

Prediction of the transport and destination of lead, through the subsoil, from the leaching of a final disposal site of solid urban waste

G.M. Marielena¹, H.H. Anne Margreth², R.V. María Neftalí³, F.V. Georgina⁴

¹ Faculty of Engineering, National Autonomous University of Mexico, Coyoacan, Mexico City, C.P. 0450, Mexico

² Mexican Institute of Water Technology, Jiutepec, Morelos, C.P. 65550, Mexico

³ Engineering Institute, Autonomous University of Mexico, Coyoacan, Mexico City, C.P. 0450, Mexico

⁴ Faculty of Engineering, Autonomous University of Mexico, Coyoacan, Mexico City, C.P. 0410, Mexico

Presenting author email: mari_g_m@hotmail.com

Abstract

For this study, a hypothetical scenario was considered in which a leachate leak occurred in the evaporation pond and modeling of the reactive transport through the subsoil of the lead contained in the leachate from a sanitary landfill located in the State of Mexico in Mexico, using the geochemical code PHREEQC. Initially, a conceptual model of the environment was determined, in which it was identified that the study area is made up of volcano-sedimentary materials of firm to hard consistency and low permeability, up to a depth of 120 m and the water table is 60 m from depth. Sampling and analysis of leachate was carried out to know some of its physicochemical characteristics and lead content; with this, it was obtained that the pH of the leachate (6.5 to 7.89) indicates that the landfill is in a methanogenic phase. Laboratory analyzes evidenced the presence of lead in all the samples, in a range of 0.559 mg/L to 0.894 mg/L, which represents a low concentration of lead, compared to others reported for landfills.

For the modeling, the mechanisms of transport by advection and hydrodynamic dispersion, as well as adsorption, were considered, for which two surfaces were established, one of clay and the other of hydrated ferric oxide. The initial leachate solution was defined with the highest lead concentration found (0.894 mg/L) and a 60 m column was established for transport. The results obtained from the modeling indicated that all the lead contained in the leachate infiltrated in the subsoil of the sanitary landfill, will be retained in the environment, and will not be able to get dissolved up to a depth of 60 m, so that the lead will not will impact groundwater.

Keywords: urban solid waste, sanitary landfill, lead, modeling, transportation

1. Introduction

The generation of waste is a characteristic derived from the various activities of the human being, however, urban growth and poor management of these have caused them to become a problem that has led to the contamination of soil, water, atmosphere and has even caused damage to public health [1]. The total generation of urban solid waste in Mexico for the year 2017 was 120,128 t/day [22], of which 79% were sent to final disposal; This amount contrasts with other countries such as Switzerland, the Netherlands, Germany, Belgium, Sweden, Austria and Denmark; in which the final disposal in sanitary landfills is less than 5%. In what corresponds to recycling, Mexico recycles approximately 9% of waste, in contrast to countries such as; Germany, Belgium, Sweden and Denmark, whose recycling percentages are above 35% or Canada, which recovers 24% of its waste. This condition in Mexico in terms of urban solid waste management practices, causes a loss of waste that can be reincorporated into a production system and consequently reduce the demand and exploitation of new resources [13].

In Mexico, the sector of urban solid waste and the operation of final disposal sites is of particular interest due to its important role in waste management and sustainability [3]. The Official Mexican Standard NOM-083-SEMARNAT-2003 defines final disposal sites as the place where urban solid waste and special handling waste are deposited permanently. Likewise, it classifies these sites as controlled sites and uncontrolled sites or dumps, according to the infrastructure and management and operation conditions. It also establishes the parameters for the selection, design, operation and closure of final disposal sites, and indicates the basic infrastructure to control the vectors that could cause an environmental impact. In Mexico, the final disposal of urban solid waste in sanitary landfills or controlled final disposal sites predominates [13]; it should be noted that of the 2,203 existing final disposal sites [22], the majority are dumpsites [18].

