

# An investigation of the potential environmental contamination from the leachate of the Rasht waste disposal site in Iran

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**Abstract** This research deals with detecting the underground layer geometrics and conditions, probable faults and crushed zones, thickness of alluvium, determination of pollutant plume development, and groundwater quality in the study area. The Visual HELP and HYDRUS programs were used to evaluate hydrologic properties and contaminant transport, respectively. All required data which consisted of the geotechnical study (drilling the borehole and dug well), the geophysical study (geoelectrical sounding), and geochemical analysis of water and leachate samples were collected in the field operations. According to the results of ten years of hydrologic data evaluation of this site by using the Visual HELP program, the mean rate of leachate percolation from this site was equal to 1.095 m per year. From the interpretation of the vertical electrical resistivity soundings curves which was calibrated by the geotechnical borehole log, a four-layer geoelectrical model was recognized. These layers consisted of surface layer (loose and fine grained topsoil), medium grained alluvium, saturated layer, and high resistivity bedrock. The physico-chemical analysis of ten surface and groundwater sample indicated that these samples were not contaminated by leachate yet. Based on 1D, 2D and 3D modeling of contaminant transport using the HYDRUS 1D and the HYDRUS 2D/3D program, during the ten years of the onset of leachate seepage, contaminant plume migrated a short distance (<200 m) due to high compaction of fine grain sediments. In addition, the high compaction of a third layer (M3) in a depth of more than 40 m introduces a natural

barrier to infiltration and movement of contaminant to the deep subsurface.

**Keywords** Waste disposal site · Visual HELP · HYDRUS · Contaminant · Leachate · Rasht · Iran

## Introduction

Exposures to environmental contaminants are significant risk factors in human health and disease. To understand and manage these risk factors, environmental and public health managers must have knowledge of the source of the exposure, the fate of contaminants, exposure levels, and routes of the exposure for such contaminants (Jhamnani and Singh 2009). Municipal solid waste (MSW) has already become one of the main factors which adversely influences the environment during the course of city construction and growth. Landfills have been identified as one of the major threats to groundwater resources; areas near landfills have a greater possibility of groundwater contamination because of the potential pollution source of leachates originating from the nearby site (Nixon et al. 1997; Aldecy et al. 2008). Such contamination of groundwater resources poses a substantial risk to the local resource user and to the natural environment (Mor et al. 2006). Municipal landfill leachates are highly concentrated complex effluents which contain dissolved organic matters; inorganic compounds, such as ammonium, calcium, magnesium, sodium, potassium, iron, sulphate, chloride and heavy metals such as cadmium, chromium, copper, lead, nickel, zinc, among others (Lee et al. 1986; Christensen et al. 1998; Ogundiran and Afolabi 2008).

The produced leachate from waste disposal sites can migrate to the surface and groundwater. The resultant

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contamination brings a latent hazard, which cannot be ignored, to the cities. Currently, many MSW landfills cannot meet environmental standards and present a serious threat to the environment. Leachate from a MSW landfill is a latent pollution source and the occurrence of leachate leakage, due to poor management can result in serious environmental contamination in the surrounding areas of waste disposal sites. Therefore, numerical simulation of the environmental pollution process from MSW landfill leachate is important to the evaluation of environmental risk associated with MSW landfills, to understand the mechanisms that control leachate formation, quality, quantity, most importantly migration characteristics (El-Fadel et al. 1997b) and also to the environmental pollution control process. With the MSW problems being increasingly prominent, the occurrence of groundwater contamination is increasingly frequent; accordingly, the study of groundwater contamination resulting from MSW landfill leachate has become an increasingly focused issue. Many researchers have carried out experimental and theoretical studies on MSW landfill leachate and groundwater contamination, including the mechanisms driving leachate generation, its translation and transport process, its quality and environmental effects, pollution range, distribution characteristics and its fate in the environment. Incidents of groundwater contamination by landfill leachate have been widely reported since the early 1970s (Zanoni 1972; Garland and Mosher 1975; Dunlap et al. 1976; MacFarlane et al. 1983; Reinhard et al. 1984; Albaiges et al. 1986; Butow et al. 1989; El-Fadel et al. 1997a; Malina et al. 1999; Longe and Kehinde 2005; Longe and Enekwechi 2007).

Recently, solid waste has been one of the major contributors to the environmental problems in Iran. The problem of groundwater contamination by waste disposal sites is steadily growing worse in Iran due to the way of managing the municipal solid wastes. Solid waste disposal sites are well known to release large amounts of hazardous and deleterious chemical (leachate). Leachate is produced primarily in association with precipitation that infiltrates through the wastes and normally results in the migration of leachate into the groundwater zone and pollutes it. There is little or no monitoring of the extent of leachate infiltration and its movement into the groundwater at waste disposal facilities in Iran. Hence, the extent of short and long term contamination of groundwater is unknown. In this study, one of the most important active waste dump sites at Rasht province, Iran, is evaluated. In this site, the waste dumping was started in 1983 and, according to municipality reports for 1997, with per capita waste generation in this city, which is about 800 g per day, about 470 metric tons of domestic waste was dumped in this location. Since the Rasht waste disposal sites have the potential to deteriorate

into the ground water system, there is a need to study the degree of pollution in the groundwater and to assess the migration and dispersion potential of pollutant species and their impact on water quality.

Because there was no information about the environment impact of the Rasht waste disposal site, the researchers tried to collect all primary data just during this study for the first time. So, during the main objectives of this study- the hydrological evaluation and contaminant transport modeling of the Rasht waste disposal site—all required and complementary data were collected. For this aim, the authors used the Visual HELP and HYDRUS (1D, 2D and 3D) programs for evaluation of hydrologic properties and contaminant transport modeling, respectively. In this research, all required data were collected in field operations, which consisted of geotechnical study (drilling the borehole and dug wells), geophysical study (geoelectrical sounding), geochemical analysis of water and leachate samples, and field excursion for collecting supplementary data.

## Materials and methods

### Water flow and contaminant transport in the unsaturated zone

The unsaturated zone is often regarded as a filter removing undesirable substances before they affect aquifers, and the hydrogeological properties of the unsaturated zone are the most important factor for groundwater deterioration induced by surface contamination (Stephens 1996; Selker et al. 1999). The knowledge and understanding of water flow and solute transport in the unsaturated zone is becoming increasingly important, especially in terms of water resources planning and management, water quality management, and the mitigation of groundwater pollution. The presence of different phases (soil/sediment matrix, water, solute and air) results in many different physical and chemical processes taking place. These processes are often complex and require simplifying assumptions to provide achievable and verifiable results in terms of water and solute solution profiles in the shallow sediments (Selker et al. 1999; Gandola et al. 2001).

### Water flow

The process of uniform variably saturated flow is generally assumed to be governed by Darcy's law and the conservation of mass. The theoretical basis for the equation describing saturated and unsaturated flow and solute transport has been well documented in the literature (for example; Bear et al. 1995; Zheng and Bennett 1995;

Domenico and Schwartz 1998; Sudicky 1998; Radcliffe and Šimůnek 2010). The uniform variably-saturated water flow is described using the Richards equation (Šimůnek et al. 2011):

$$\frac{\partial \theta(h)}{\partial t} = \frac{\partial}{\partial x_i} \left( K(h) \left( K_{ij}^A \frac{\partial h}{\partial x_j} + K_{iz}^A \right) \right) - S(h), \tag{1}$$

where  $\theta$  = volumetric water content ( $L^3 L^{-3}$ );  $h$  = pressure head (L);  $K$  = unsaturated hydraulic conductivity ( $L T^{-1}$ );  $K_{ii}^A$  = components of a dimensionless anisotropy tensor  $K^A$  (which reduces to the unit matrix when the medium is isotropic);  $S$  = general sink/source term ( $L^3 L^{-3} T^{-1}$ ) accounting for root water uptake;  $t$  = time (T); and  $x_i$  = spatial coordinate (L).

