water contamination because of higher velocity. Concentration distribution contours for 5, 25, 50, and 75 years after leachate reaches the ground water table shows that even after 75 years the contaminant won't reach the downstream recipient so it fulfill the water contaminant limitation for K-N and can be used as drinking water.

***Keywords**: Advection- Diffusion- Sorption- First Order Reaction Equation, Finite Volume, Finite Difference, Sensitivity Analysis.*

1. Introduction

Groundwater quality varies due to chemical, geochemical, and biochemical reactions of the pollutants in the subsurface flow systems. To reliably predict the fate of contaminant transport in groundwater, an accurate numerical modeling is required. There are many numerical investigations of advection dispersion equation (transport equation). However, A few studies have been devoted to the more general advection dispersion reaction equation (ADRE). The numerical solution of ADRE has its complexity because of the reaction term. The reaction term account for degradation or adsorption, this term can cause numerical instability problems. Several approaches have been developed to improve the numerical accuracy. Among the numerical methods for solving ADRE, finite difference method (FDM) and finite volume method (FVM), seems to be more popular for the ease of implementation and their relative simplicity (Ataie et al., 1999; Moldrup et al., 1996; Stanbro et al., 2000; Sheu et al., 2000). However, using finite element method (FEM) it is easier to handle complex geometries. There have been extensive debates as to whether FEM or FDM is preferable in groundwater modeling (Zheng et al., 2002). Some investigation showed that FDM introduces larger numerical errors than FEM (Noye etal., 1990; Hossain et al., 1999; Liu et al., 1996; Sheu et al., 2002; Zheng and Bennett, 2002). The effect of numerical dispersion has been considered in numerical studies by many researchers (De Smedt and Wierenga, 1977; Van Genuchten and Gray, 1978; Notodarmojo et al., 1991; Dudley et al., 1991). Incidents of groundwater contamination by landfill leachate have been widely reported since the early 1970s (Albaiges et al., 1986; Dunlap et al., 1976; El-Fadel et al., 1997a; Garland and Mosher, 1975; MacFarlane et al., 1983; Malina et al., 1999; Reinhard et al., 1984; Zanoni, 1972). This created the need to understand the mechanisms that control leachate formation, quality, quantity, and most importantly migration characteristics with associated with spatial and temporal variations during landfill operations and after closure. Leachate discharged from landfills is the main route for the release of the organic and inorganic contaminants commonly encountered in the refuse. Transport processes in landfills are associated with a high degree of uncertainty. While these processes are individually well understood and can be simulated reasonably well in a laboratory setting, their occurrence and interaction in landfills are still not fully comprehended (El-Fadel et al., 1997b). Lack of equipment in developing country and no access to well-known softwares in the field of groundwater contamination because of their cost, made us to develop a transport model in saturated porous media to reach a punctual solution of transport equation and Compare different numerical methods to know how exact they are.

2. Numerical approximation of ADRE

The two-dimensional advection dispersion equation with a first-order reaction is written as: (Zheng., 1990; Zheng and Bennett, 2002)

$$\frac{\partial C}{\partial t} = D_{xx}\frac{\partial^2 C}{\partial x^2} + D_{xy}\frac{\partial^2 C}{\partial x \partial y} + D_{yx}\frac{\partial^2 C}{\partial x \partial y} + D_{yy}\frac{\partial^2 C}{\partial y^2} - \vartheta_x \frac{\partial C}{\partial x} - \vartheta_y \frac{\partial C}{\partial y} - kC \tag{1}$$

Where C is dissolved concentration [ML^3], t is time [T], k is first-order reaction rate coefficient [T^{-1}], D_{xx}, D_{yy} are principal-terms of dispersion coefficient [L^2T^{-1}], D_{yx}, D_{xy} are cross-terms of dispersion coefficient [L^2T^{-1}], and υ_x, υ_y are velocity component in X and Y directions [LT^{-1}]. The equation is for the cases of negligible spatial variability of the dispersion coefficient and velocity components and in these cases D_{yx} and D_{xy} are equal, so simplified form of Eq. (1) is presented (Zheng., 1990; Zheng and Bennett, 2002):

$$\frac{\partial C}{\partial t} = D_{xx}\frac{\partial^2 C}{\partial x^2} + 2D_{xy}\frac{\partial^2 C}{\partial x \partial y} + D_{yy}\frac{\partial^2 C}{\partial y^2} - \vartheta_x \frac{\partial C}{\partial x} - \vartheta_y \frac{\partial C}{\partial y} - kC \tag{2}$$