Under confinement conditions, residues represent a potential source of contamination and, if containment failures exist, contaminants can be released [19]. Of the total final disposal sites that exist in Mexico (2,203), almost 50% of these do not have basic infrastructure for environmental protection such as geomembrane, infrastructure for leachate collection, biogas collection, etc. [22] and there are final waste disposal sites cataloged by the environmental authority as contaminated sites [21].

One of the main products generated in a final disposal site is leachate and these are associated with various environmental problems [24]. Four groups of contaminants are present in leachates: dissolved organic matter (volatile fatty acids and more refractory organic matter such as humic substances), inorganic macro compounds (Ca^{2+} , Mg^{2+} , Na^+ , K^+ , NH_4^+ , Fe^{2+} , Mn^{2+} , HCO_3^-), heavy metals (Cd^{2+} , Cr^{3+} , Cu^{2+} , Pb^{2+} , Ni^{2+} , Zn^{2+}), xenobiotic organic compounds present in low concentrations (aromatic hydrocarbons, phenols, pesticides, etc.) and microorganisms such as total and thermo-tolerant coliforms [29]. Other compounds can be found in very low concentrations, such as: B, As, Se, Ba, Li, Hg and Co, with a less relevant importance [4].

Among the main contaminants contained in leachates are heavy metals, which represent a long-term threat to human health, because they are toxic and do not degrade over time [28]. Two of the most abundant metallic contaminants present in leachates are lead and zinc [11]. There are some studies carried out in landfills in different parts of the world and in Mexico, such as that of Abriga et. to the. [1], De León-Gómez et. to the. [7], Fernandez et. to the. [9], Stefania et. al [23], and Nyirenda and Mwamba [17], in which the presence of Pb in groundwater is evidenced in concentrations higher than the permissible limits for drinking water quality indicated by the World Health Organization (WHO), which establishes a concentration of 0.01 mg/L for Pb [27] and is the predominant contaminant in the cited studies.

Table 1 shows the concentrations reported for each of the mentioned studies, where the values that are above the permissible limits established by the WHO for heavy metals are highlighted in bold; also, the depth of the static level reported for the groundwater below the sanitary landfills is indicated.

Table 1 Heavy metal concentrations in groundwater derived from leaching from landfills. Own elaboration

No.	STUDY AREA	METAL CONCENTRATIONS IN GROUNDWATER (mg/L)					Prof. of the static level (m)	Reference
		Parameter	No. of samples	Average	STD.	Min.		
1	Municipal sanitary landfill of the city of Linares, Nuevo León.	Pb	5	0.02	0.03	0.0001	0.0704	2.5-20 De León-Gómez et al. (2015)
		Cr	4	0.01	0.002	0.006	0.012	
		Zn	9	0.44	0.81	0.005	2.5	
		Fe	7	0.31	0.15	0.154	0.64	
		As	11	0.001	0.001	0.0004	0.0049	
		Mn	10	0.37	0.71	0.001	1.85	

No.	STUDY AREA	METAL CONCENTRATIONS IN GROUNDWATER (mg/L)					Prof. of the static level (m)	Reference	
		Parameter	No. of samples	Average	STD.	Min.			Max.
2	Revdalen Landfill, Norway	Pb	538	0.002	7.30	0.000	0.069	20.00	Abriga <i>et al.</i> (2020)
		Cr	205	0.0009	7.90	0.000	0.111		
		Zn	247	0.22	789.0	0.000	5.739		
		Cu	198	0.01	51.00	0.000	0.583		
		Cd	212	0.0003	1.400	0.000	0.011		
		Hg	113	0.0021	14.10	0.000	0.145		
3	Landfill in the alpine region. North of Italy	Pb	1004	0.005	-	0.001	0.050	25-30	Stefania <i>et al.</i> (2018)
		Cr	1004	0.05	-	-	-		
		As	1004	0.013	-	-	-		
		Fe	1004	3.58	-	-	-		
		Cu	1004	0.02	-	0.001	0.010		
		Ni	1004	0.06	-	0.001	0.010		
4	Pacara Pintado landfill in northwestern Argentina	Mn	1004	0.51	-	0.001	0.020	2.56	Fernández <i>et al.</i> (2013)
		Pb	13	0.023	0.02	0.005	0.075		
		Cr	13	0.006	0.005	0.0005	0.019		
		Zn	13	0.068	0.027	0.030	0.120		
		Fe	13	5.04	4.92	0.120	17.200		
		Cd	13	0.0002	0.0001	0.0001	0.0006		
5	Chunga Landfill, Lusaka, Zambia	Mn	13	6.30	5.29	0.09	18.64	-	Nyirenda and Mwamba (2022)
		Cu	13	-	-	0.013	0.037		
		Cd	13	-	-	0.004	0.089		
		Cr	13	-	-	-	0.233		
		Pb	13	-	-	0.062	0.660		
Zn	13	-	-	0.008	0.037				