Because of the strongly nonlinear make-up of the Richard equation, only a relatively few simplified analytical solutions can be derived. Most practical applications of Eq. (1) require a numerical solution, which can be obtained using a variety of methods such as finite differences or finite elements. Equation (1) is generally referred to as the mixed form of the Richards equation since it contains two dependent variables (i.e., the water content and the pressure head). Solutions of (1) require knowledge of two soil hydraulic functions describing the soil water retention and hydraulic conductivity properties of the porous material. The soil water retention curve (or also called the soil water characteristic curve, the capillary pressure-saturation relationship, or the pF curve) describes the relationship between the water content and the pressure head.

The HYDRUS software (Šimůnek et al. 2011) implements the soil-hydraulic functions of van Genuchten (1980) who used the statistical pore-size distribution model of Mualem (1976) to obtain a predictive equation for the unsaturated hydraulic conductivity function in terms of soil water retention parameters. The expressions of van Genuchten (1980) are given by Eqs. (2 and 3):

$$\theta(h) = \begin{cases} \theta_r + \frac{\theta_s - \theta_r}{[1 + |\alpha h|^n]^m} & h < 0 \\ \theta_s & h \geq 0 \end{cases} \tag{2}$$

$$K(h) = K_s S_e^l [1 - (1 - S_e^{1/m})^m]^2, \tag{3}$$

in which  $\theta_r$  and  $\theta_s$  denote the residual and saturated water content (–), respectively. The  $K_s$  is the saturated hydraulic conductivity ( $L T^{-1}$ );  $\alpha$  is the inverse of the air-entry value (or bubbling pressure) ( $L^{-1}$ ).  $n$  is a pore-size distribution index (–), and  $l$  is a pore-connectivity parameter assumed to be 0.5 (–).

Solute transport

While water flow in the various software packages is simulated in a relatively similar manner (i.e., using the Richards equation to describe uniform flow), different

solute transport modules exist in the different codes. Some modules simulate single-component transport, while others contain biogeochemical modules for simulating potentially very complex interactions between the various solutes. In this section we provide brief descriptions of the HYDRUS transport modules (Šimůnek et al. 2011). Solute Transport (Convection-Dispersion Equation) flow is described using the Eq. (4):

$$\frac{\partial(\rho s)}{\partial t} + \frac{\partial(\theta c)}{\partial t} = \frac{\partial}{\partial z} \left( \theta D \frac{\partial c}{\partial z} + q c \right) - \phi, \tag{4}$$

in which  $c$  is the solution concentration ( $M L^{-3}$ ),  $s$  is the adsorbed concentration ( $M M^{-1}$ ),  $\theta$  is the water content ( $L^3 L^{-3}$ ),  $\rho$  is the soil bulk density ( $M L^{-3}$ ),  $D$  is a dispersion coefficient ( $L^2 T^{-1}$ ),  $q$  is the volumetric flux ( $L T^{-1}$ ) and  $\phi$  is the rate constant representing reactions ( $M L^{-3} T^{-1}$ ).

HELP program

The Hydrologic Evaluation of Landfill Performance (HELP) model is a quasi-two-dimensional, deterministic, water-routing model for determining water balances. The HELP model requires daily climatologic data, soil characteristics, and design specifications to perform the analysis. This program assumes Darcian flow for vertical drainage through homogeneous, temporally uniform soil and waste layers. Percolation through soil liners is modeled by Darcy’s law, assuming free drainage from the bottom of the waste dump in the disposal site. Vertical drainage is assumed to be driven by gravity alone and is limited only by the saturated hydraulic conductivity and available storage of lower segments. In the HELP model the unsaturated hydraulic conductivity is computed by the Campbell

**Table 1** Separated blocks from the Rasht waste disposal site

Block no.	Area (hectare)	Waste thickness (m)	Angle of block bottom
1	6.5	45	25
2	2	15	18
3	2.8	20	15

**Table 2** Hydrologic properties of wastes

Vertical percolation parameters of municipal waste	
Total porosity (vol/vol)	0.67
Field capacity (vol/vol)	0.29
Wilting point (vol/vol)	0.07
Sat. hyd. conductivity (cm/day)	86.4
Subsurface inflow	0

**Table 3** Hydrologic parameters of three blocks of the Rasht waste disposal site in ten years of evaluation

Block no.	Measurement unit	Precipitation	Runoff	Evapotranspiration	Soil water	Percolation or leakage
1	Volume (m <sup>3</sup> )	869,170	453.23	137,200	9,227,400	694,950
	Rate (m)	13.37	0.69	2.11	141.96	10.69
2	Volume (m <sup>3</sup> )	267,440	162.41	42,318	563,080	223,370
	Rate (m)	13.37	0.08	2.11	48.15	11.1
3	Volume (m <sup>3</sup> )	374,410	203.46	59,205	1,787,900	310,440
	Rate (m)	13.37	0.007	2.11	63.85	11.08

**Table 4** Summary of physical properties of geological structures at the waste disposal site

Properties	Value
Water content (%)	8–12
Poisson coefficient	0.3–0.4
Permeability (cm/s)	10 <sup>-3</sup> –10 <sup>-4</sup>
Elasticity module (kg/cm <sup>2</sup> )	650–750
Specific weight in dug wells (g/cm <sup>3</sup> )	1.8
Mean particles density of borehole materials (g/cm <sup>3</sup> )	2.72

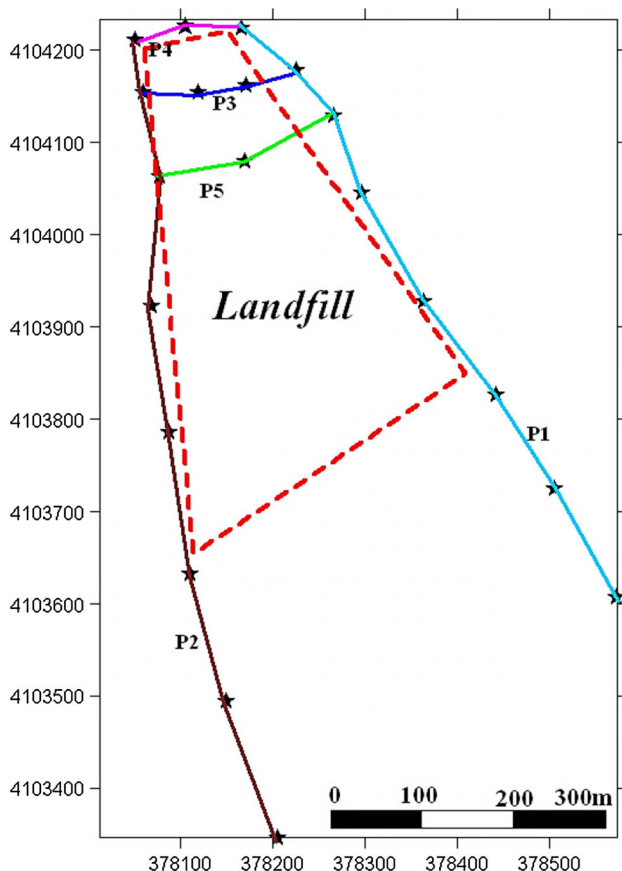
hydraulic equation using Brooks-Corey parameters. In this study, the main application of Visual HELP was determination of the leachate production rate (Schroeder et al. 1994).