2.1. A general form of FD approximation for Diffusion term

2-dimensional diffusion equation in transport equation is as follows:

$$\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} \tag{3}$$

The finite difference which is chosen to solve the diffusion term is β Formulation which is known as the crank-Nicolson formulation with β=0.5.

$$\frac{C_i^{n+1}-C_i^n}{\Delta t}=D_x\left[\beta\frac{C_{i+1,j}^{n+1}-2C_{i,j}^{n+1}+C_{i-1,j}^{n+1}}{\left(\Delta x^2\right)}+(1-\beta)\frac{C_{i+1,j}^{n}-2C_{i,j}^{n}+C_{i-1,j}^{n}}{\left(\Delta x^2\right)}\right]+D_y\left[\beta\frac{C_{i,j+1}^{n+1}-2C_{i,j}^{n+1}+C_{i,j-1}^{n+1}}{\left(\Delta y^2\right)}+(1-\beta)\frac{C_{i,j+1}^{n}-2C_{i,j}^{n}+C_{i,j-1}^{n}}{\left(\Delta y^2\right)}\right] \quad (4)$$

This is an explicit-implicit method to solve the diffusion term. Figure 1 shows the diffusive behavior of contaminant injected to a point constantly. The input parameters are given in table 1.

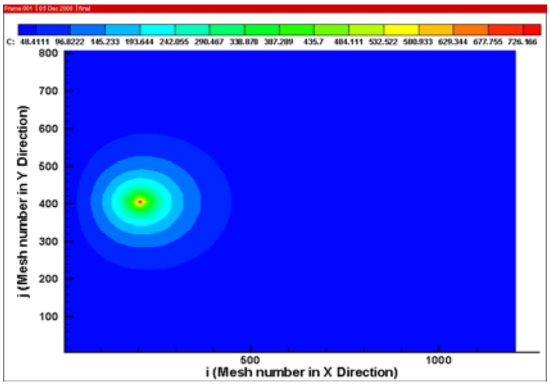

Figure1. 2D- Diffusion of leachate according assumed parameters given in table 1

Table1. Input parameters for 2D diffusion of leachate in a point source case

Parameters	Units	Values
Deltax	mm	10
Deltay	mm	10
Deltat	s	0.25
iInjection	-	21
jInjection	-	41
n	-	12
m	-	121
Time	s	500
β	-	0.5
Injection rate (Q)	$mm^2 s^{-1}$	12.5
C0	$mg L^{-1}$	1000
Prosity		0.25
Velocity (Ux)	$mm s^{-1}$	2
Velocity (Uy)	$mm s^{-1}$	0
Diffusion Coefficient (Dx)	$mm^2 s^{-1}$	60
Diffusion Coefficient (Dy)	$mm^2 s^{-1}$	36

2-2 A general form of finite volume approximation for advection term

One dimensional advection equation in transport equation is as follows.

$$\frac{\partial C}{\partial t}=-\frac{\partial (Cu)}{\partial x} \quad (5)$$

Where C is contaminant concentration (mg L^{-1}), U, is velocity (m s^{-1}), and t indicates time (s). Finite volume base formulations for all advection solution methods are shown in Eq. 6-7.

$$C_j^{n+1}*\Delta x=C_j^n*\Delta x+C_{s\,j_{in}}^{n\to n+1}-C_{s\,j_{out}}^{n\to n+1} \quad (6)$$

$$C_{s\,j_{out}}^{n}=C_j^n*u_{out}*\Delta t \quad (7)$$

2.3. FD approximation for Decay term in ADRE

First order reaction term (Decay) in transport equation is as follows. λ is First order reaction term coefficient, [T^{-1}],C is dissolved concentration, [ML^{-3}] and t is time, [T].

$$\frac{\partial C}{\partial t}=-\lambda C \quad (8)$$

Finite difference approximation of reaction term is:

$$\frac{C_j^{n+1}-C_j^n}{\Delta t}=-\lambda C_j^n \quad (9)$$

There are different ways of considering reaction terms and solving the ADRE. First Order, Lax, Quickest, Order3 and Order4 (Abbott and Basco, 1990; Hirsch, 2007) are some of these methods which are applied in this model. We use finite volume Order4 method to reach a more accurate model.