In a sanitary landfill, the transport of contaminants focuses on knowing the fate of the substances that make up a leachate and their movement below the disposal site and in aquifer areas, in the case of subsurface migration in landfills, it has a predominantly vertical movement if there is no impermeable cover and the seepage rate of the bottom leachates can be estimated using Darcy's law through the following expression (equation 1) [24].

$$Q = -KA \frac{dh}{dl} \quad \text{Equation 1}$$

Where:

- Q = Leachate discharge per unit time, $m^3/año$
 K = Permeability coefficient, $m^3/m^2 \text{ year}$ or hydraulic conductivity
 A = Area in profile through which leachate flows, m^2
 dh/dl = Hydraulic gradient, m/m
 h = Head loss, m
 l = Length of the flow path, m

According to Sánchez [20], when a contaminant comes into contact with groundwater (in solution), two groups of processes take place; the first, in which the contaminant does not interact with the medium, generally it is about conservative or non-reactive contaminants, such as chloride, and it only moves with the flow of groundwater by advection, diffusion and dispersion. On the other hand, there are processes such as adsorption, precipitation, dissolution and different chemical reactions that occur when there is an interaction between the pollutant and the environment; in this case they are reactive solutes.

In this way, the modeling of the transport of a certain contaminant coming from the final waste disposal sites is an exercise that contributes to the evaluation of the possible risks associated with these sites, it allows estimating the scope that they may have when they are released into the environment and, consequently, preventive actions can be taken for its protection.

The objective of this research work is to predict the transport and destination of lead, through the subsoil, coming from a final disposal site for urban solid waste, through the analysis and modeling of scenarios that put at risk the health of the population and ecosystem integrity.

2. Methodology

The general methodology for the fulfillment of the objective contemplated various stages, from the investigation of the state of the art regarding the disposal of urban solid waste in Mexico, the selection of the site, the collection of information on the physical environment and operational aspects of the site and the field work focused both on the recognition of the study area and on sampling work, to then carry out the pertinent laboratory analyzes that allowed obtaining data on the concentrations of the pollutant that were included in the model and finally the establishment of the scope of a possible contamination through the chosen model.

The selection of the site was carried out through a spatial analysis through the implementation of Geographic Information Systems (GIS) and the extraction and superposition of data, in which the superposition of the location of final disposal sites and risk zones was carried out and/or vulnerabilities associated with geological and hydrometeorological disturbing agents relevant to these sites. Derived from said GIS analysis, a sanitary landfill located in the State of Mexico, Mexico was selected; which is located in an area of high risk of flooding (**Fig. 1**), as well as in an area classified as floodable for a return period of $T_r=100$; In addition, there is a population near the site at a distance less than that specified as a minimum by the applicable Mexican legislation. For the modeling, a physical scenario was considered in which the impermeable cover of the leachate evaporation lagoon of the studied sanitary landfill did not perform its function (a possible break) and an incorporation of mass concentrations of the selected contaminant occurred, in this case of lead metal. The location of the study sanitary landfill for the present investigation is shown in **Fig. 2**.