#### HYDRUS program

HYDRUS numerically solves the Richards equation for saturated-unsaturated water flow and convection-dispersion type equations for solute transport. The governing flow and transport equations are solved numerically using Galerkin-type linear finite element schemes. The governing convection-dispersion solute transport equations are written in a very general form by including provisions for nonlinear non-equilibrium reactions between the solid and liquid phases, and a linear equilibrium reaction between the liquid and gaseous phases. The transport models also account for convection and dispersion in the liquid phase, as well as for diffusion in the gas phase, thus, permitting one to simulate solute transport simultaneously in both the liquid and gaseous phases. In this study, water flow and contaminant transport are investigated in the three states of 1D, 2D and 3D by HYDRUS software (Šimůnek et al. 2011).

#### Study area

The solid waste disposal site of Rasht city is located in Saravan County, at latitude 37° 04'N and longitude 49° 37'E. This site covers about 12 hectares in the area where the geological formation consists of mainly thin layers of dense silty sand with gravel and cobble, and very dense crushed rocks. In this area, there are not thick and extended alluvium deposits, so this area cannot be considered as an aquifer. The topography in the vicinity of the waste disposal site is generally rough, with local elevations at the site ranging from a high of 320 m above sea level to a low of 100 m. This area has the close structure so that it can be considered as a separate watershed. The climate of the area is Mediterranean which is characterized by temperature (mean annual of ≈ 16 °C), and relative humidity (≈ 82 %). This area is one of the wettest areas in Iran because of its average annual rainfall (≈ 1,360 mm). Generally, the

**Fig. 1** The location of VES stations and profiles in the study area

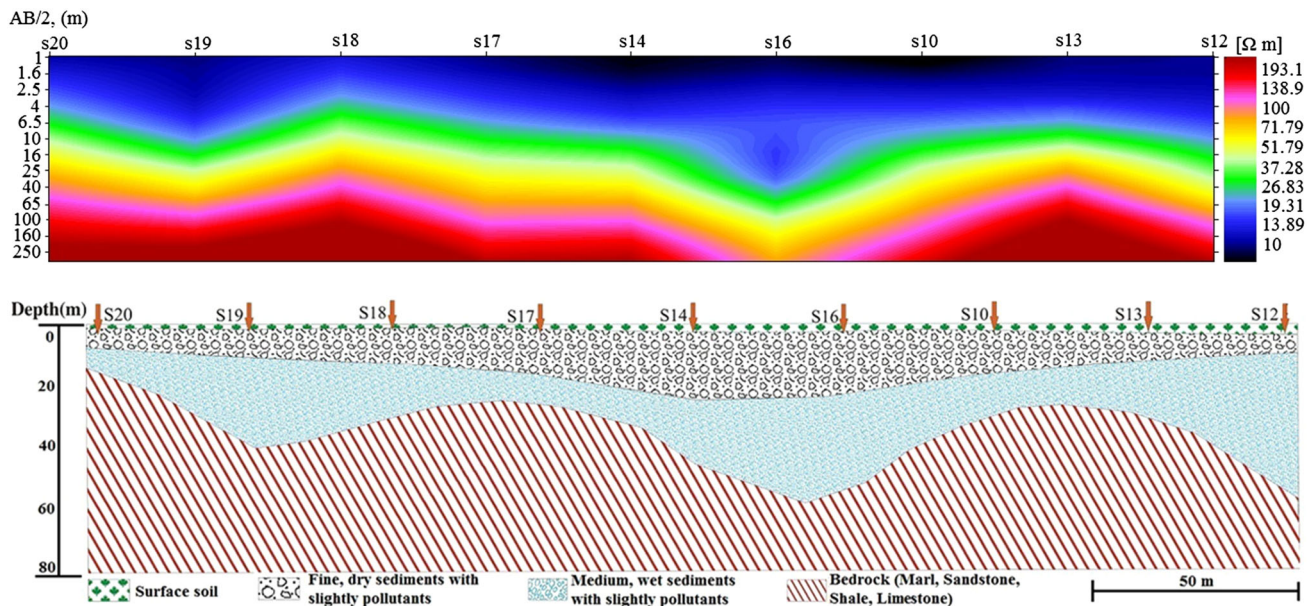


Fig. 2 Geoelectrical pseudo-section and interpreted geological section of profile N.2. (For example)

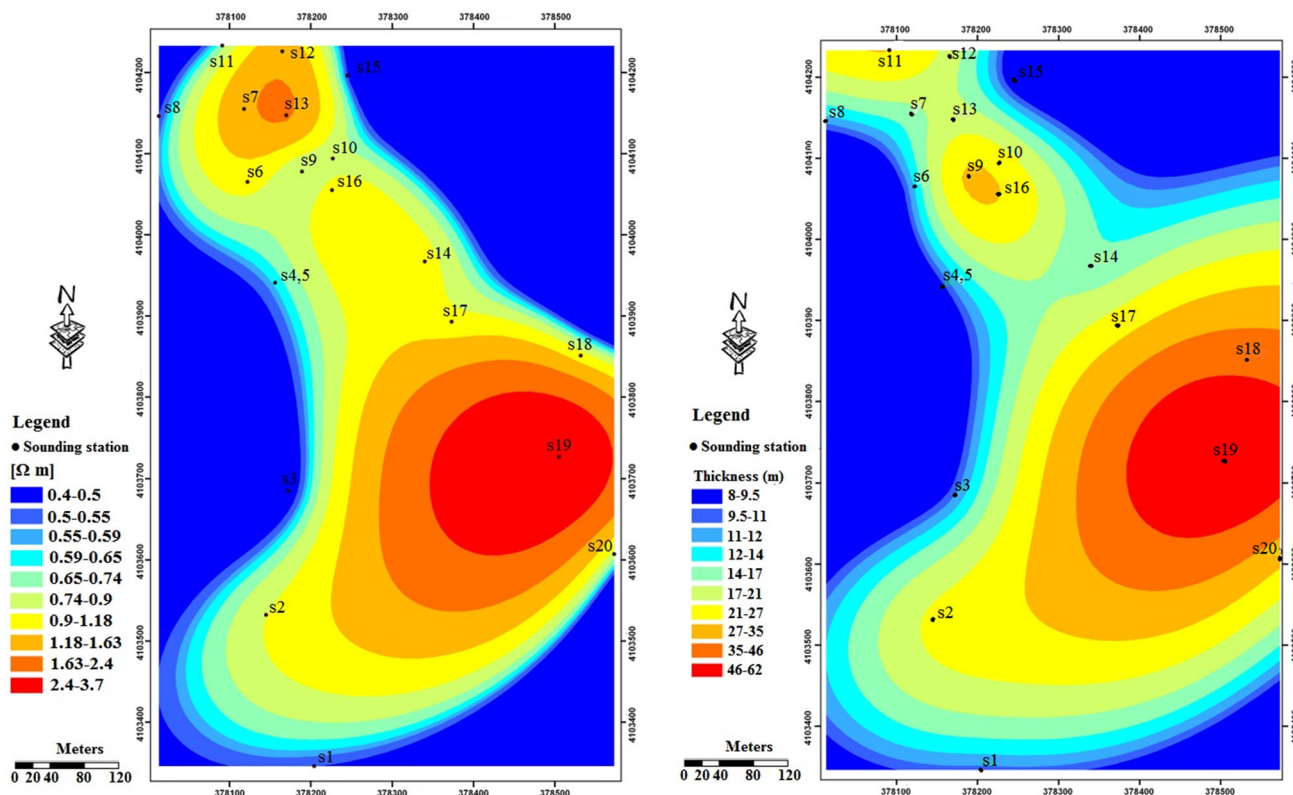
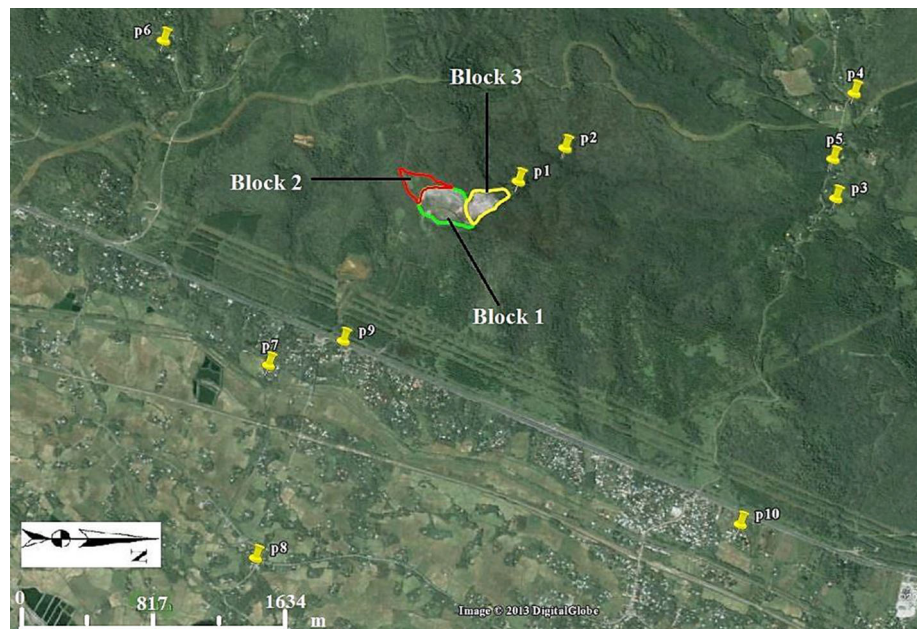