3. Site description

The landfill examined in this paper is located 16 km south of Beirut (Lebanon) and 4 km inland at an average altitude of 250 m above mean sea level. The landfill, once the site of an abandoned quarry, is planned for development over an area of 20–27 ha approximately, and receives 1700–2100 ton/day of waste generated from the

Beirut area and its surroundings. The landfill will have an active life of 10 years and the final waste height may reach 100 m, making it one of the deepest in the world. Long term monthly meteorological data were taken from the Beirut International Airport (BIA) and the American University of Beirut (AUB) weather monitoring stations located within 15 and 20 km from the site, respectively. Total annual precipitation was 760 mm/year with average temperature, wind, and humidity of 21°C, 4 m s^{-1} and 63%, respectively (Bou-Zeid and El-Fadel, 2004).The landfill consists of three cells with different areas and capacities (Table 2).

Table 2. Areas and capacities of landfill cells

Cell	Area (m2)	Expected waste capacities (ton)
1	75,000-77,800	1,362,167-1,725,000
2	52,609-138,000	928,108-5,580,000
3	63,800-124,000	1,009,725-4,800,000
Total	**194,209-262,000**	**3,300,000-12,105,000**

4. Modeling approach

Leachate migration assessment typically involves two steps. First, leachate generation and infiltration through the landfill liner is quantified, then the migration of contaminants is modeled or measured in the porous subsurface until the point of compliance (the point where pollution level is to be assessed). The second step is presented in this paper. The theory and governing equations of flow and transport in porous media has been the subject of extensive work, particularly in the past two decades, in response to problems arising from subsurface contamination. Numerous analytical or numerical models have been developed to simulate leachate flow and transport in the subsurface (El-Fadel et al., 1997b, US EPA, 1993). All these models solve mass, momentum and heat transport equations; however, model capabilities and solution schemes may differ widely. In this study a FORTRAN program was developed to model the landfill based on finite volume-finite difference methods and explicit-implicit solution of governing equation. Programming is based on two-dimensional numerical model for the analysis of dissolved material transport in subsurface transport modeling. The model simulates transport processes under steady state condition. It can simulate confined or unconfined, isotropic, homogeneous aquifers, fully saturated media, single or multi phase systems.

4.1. Leachate generation

In this paper, only the results that were used in subsurface transport simulations are presented. These results represent the baseline scenario likely to occur in view of the site characteristics. The landfill life was divided into three periods. The first period spans the first three years of the operational life of the site when cell 1 is open; this cell has a different configuration than the rest of the landfill and is expected to produce more infiltration. The second period extends between years 3 and 10; cells 2 and 3 are operational during that period while cell 1 is closed and capped. Fig. 2 is a cross sectional view of the landfill depicting the different layers in the three cells. The third period starts at year 10 when all cells are closed and the final cap of the landfill is installed. Figures 3-a and 3-b present the simulated leachate generation and infiltration into the subsurface from the landfill for the three periods, respectively.

4.2. Modeling domain

The geologic formations at the site date back to the cretaceous age. They consist of weathered number carbonaceous rocks including marls, marly limestone, dolomitic limestone, fossiliferous limestone and occasional sandstones. Perched groundwater was located beneath the site at depths as low as 15mbelow ground level; however, the main groundwater table lies at around 220 m below ground level, around 20–30 m above sea level. The general groundwater flow direction is westward towards the Mediterranean Sea with an approximate gradient of 0.05 (Bou-Zeid and El-Fadel, 2004). This indicates that locations that might be adversely affected by the landfilling activity include water wells along the flow path from the landfill to the seashore. The nearest population center to the disposal site is located 2.5 km down gradient. Fig. 4 presents a general schematic view of the simulated domain.