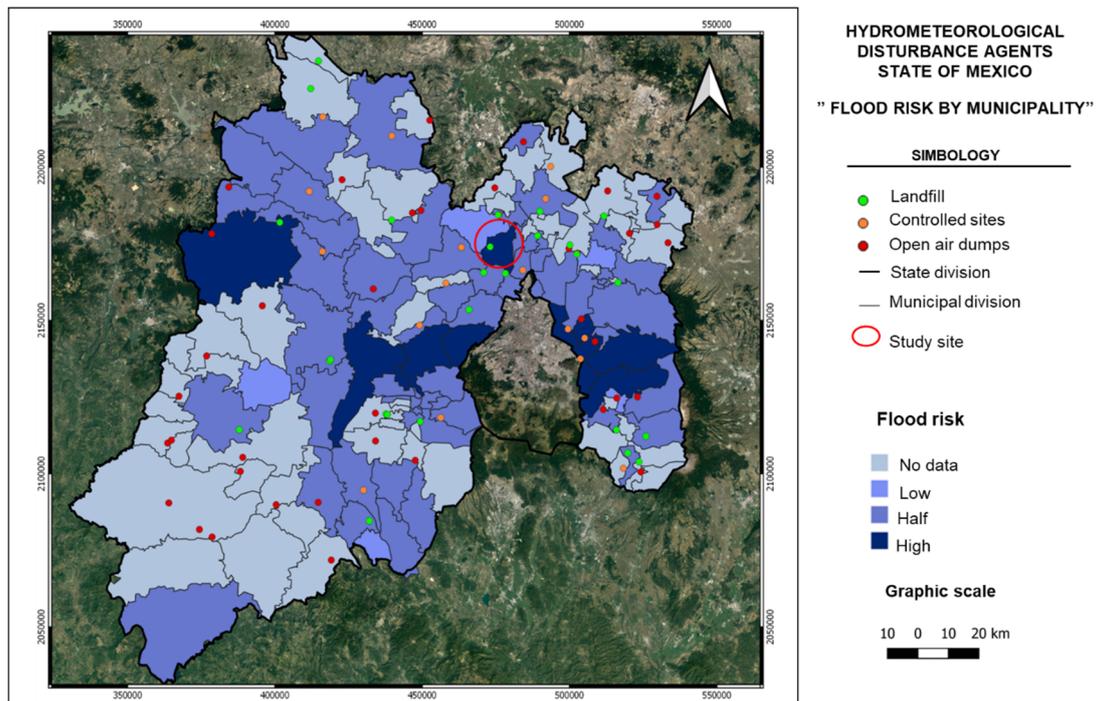


Fig. 1 Flood risk level by municipality for the State of Mexico, Mexico. Made from vector data [5]