Fig. 3 Iso-resistivity map of first layer (left) and Iso-thickness map of second geoelectrical layer (right)

**Fig. 4** Position of sampling points and separated blocks of the waste disposal site



regional pattern of groundwater movement is controlled by topography. Hence, according to the topography, the approximated regional groundwater flow direction is dominantly towards to the north of the study area (to the Siahroud River).

## Results and discussion

### Hydrological evaluation of site

In the first step of this study, the researchers focused on hydrological evaluation of this site by using the Visual HELP program. In this part, construction of a site model for entering the data and analysis of its performance was necessary. In the study area, because of rough topography and heterogeneous structure (different slope of bottom and also thickness of waste), this site divided into three discrete blocks (Table 1).

The presented properties in Table 1 are based on visual observations by the authors, field measurements, analysis of the satellite images, available reports and several interviews and meetings with responsible persons in the municipality of Rasht city. The mentioned blocks cover about 11.3 hectare of the study area (Fig. 4). For each block designation in the HELP model, only the thickness of waste was used. Determination of the rate of leachate leakage from the bottom of the waste disposal site was the main aim of this part of the study as an introduction to contaminant transport in the next part. The hydrologic properties of wastes and relevant parameters to vertical leachate leakage are presented in Table 2.

The main climatic data which was used in the Visual HELP program consisted of precipitation, temperature and solar radiation. These data were collected from the Rash synoptic station for ten years (2002–2012). After model design for three blocks, the HELP program was run for each block separately. The results are presented in Table 3. As shown, the mean rate of leachate percolation from this site is equal to 1.095 m per year. The rate of recharge to the waste disposal site is assumed to be greater than that to the surrounding area because the sand and gravel used as cover material allows rapid infiltration and minimal evapotranspiration. Channeling of water through the refuse has allowed leachate to move down toward the base of the waste disposal site even though the capacity of the material to retain moisture may not yet have been reached.

### Geotechnical evaluation of the site

In this part, determination of the physical and geotechnical properties of the geological structure of this site was followed. For this aim, one geotechnical borehole and two dug wells were excavated in northern and southern parts of the disposal site, respectively. The depth of the geotechnical borehole was 35 m (with a 95–112 mm diameter) and two dug wells excavated to depths of 4 m and 2 m. According to soil and rock cores which were prepared from the borehole, and based on unified soil classification, the soil type at a depth of 0–13 m is SM (silty sand with gravel), at a depth of 13–20 m is very dense and dry crushed rock, and at a depth of 20–35 m is the high compaction SM type. There was no evidence of water or sharp increase in soil moisture in the borehole and dug wells. A

**Table 5** Coordinates of the sampling locations and their direct distance to the waste disposal site

Sampling point	Sample source	Position	Direct distance to disposal site (m)
p1	Leachate	37°04'28"N, 49°37'48"E	10
p2	Leachate	37°04'37"N, 49°37'39"E	365
p3	Leachate	37°05'32"N, 49°37'49"E	1,980
p4	River	37°05'35"N, 49°37'22"E	2,140
p5	Deep well	37°05'31"N, 49°37'39"E	2,000
p6	Deep well	37°03'16"N, 49°37'16"E	1,670
p7	Deep well	37°03'39"N, 49°38'36"E	1,400
p8	Deep well	37°03'38"N, 49°39'25"E	2,420
p9	Deep well	37°03'54"N, 49°38'29"E	960
p10	Deep well	37°05'15"N, 49°39'12"E	2,500

summary of the physical properties of the geological structure in the waste disposal site is presented in Table 4.

#### Geoelectrical evaluation of the site

Geoelectrical methods have an important, albeit difficult role, to play in landfill investigations. In the present economic conditions, with the environmentally sensitive regime, model development is essential in gradients for a successful site investigation of landfills (Meju 2000; Lashkaripour and Nakhaei 2005; Svensson 2008; Teixidó 2012). This research project deals with detecting the underground layers geometrics and conditions, probable faults and crushed zones, thickness of alluvium, determination of pollutant plume development, and groundwater quality in the study area. The resistivity survey was completed with 24 Vertical Electrical Soundings (VES) by the Schlumberger array in five profiles include 4–9 sounding stations, with a maximum current electrode spacing (AB) ranging from 600 to 800 m, in two stages. The profile spacing was 400 m and sounding spacing was 50 m. Position and extension of all the VES's are indicated in Fig. 1. Five geoelectrical sections were drawn along the profiles. The apparent resistivity measurements were made with GEOB 2003 equipment that is light and powerful for deep penetration. The field curves were interpreted by the well-known method of curve matching with the aid of the Russian software IPI2WIN.

VES success must rely on the careful interpretation and integration of the results with the other geologic and hydro-geologic data for the site. Therefore, lithological information obtained from the geotechnical borehole log could be used to calibrate the VES field curves. Where test hole-log information was available, the solution to automatic interpretation procedure was constrained by keeping known layer thicknesses constant during the program computations. Measuring the current penetration depth was calculated to be  $0.5001(AB/2)^{0.998}$  by a correlation between depth and AB/2 graphs. It is equal to approximately AB/4. From the interpretation of the resistivity curves, a four-layer resistivity and thicknesses indicated four subsurface layers (Fig. 2, for example). These layers consisted of a surface layer (loose and fine grained topsoil), medium grained alluvium, a saturated layer, and high resistivity bedrock.

Depth and thickness of sub-surface layers were identified and the extent of the probable aquifer and type of bedrock were also indicated. Average thickness and resistivity of the thin, organic pollutant-rich, top layer were calculated in the upstream as 2 and  $2\Omega$ -m and in the downstream as 7 m and  $5\Omega$ -m, respectively (Fig. 3, left). The aquifer thickness increases towards the downstream and central parts of the adjacent valley. The average thickness of the low resistivity saturated alluvial aquifer in the central and eastern part has been estimated to be about 42 m and in the west about 10 m (Fig. 3, right). Also, we can separate two distinct areas in the east and west sides based on the difference of electrical resistivity. The bedrock of the aquifer shows different resistivity values due to two types of bedrock and respect to same degree of saturation and values of small tectonically fractures with the NE-SW direction. Bedrock generally consists of sandstone and mudstone, but in some parts it has appeared as limestone and intermediate volcanic rocks.