4.3. Modeling process, input data, and boundary conditions

The selection of the contaminants to be modeled was based on the corresponding concentrations in site specific leachate samples, susceptibility to natural attenuation, and drinking water standards (Bou-Zeid and El-Fadel, 2004). An initial screening was conducted assuming no attenuation in the unsaturated zone. The screening revealed that Kjeldahl Nitrogen (K-N), Manganese (Mn), and Iron (Fe) would be the most critical indicators. Kjeldahl-N was retained as the main indicator since it is less affected by attenuation and retardation mechanisms than the other indicators and its concentration in the leachate remains relatively high (Kruempelbeck and Ehrig, 1999). Note that the Lebanese drinking water standards indicate a maximum allowable concentration of Kjeldahl-N of 1 mg/l (Bou-Zeid and El-Fadel, 2004). The trends of the parametric sensitivity analysis for Kjeldahl-N should be valid for other pollutants.

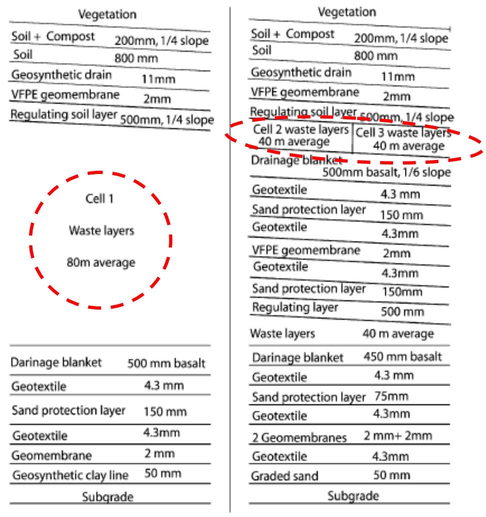

Figure 2. Cross sectional view of landfill operation (After Bou-Zeid and El-Fadel, 2004)

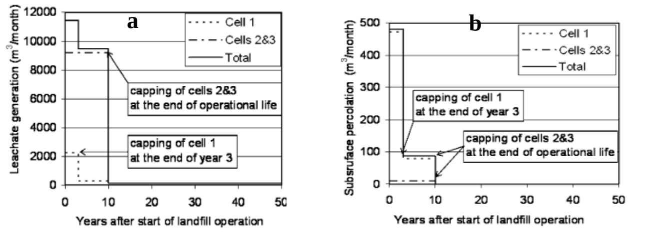

Figure 3. a) Leachate generation, **b)** Subsurface infiltration (After Bou-Zeid and El-Fadel, 2004)

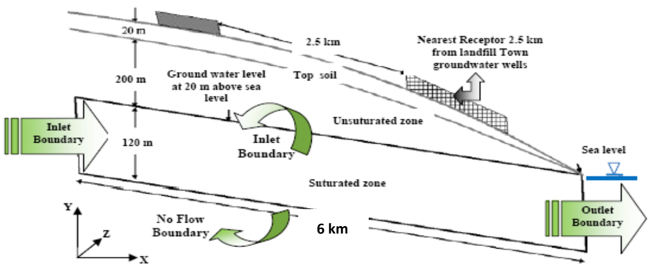

Figure 4. cross sectional of simulated domain (After Bou-Zeid and El-Fadel, 2004)

4.3.2. The saturated zone

The unconfined aquifer, which has an average thickness of 120 m approximately, is underlain by an aquiclude that forms a no-flow boundary condition for water and contaminants. The inputs parameters for the baseline scenario are summarized in Table 3. Leachate flow rate through the landfill base becomes subsurface infiltration. Subsurface infiltration decreases with capping of landfill cells (Fig. 3-b). An initial K-N concentration of 2500 mg/l in the subsurface infiltration is taken from (Bou-Zeid and El-Fadel, 2004). Concentrations are assumed to decrease to reflect contaminant attenuation in the landfill (Table4). The X-axis is from the site towards the sea; the Y-axis is from the bottom to the top of the aquifer. Elements are geometrically uniform in the X and Y directions. The mesh size is Δx=10 m, Δy=2m and Δt=0.2 day. The bottom and vertical sides parallel to the stream velocity are set as no flow boundaries. The top and up streams are inlet boundaries while the downstream side is an outlet boundary.