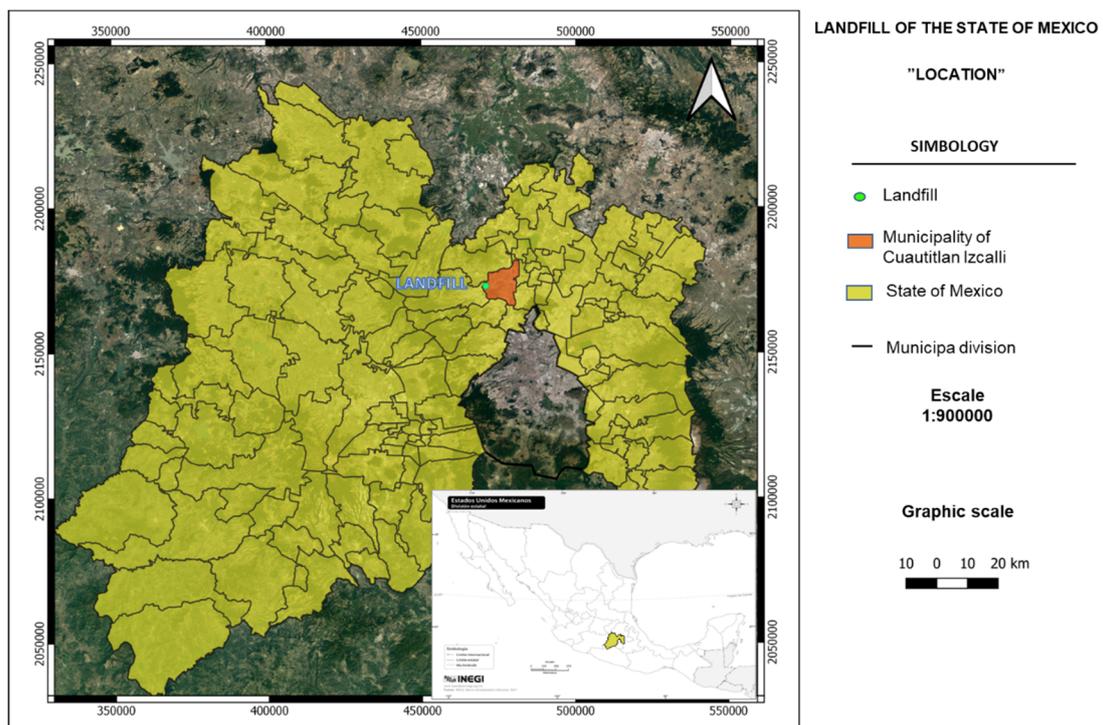


Fig. 2 Location of the sanitary landfill in the State of Mexico, Mexico. Made from vector data [5]

As part of the initial activities, a conceptual model of the study area was proposed, based on the prevailing environmental conditions of the landfill from a bibliographic search and previous geophysical, geotechnical, and geological studies; as well as chemical analyzes of leachate and groundwater. The elements that were considered for the conceptual model were: definition of a stratigraphic profile in the study area and the lithological classification, the hydraulic conductivity K , the physicochemical characteristics of the leachate, and the quality of groundwater.

Subsequently, in order to know the characteristics of the leachate generated in the study sanitary landfill and obtain the necessary parameters that were introduced in the transport model, a sample of the

leachate was carried out at three points of interest (**Fig. 3**), Sampling, and a groundwater sample was taken upstream of the landfill 2.7 km to the southwest. Likewise, some parameters were measured in the field such as: pH, electrical conductivity (EC), dissolved oxygen (DO), temperature (T) and Total Suspended Solids (TSS). Measurements were made with a portable multiparameter.

The samples of leachate and well water was analyzed in the Environmental Engineering laboratory of the Faculty of Engineering of the UNAM, for the determination of total lead according to the method NMX-AA-051-SCFI-2001 Determination of metals by Atomic Absorption in natural, drinking, wastewater and treated wastewater - test method. The equipment used was a GBC-Avanta Atomic Absorption Spectrophotometer. For lead determination, the flame atomic absorption method was used.



Fig. 3 Location of leachate sampling points in the sanitary landfill of the State of Mexico, Mexico

The lead transport mechanisms used in the modeling were advection and hydrodynamic dispersion, as well as adsorption as an attenuation mechanism because this process is one of the most relevant in the interaction between metals and soils; lead is also known to adsorb to soil constituent surfaces, such as clay, oxides, hydroxides, oxyhydroxides, and organic matter [8]. For this purpose, the geochemical code PHREEQC Version 3 was used. This code is written in the C++ programming language and with it various aqueous geochemical calculations can be performed [26], it is developed by the United States Department of Geological Research (USGS) and it can model one-dimensional transport processes such as dispersion, advection, and diffusion. Transport processes can be combined with kinetic and equilibrium chemical reactions that allow the simulation of the simplest reactive transport codes [25].

The modeling considers the transport of lead in the unsaturated zone, due to the fact that the depth of the water table in the study area is not very shallow. According to records of the static level in wells of the CONAGUA piezometric network located in the surroundings of the sanitary landfill, the depth of the water table is approximately 60 m. In this way, for the modeling of reactive transport, a column with a length of 60 m was established, which corresponds to the distance that dissolved lead would have to be transported to reach the groundwater level and impact groundwater; this column was defined by 60 cells with a length of 1 m each.