#### Hydrochemical evaluation of site

In this part of the study, the water (surface and groundwater) and leachate samples were analyzed for testing the amount of contaminants and toxic elements. The ten samples were collected randomly from different distances from the center of the waste disposal site (to a maximum direct distance 2,500 m) during November, 2012 (Fig. 4). A Global Positioning System (Garmin GPS model) was used to take the coordinates of the sampling locations (Table 5).

The samples were collected according to standard methods, in 1 liter polyethylene bottles and were to be properly sealed, and analyzed for 26 physical and chemical parameters (APHA 1998). Samples analysis was done in the Gilan Regional Water Corporation (GRWC) laboratory.

**Table 6** Physical and chemical properties of water and wastewater samples

Parameter	Unit	p1	p2	p3	p4	p5	p6	p7	p8	p9	p10
pH	–	8.58	8.59	8.67	8.57	7.83	8.12	8.21	7.66	7.65	7.49
Turbidity	FNU	290	330	240	0.58	0.61	0.53	0.71	1.02	0.95	1.05
Color	HAZEN	12,700	9,200	4,600	12	15	14	9	8	23	26
Temp.	°C	25	25	24	24	24	24	24	24	24	24
EC	µs/cm	21,900	21,000	9,700	360	597	584	882	1,380	894	1,428
TDS	mg/l	13,797	13,220	6,111	227	376	368	556	869	563	900
Ca	mg/l	76,000	76,000	40,000	48	62	90	98	96	68	90
Mg	mg/l	45,600	45,600	21,600	7.2	21.6	7.2	9.6	44.4	43.2	44.4
Na	mg/l	202,400	188,600	87,400	9.89	21.62	13.34	69.92	101.1	37.95	128.8
K	mg/l	148,200	148,200	62,400	1.56	3.51	3.9	1.17	5.07	3.51	12.09
Cl	mg/l	213,000	213,000	71,000	7.1	17.75	21.3	78.1	195.25	78.1	177.5
SO <sub>4</sub>	mg/l	124,800	96,000	86,400	19.2	36.51	38.88	93.6	149.76	61.92	219.84
CO <sub>3</sub>	mg/l	2,000	2,000	2,000	0.1	0	0	0	0	0	0
HCO <sub>3</sub>	mg/l	9,400	9,400	3,800	2.8	4.3	4.4	4.7	4.5	5.3	4.6
TH	mg/l	380,000	380,000	190,000	150	245	255	285	425	350	410
NH <sub>4</sub>	mg/l	2,068	1,780	700	0	0	0	0	0	0	0
NO <sub>3</sub>	mg/l	653	632	260	2.94	1.62	8.93	11.05	0.61	0.023	1.03
NO <sub>2</sub>	mg/l	0	0	0	0	0	0.011	0	0.033	0	0.052
PO <sub>4</sub>	mg/l	60	55	20	0.12	0.42	0.1	0.81	0.05	0.22	0.09
Fe	mg/l	0.6	1	0.3	2.44	1.73	1.37	2.03	2.8	1.37	2.71
Mn	mg/l	0.1	0.38	0.012	1.28	0.97	0.79	1.46	1.51	0.86	1.87
SiO <sub>2</sub>	mg/l	312	240	88	7.14	15.53	10.55	14.71	17.68	20.07	8
H <sub>2</sub> S	mg/l	19	16.8	14	0.02	0.02	0.02	0.03	0.03	0.03	0.03
TSS	mg/l	86	82	34	2	2	2	2	2	2	2
BOD	mg/l	1,092	820	630	18.6	6.4	32.5	7.8	6.5	5.85	4.65
COD	mg/l	5,060	4,350	1,680	50.2	24	10.2	26.8	22.6	18.4	14.1

Results of water and wastewater analysis in ten positions are presented in Table 6.

The present study indicated that the river and groundwater samples (p4–p10) due to disposal of municipal solid waste landfill at Saravan were colorless and odorless and were slightly alkaline in nature with their pH ranging from 7.49 to 7.83 at the p5, p8, p9, and p10 sites. The high alkalinity of samples (except leachate samples) was observed at the p4, p6, and p7 sites.

Leachate samples from the Rasht waste disposal site were rich in sodium, potassium, calcium, magnesium, and chloride (Table 6). The high conductivity of the leachates also reflected the high soluble salt contents. The salt content slightly increases with waste disposal site age as a consequence of the decomposition of organic matter. The TDS values of groundwater sample were 869 at p8 and 900 mg/l at p10 sites. TDS concentration in groundwater samples decreased with the increase in the distance from the landfill sites.

In the landfill, the activity of microorganisms can be increased by time processing and this means that the BOD

**Table 7** The hydraulic parameters (van Genuchten-Mualem parameters) of materials

Material	$\theta_r$	$\theta_s$	$\alpha(1/m)$	$n(-)$	$K_s(m/day)$
Silty sand with gravel and cobble (M1)	0.034	0.41	0.034	1.75	1.02
Crushed and dense rocks (M2)	0.041	0.43	0.033	1.48	0.38
High compaction silty sand (bed rock) (M3)	0.01	0.2	0.001	1.2687	0.000001

(biochemical oxygen demand) can increase in the leachate. The age of the landfill can be determined by the amount of BOD in the leachate. According to the results, the age of the Rasht waste disposal site is located in the medium to old range (Alloway and Ayres 1997). In addition, there was no evidence of leachate contaminant in other samples (p4, p5, p6, p7, p8, p9, p10).

The COD is low in the initial stages of landfills (the normal could reach to over ten thousand mg/l). The



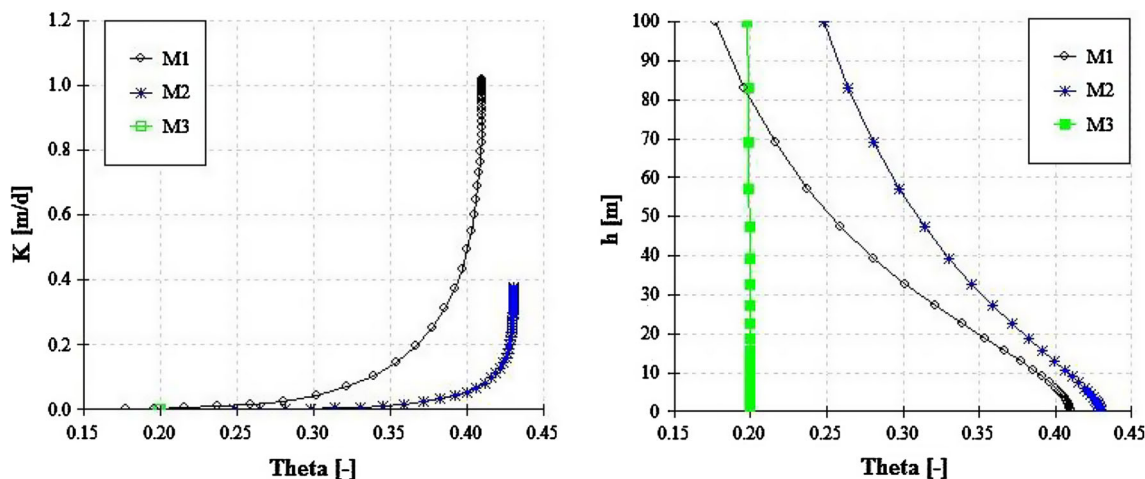


Fig. 5 The soil hydraulic functions  $K(\theta)$  (left) and  $h(\theta)$  (right)