5. Results

5.1. Base line scenario

Model simulations and a series of sensitivity analysis were conducted. Sensitivity analysis included variations in model parameters such as hydraulic gradient, aquifer hydraulic conductivity, source strength, diffusivity and longitudinal and transverse dispersivities. Figure 6 illustrates concentration distribution contours for 5, 25, 50, and 75 years after the leachate reaches the ground water table. Note that the contour for the drinking water standard is far from the receptor location. This indicates that, for the base scenario, the potential contamination is confined within several hundred meters of landfill boundary.

Table 3. Input parameter to the simulating program

Parameter	Base value
Thickness (m)	120
Saturated hydraulic conductivity(m/s)	5*10-4
Gradient (m/m)	30/6000
Total porosity (%)	15
Effective porosity (%)	12
Diffusivity in water (m^2/year)	0.06
Longitudinal dispersivity (m^2/year)	0.6
Transverse dispersivity (m^2/year)	0.06
Background contaminant level (kg/m^2)	0

Table 4. Assumed variation of leachate source strength with time

Time (yr)	Flow (m/ye)	Kjeldahl-N concentration (mg/l)
0-3	0.022	2500
3-10	0.01	1500
10^{+}	0.005	1000

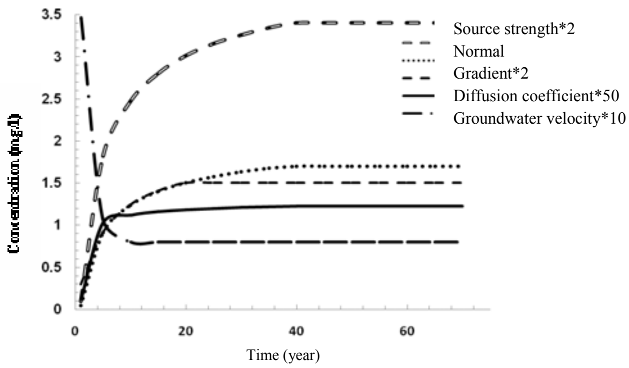

Figure 5. Sensitivity of simulated Kjeldahl-N concentration at the border of the landfill site to changes in input parameters.