3. Characteristics of the landfill in the State of Mexico, México

The study sanitary landfill is operated by a private company and began operations in 2010, it has a 30-year concession. It has an area of 54 ha and has a capacity of 1,500 tons/day; operates 7 days a week. The cargo vehicles that arrive to deposit the waste are mainly public service trucks and private cargo transport.

Regarding the infrastructure of the sanitary landfill, it has an office and parking area, a weighing area equipped with two scales, a mechanical workshop area for the maintenance of the heavy machinery used to operate the cells. It has compacting tractors and chain-type tractors to compact and store the waste, as well as cargo trucks to transport the roofing material that is extracted from a bank of materials located on the same site. Likewise, it has a rainwater channel, in which wastewater is discharged irregularly, coming from the residential area adjacent to the landfill.

The cells that make up the sanitary landfill, both those that have already been completed and those that are in the process of being filled, have geotextile and geomembrane at the base for waterproofing and soil protection, according to what was mentioned by site personnel.

In addition, the sanitary landfill has a system for collecting leachate by pumping, which is stored in the evaporation lagoon, and is also recirculated. The leachate evaporation lagoon has a 200g geotextile and a 60-thousandth caliber geomembrane for waterproofing. Regarding the biogas generated, in the sanitary landfill there are 28 wells for the collection and burning of the gas.

As regards the environmental monitoring part, semi-annual monitoring of the composition of the leachate is carried out, as well as the quality of the groundwater. Leachate monitoring is carried out in the leachate lagoon and groundwater monitoring is carried out upstream, in a water extraction well for services; and downstream, at the Rosario Dam. Likewise, biogas monitoring is carried out. It should be noted that the sanitary landfill does not have groundwater monitoring wells.

4. Parameters used in the modeling

To model the transport of lead, it was necessary to define certain physical parameters related to the characteristics of the medium and physicochemical parameters for the contaminating solution. The parameters used in the model were the following: an infiltration solution was considered, which is represented by the leachate from the sanitary landfill that presents the highest concentrations of total lead (sample LIX-02-CARC1), whose concentration is 0.894 ppm; with a pH of 7.89, a temperature of 26 °C and a density of 1.00 g/cm³. The chloride and sodium concentrations used were 200 ppm and 400 ppm, respectively; considered in the literature as the maximum value for leachate in mature landfills (over 10 years old) [24].

The bioavailability and movement of a toxic contaminant in clay mineral soils is affected by adsorption and desorption reactions [14]. In this way, two adsorption surfaces were used in the modeling, a montmorillonite clay adsorption surface and another hydrated ferric oxide adsorption surface, since iron oxides in the soil are important because they exist practically in any type of soil, even in small quantities [2]. The adsorption surface for hydrated ferric oxide (Hfo) was defined from predetermined data from the PHREEQC database that are derived from Dzombak and Morel (1990) [26].

Likewise, the following characteristics were established for the surfaces: a specific surface of 754.40 m²/g for clay, according to data of typical values for this material. For hydrated ferric oxide, a specific surface area of 600 m²/g, value taken from the PHREEQC database.

For the reactive transport modeling, a column with a length of 60 m was established, defined through 60 cells with a length of 1 m each. This is based on the fact that the depth of the water table in the study area is 60 m, which would correspond to the distance that dissolved lead would have to be transported to reach the water table and impact groundwater.

To calculate the speed at which the particles traverse the porous medium, the real speed was calculated from the Darcy speed, using the following equations:

$$V = -K \frac{dh}{dl} \quad \text{Equation 2}$$

Where:

V	=	Darcy velocity, m/year
K	=	Permeability coefficient, m/year
dh/dl	=	Hydraulic gradient, m/m
h	=	Head loss, m
l	=	Length of the flow path, m

$$V_{real} = \frac{V}{me} \quad \text{Equation 3}$$

Where:

V_{real}	=	Real velocity, m/year
V	=	Darcy velocity, m/year
me	=	Effective porosity