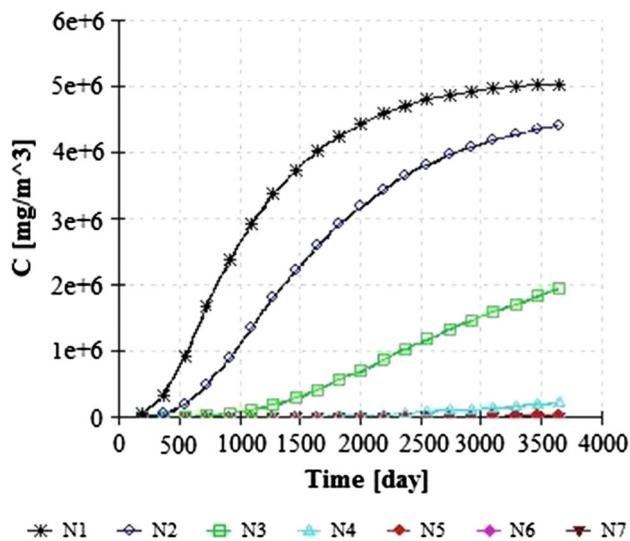


Fig. 6 Chloride concentration breakthroughs in observation nodes

reduction of COD is slow and the decrease of BOD is fast by the time it was processed (Ryding 1992). The result of analysis for COD is similar to BOD so that the surface and groundwater samples were not contaminated by waste leachate. In the collected samples, the COD values were decreased in the groundwater samples as the distance from the solid waste dumping sites increased.

The high concentration of salt in the leachate mostly is chloride. As shown in Table 6, concentration of chloride in leachate samples (p1, p2) is more than in sample p3 which is farther from the leachate source and this indicates that concentration of chloride is reduced by added surface waters to the leachate. Chloride concentrations in leachate and water samples collected during the study ranged from 213,000 to 7.1 mg/L. A value of 213,000 mg/L was used as an average source concentration in the simulations. Results

of chloride concentration in surface and groundwater showed that these samples are not contaminated by leachate.

The N and P are main components within the inorganic pollutant from the leachate. The concentration of N and P is high when the landfill is processing. In this study, the N component concentration was checked in three composites consisting of  $\text{NH}_4$ ,  $\text{NO}_3$  and  $\text{NO}_2$ . The high concentration of ammonium and nitrate was detected only in leachate samples, as the results of the analysis showed.

The low content of heavy metals (Fe, Mn) in leachate was because the domestic waste was not filled together with the industrial waste or sludge in this site. The amount of heavy metals is related to the industrial level of local urban waste and how much industrial waste will be in the land-filling. The domestic waste only contains heavy metals at a low level. The highest iron concentrations in the samples were 2.8 and 2.71 mg/l for p8 and p10, respectively. Also, the highest manganese in all samples were 1.87 and 1.51 mg/l for p10 and p8, respectively. The high concentration of these metals in samples p4–p10 rather than leachate samples showed that their source is not from the leachate of the waste disposal site. The concentration of trace metals (Fe, Mn) in the waste disposal leachates were relatively low (<1 mg/l).

### Contaminant transport modeling

The HYDRUD program was used to simulate (in three states of 1D, 2D and 3D) the advective-dispersive transport of a conservative solute (in this study, chloride) downgradient from the waste disposal site. Solution of the solute-transport equation requires data on distribution of the groundwater head and the specification of boundary and initial conditions. Values for coefficients

Fig. 7 The 2D designed model

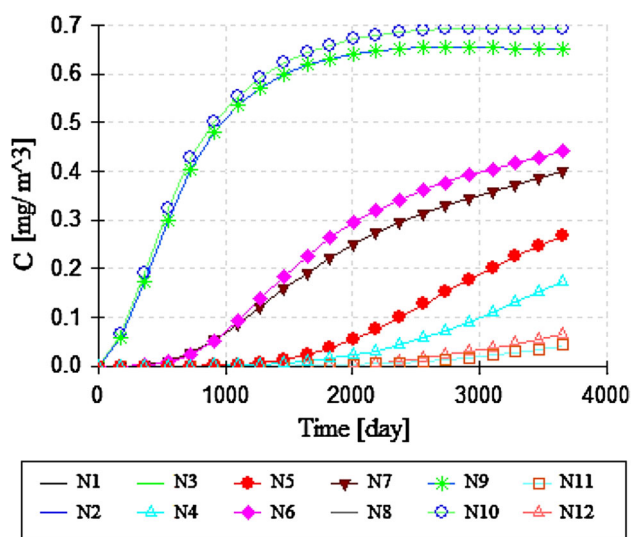
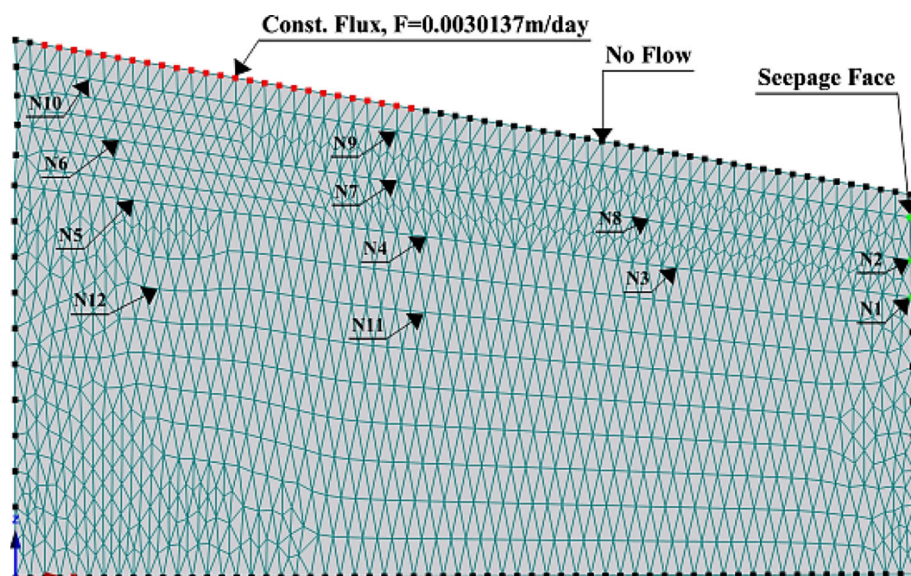


Fig. 8 The concentration breakthroughs in observation nodes

of longitudinal and transverse dispersivity and data on the configuration, rates of inflow, and solute concentrations associated with the contaminant source are also required.

### 1D modeling

In this step, all collected data from previous parts were the basis of model design in three modeling states. The hydraulic parameters and properties of material which are used in models are presented in Table 7 and Fig. 5. In the HYDRUS package, the neural network prediction module ROSETTA (Schaap et al. 2001) was used for determining the hydraulic parameters by using graining tests of borehole core samples.