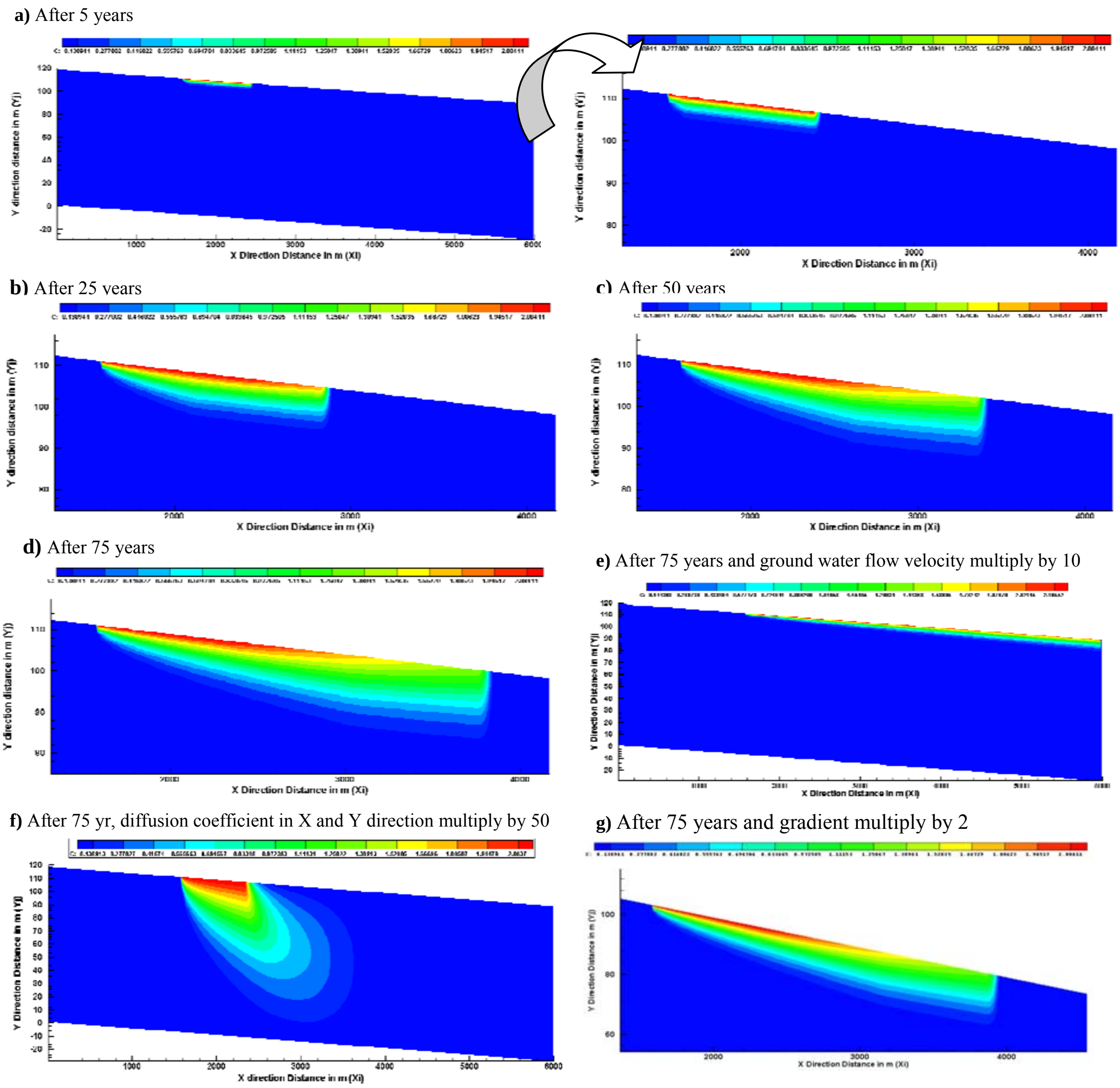

Figure 6. Concentration distribution contours for 5, 25, 50, and 75 years after leachate reaches the ground water table

5.2. Sensitivity analysis

A sensitivity analysis was conducted to assess the effect of model parameters variation on contaminant transport. Dispersivities in the longitudinal and transverse direction were varied simultaneously. While higher groundwater velocities increase the speed of the plume spread, they increase dilution ratio and hence tend to decrease the concentration. The effect of increasing dispersivities (by a factor of 50) is to enhance transport in the transverse direction, this leads to a wide but short plume as depicted in Fig.6-f. Increasing the Groundwater velocity of the aquifer considerably reduces contaminant concentration due to increased dilution (Fig.6-e). The temporal variation of the pollution levels is another aspect that is of significance when potential pollution from landfills is assessed. So the effect of varying model parameters on the history of concentrations was assessed. Fig. 5 shows a typical concentration history pattern. Dispersivity increscent by a factor of 50 reduces concentrations. Doubling the source strength produces a predictable increase in concentrations. The increase in hydraulic gradient consistently reduces concentrations in the vicinity of the site due to higher velocities and dilution ratios.

5. DISCUSSION AND CONCLUSION

The sensitivity of the model to variation in input parameters indicates that while higher groundwater velocities increase the speed of plume spread, they also increase the dilution ratio and hence decrease the concentration. The most significant changes in pollution patterns were associated with changes in dispersivities, partition coefficient, source strength, and groundwater flow velocity and it is indicating that these parameters should be carefully selected when similar modeling studies are performed. Concentration distribution contours for 5, 25, 50, and 75 years after leachate reaches the ground water table shows that even after 75 years the contaminant won't reach the downstream recipient t so it fulfill the water contaminant limitation for K-N and can be used as drinking water. The unavailability of site-specific groundwater flow measurements to calibrate the model presents some limitations on the quality of the results.

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