Considering a permeability coefficient of 0.32 m/year, a head loss and flow path length equal to 60 m and substituting these values in equation 2, a Darcy velocity equal to -0.32 m/year was obtained.

$$V = \left(-0.32 \frac{m}{year}\right) * \frac{60 m}{60 m}$$

$$V = -0.32 m/year$$

To calculate the real speed, the Darcy speed obtained and the effective porosity were substituted in equation 3, which in this case is considered equal to 19%, the value assigned for a medium made up of silts, according to data from the literature. In this way, a real velocity of 1.68 m/year was obtained.

$$V_{real} = \frac{-0.32 m/year}{0.19}$$

$$V_{real} = 1.68 m/year$$

From the actual calculated velocity, a time step for each cell was estimated equal to 9×10^7 s, which represents the time it takes for the solution in the column, in this case the leachate, to travel to the next cell. The establishment of this parameter was defined from the hydraulic conductivity of the subsoil of the sanitary landfill, which has a value of 1×10^{-6} cm/s or 1×10^{-8} m/s for sandy silt. It is assumed that the value of the hydraulic conductivity is constant, since the medium is homogeneous up to the depth of interest.

For the contaminant solution front to move through the entire column, 60 shifts were established, equal to the number of cells initially defined; In addition, a forward flow direction was considered.

To calculate the dispersivity coefficient, various empirical and theoretical relationships have been determined, such as equation 4, which is a function of the length traveled by the flow [25].

$$aL = 0.1 * Lx \quad \text{Equation 4}$$

Where:

aL = Coefficient of longitudinal dispersivity
 Lx = Length traveled by the flow

It is worth mentioning that, according to Freeze and Cherry [10], dispersion studies reported in the literature have found that longitudinal dispersivity is in a range of 0.1 to 10 mm for homogeneous sandy materials and under controlled laboratory conditions. Normally the dispersivities values in field systems are much higher, with values up to 100 m for longitudinal dispersivity and up to 50 m for lateral dispersivity.

The value that was considered in the modeling was calculated from equation 4, considering a cell length equal to 1 m, with which a dispersivity coefficient of 0.1m.

Finally, for the calculation of the effective diffusion coefficient, the expression of equation 5 was used, for which an effective porosity equal to 19% was considered for silt, a coefficient of 2, designated for consolidated materials and a diffusion coefficient for lead, with dilution in water at 25°C equal to $9.45 \times 10^{-10} \text{ m}^2/\text{s}$ [16]. It is worth mentioning that the diffusion coefficient considered for common ions in groundwater (Na^+ , K^+ , Mg^{2+} , Ca^{2+} , Cl^- , HCO_3^- , SO_4^{2-}) have diffusion coefficients in a range of 1×10^{-9} a $2 \times 10^{-9} \text{ m}^2/\text{s}$ at 25°C, according to Robinson and Stokes [10].

$$D * = Dm * (m_e)^c \quad \text{Equation 5}$$

Where:

D^* = Effective diffusion coefficient
 D_m = Diffusion coefficient
 m_e = Effective porosity
 c = Coefficient (1.8 to 2.0 for consolidated materials; 1.3 for unconsolidated sands)

In this way, an effective diffusion coefficient equal to $3.41 \times 10^{-11} \text{ m}^2/\text{s}$, was obtained, as can be seen in the following calculation.

$$D^* = 9.45 \times 10^{-10} \text{ m}^2/\text{s} * (0.19)^2$$

$$D^* = 3.41 \times 10^{-11} \text{ m}^2/\text{s}$$

5. Results and discussion

5.1 Conceptual Model

The review of bibliographic information and background information in relation to the characteristics of the physical environment of the site, such as: geophysical, geotechnical and geological studies, allowed defining a conceptual model of the study area. A lithological profile was made, from a section in a southeast-northwest direction (section A A'), with a length of 800 m, which crosses the sanitary landfill polygon and the leachate evaporation lagoon; as shown in **Fig. 4**.