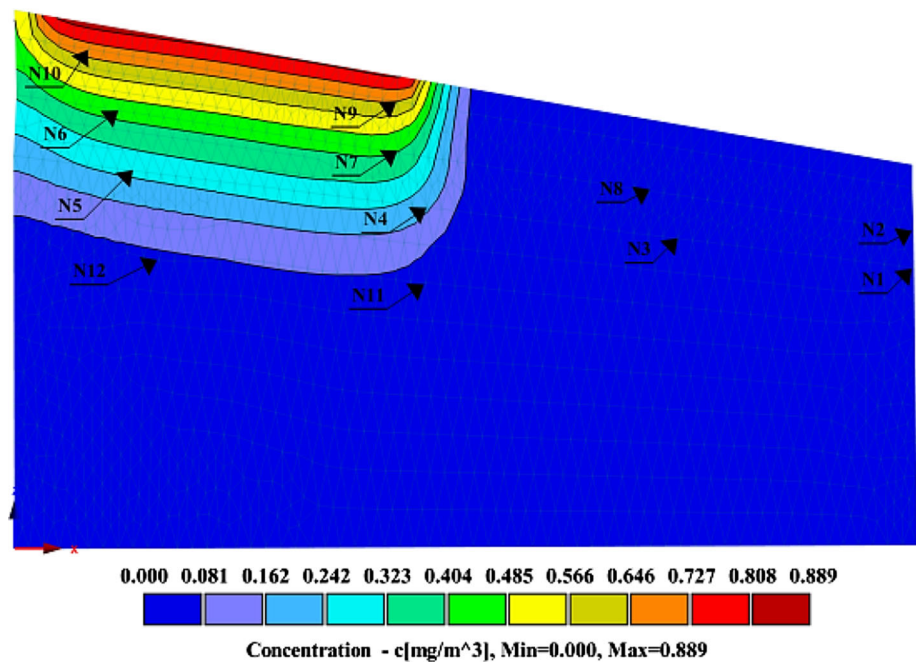
First, the 1D contaminant transport modeling in the waste disposal site was implemented. Seven observation nodes in different depths (15, 20, 30, 40, 50, 70, and 80 m, respectively) were used for analysis of the contaminant transport process. The chloride ion was selected as the conservative solute because it is not subject to adsorption, biological decay, or chemical precipitation and because it is present in high concentrations in leachate and in leachate-contaminated surface and ground water near the waste disposal site. Other conservative solutes are expected to behave in a similar fashion and could be simulated, given data on influent and background concentrations. Simulations were made with chloride concentration in the leachate equal to  $213,000,000 \text{ mg/m}^3$  (according to the results of leachate analysis). This value was considered as the average chloride concentration in leachate entering the upper sedimentary layer, the top of the 1D designed model. As an assumption, the chloride concentration was constant during all ten years. Figure 6 shows the chloride concentration breakthroughs in observation nodes during ten years.

As shown, after ten years, only the three first nodes (at depths 10, 20, 30 m, respectively) clearly show the contaminant increasing during time. According to results of geotechnical and geophysical studies which show the dense and compact geological formation in depths more than 30 m, the presented results in form of Fig. 6 are justifiable.

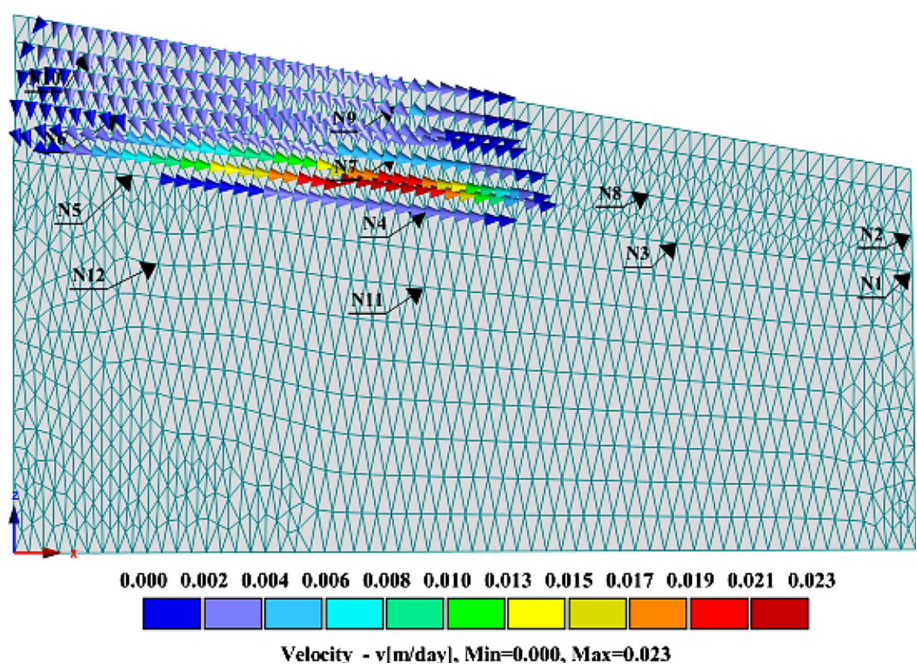
### 2D modeling

In this site, the 2D modeling of contaminant transport was followed by considering all gathering data and considerations in previous parts. Like the 1D case, the concentration of leachate in some observation points was focused to analysis the migration of contaminant in this

**Fig. 9** Contaminant contours after ten years



**Fig. 10** Velocity vectors of contaminant migration after ten years



site and its surroundings. In Fig. 7, the 2D designed model, flow boundary conditions and observation nodes are shown. The horizontal length of this model is about 900 m. In this part, the considered value for the constant flux boundary is based on results of the section “Hydrologic evaluation of site” and the Visual HELP output. The concentration breakthroughs in observation nodes are presented in Fig. 8.

The six observation nodes (N10, N9, N6, N7, N8 and N2) and four points (N5, N4, N3 and N1) were located in

sediments, the interfacial surface of sediment and bed rock, respectively. The nodes N12 and N11 were placed in bed rock. In this part of the modeling, the concentration of contaminant ( $213,000,000 \text{ mg/m}^3$  in 1D model) did not add to the boundary of the model beneath the waste disposal site and only the flux of a contaminant was used for analysis. As shown in Fig. 8 and according to contaminant concentration of N10, N9, N6 and N7, which are located exactly in different depths of the bottom of the landfill, the predominant flow is vertical. Results show that during

Fig. 11 3D model of study area

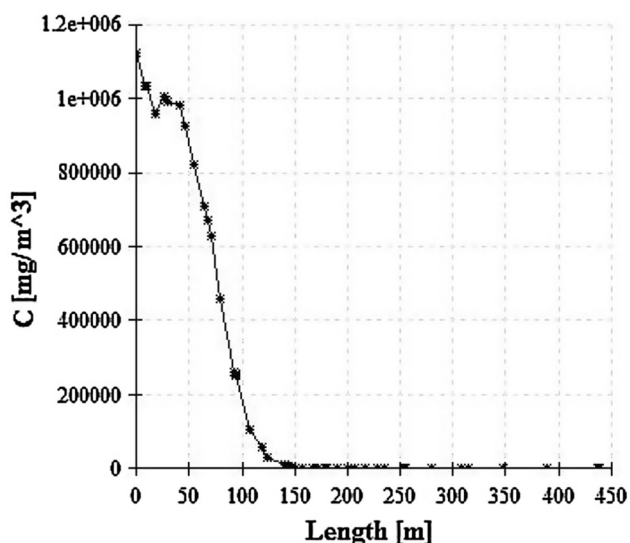
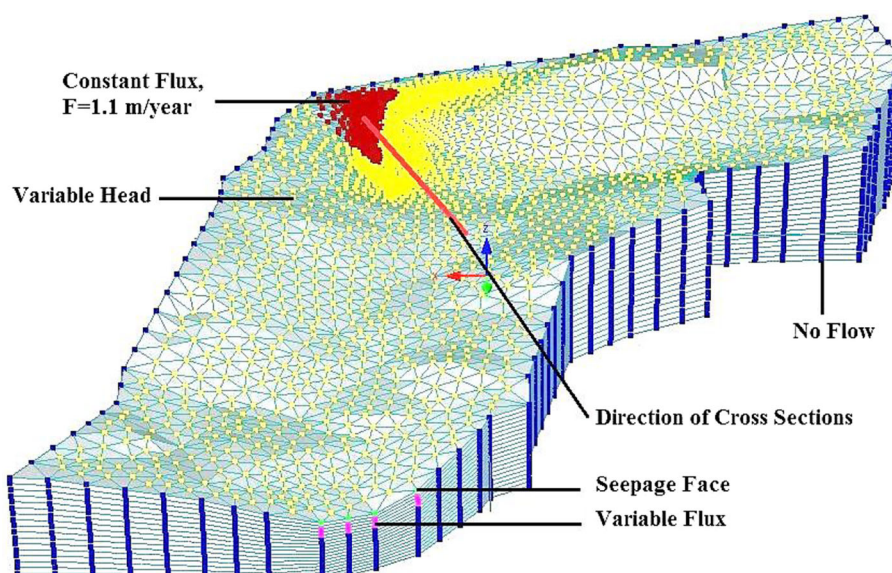


Fig. 12 Chloride concentration along cross section 1 after ten years

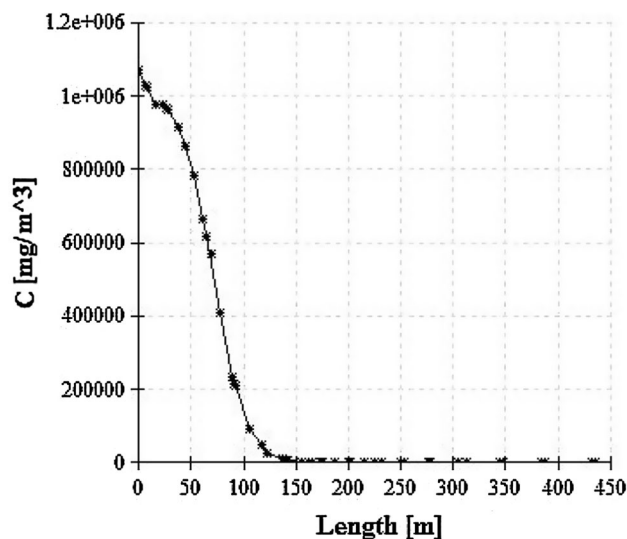


Fig. 13 Chloride concentration along cross section 2 after ten years

ten years, the contaminant plume did not reach to the down gradient of the waste disposal site (nodes N8, N3, N2 and N1) and this demonstrated the slow migration of contaminant in porous media. The result of the simulation, plotted as lines of equal chloride concentration at the end of ten years, is presented in Fig. 9. Limitations of the two-dimensional approach and assumptions and simplification made in the model must be considered in evaluations of these results, however. The velocity vectors of chloride migration are showed in Fig. 10.

According to Figs. 9 and 10 which show ten years contaminant transport modeling in this site, the contaminant plume migrated a short distance ( $<200$  m), and this is because of unsaturated porous media and high compaction of fine grain sediments. In addition, the high compaction of

a third material (M3) in a depth of more than 40 m created the natural ban for infiltration and movement of contaminant to the deep subsurface.

### 3D modeling

The 3D modeling of contaminant transport was followed as the final part of this study. In this section, with conjugation of all data (geological, geotechnical, geoelectrical and geochemical) and using the 1D and 2D models, the three dimensional model of flow and contaminant transport was constructed. The approximate area of the model is 350 hectares. In this model, five types of boundary conditions for water flow (no flow, constant flux, variable head, seepage face and variable flux) and two types of boundary

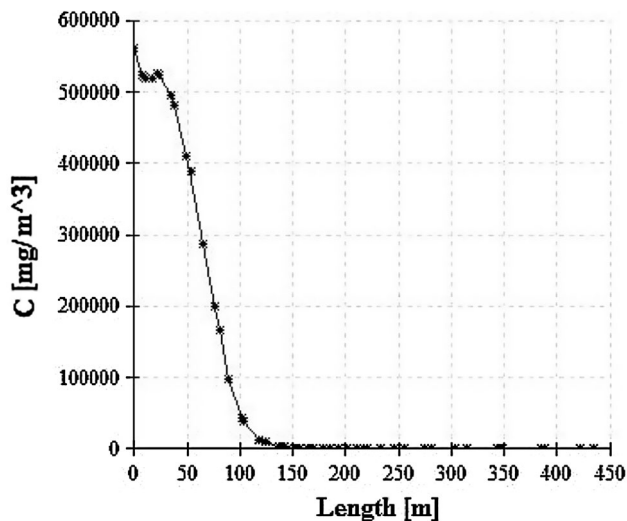


Fig. 14 Chloride concentration along cross section 3 after ten years

conditions for contaminant transport [no flow and third type (Cauchy)] were considered. Figure 11 shows the water flow boundary conditions in a constructed 3D model of the study area. For contaminant transport boundary conditions, the Cauchy type boundary conditions were considered in state for some of the water flow boundary conditions (constant flux, variable head, seepage face and variable flux).

Plume migration over a 10-year period was simulated with the HYDRUS model. Each 1-year interval consisted of ten equal time steps of 36.5 days. Reducing the time step by half caused no difference in concentrations in a 1-year test simulation. In this part, the chloride concentration in some positions and cross sections was used to evaluate contaminant infiltration and movement in the area. In each cross section, one point was located at beneath the waste disposal site and another point in the distance 450 m (in direction of topography; to the north of the catchment area). Three cross sections were selected between six observation points in different depths. In cross sections, the observation nodes, which were located at beneath the waste disposal site, were in depths of 10, 20 and 30 m and their corresponding points were in depths of 8, 17 and 25 m, respectively. The direction of cross sections was shown in Fig. 11 (red line). Note that all cross sections have the same direction but different depths. The change of chloride concentration between observation nodes 1, 2 (cross section 1), 3, 4 (cross section 2) and 5, 6 (cross section 3) are shown in Figs. 12, 13 and 14, respectively. As shown in these figures, during ten years of the onset of leachate seepage, pollution has been transformed approximately 150 m to downstream of the study area. In addition, the main motion of pollution is vertically and concentration of leachate has been decreased by increasing the depth.

## Conclusion

Results of 10 years of hydrologic evaluation of the Rasht waste disposal site by using the Visual HELP program showed that the mean rate of leachate percolation from this site is equal to 1.095 m per year. According to soil and rock cores which were prepared from the borehole, and based on a unified soil classification, the soil type in depth 0–13 m is SM (silty sand with gravel), depth 13–20 m is very dense and dry crushed rock, and depth 20–35 m is a high compaction SM type. From the interpretation of the resistivity sounding curves, a four-layer geoelectrical model indicated four subsurface hydrostratigraphical layers. These layers consisted of a surface layer (loose and fine grained topsoil), medium grained alluvium, a saturated layer, and high resistivity bedrock. The average thickness of the low resistivity saturated alluvial aquifer has been estimated to be about 42 m in the central and eastern part and about 10 m in the west part. Hydrochemical analysis of samples show that the surface and groundwater samples were not contaminated by waste leachate. Results show that during ten years of the onset of leachate seepage, contaminant plume migrated a short distance (<200 m) because of high compaction of fine grain unsaturated sediments. The high compaction of the third layer (M3) in depth more than 40 m produced the natural ban to infiltration and movement of contaminant to the deep subsurface. According to the 3D modeling using the HYDRUS program, pollution has been transformed approximately 150 meters to downstream of the study area. In addition, pollution moved dominantly vertically and concentration of leachate has been decreased by increasing the depth. The results of contaminant transport simulations indicate that using the supplementary data such as geological, hydrological and geophysical investigations in the modeling of solute transport can lead to more reliable assessment. Also, the constructed model can be used to predict movement of conservative solutes in the down gradient of the waste disposal site, calculate concentrations at specific times and locations, and analyze ground-water-quality management alternatives.

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