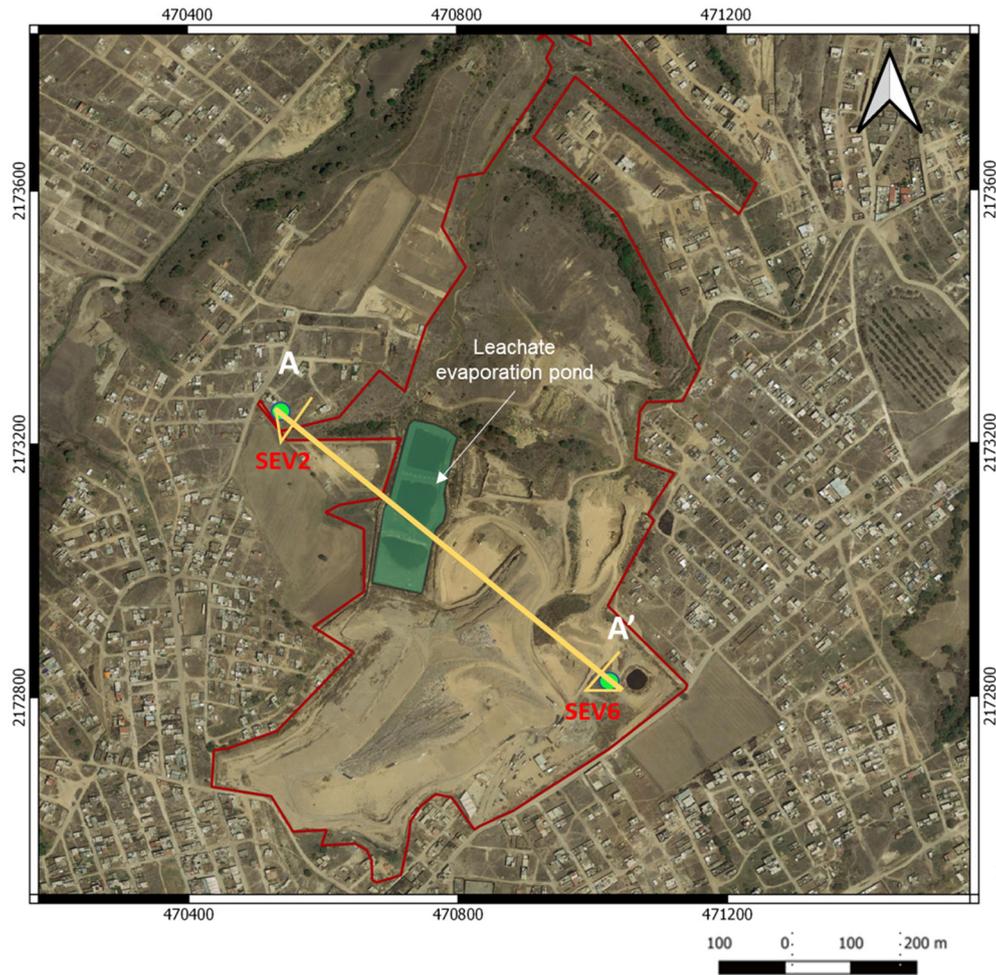


Fig. 4 Lithological section line AA' in the sanitary landfill

Based on geophysical data from vertical electrical soundings (VES's) and information from standard penetration soundings (information provided by the management of the sanitary landfill) previously carried out in the area, it was determined that the lithology that predominates in the site up to a depth of 120 m, corresponds to volcano-sedimentary materials, specifically intercalations of sandy silt tuffs of firm to hard consistency and very compact brown to reddish silty sands; with low to medium permeabilities. Underlying the tuffaceous materials is volcanic rock, of highly permeable rhyolitic and andesitic materials, which would be forming the deep aquifer in the study area, as shown in **Fig. 5**. It is worth mentioning that the type of soil prevailing in the landfill area is Vertisol, and it corresponds to black or brown soils with more than 30% expandable clays, they have a high moisture retention capacity and are good absorbers of toxic substances [15].

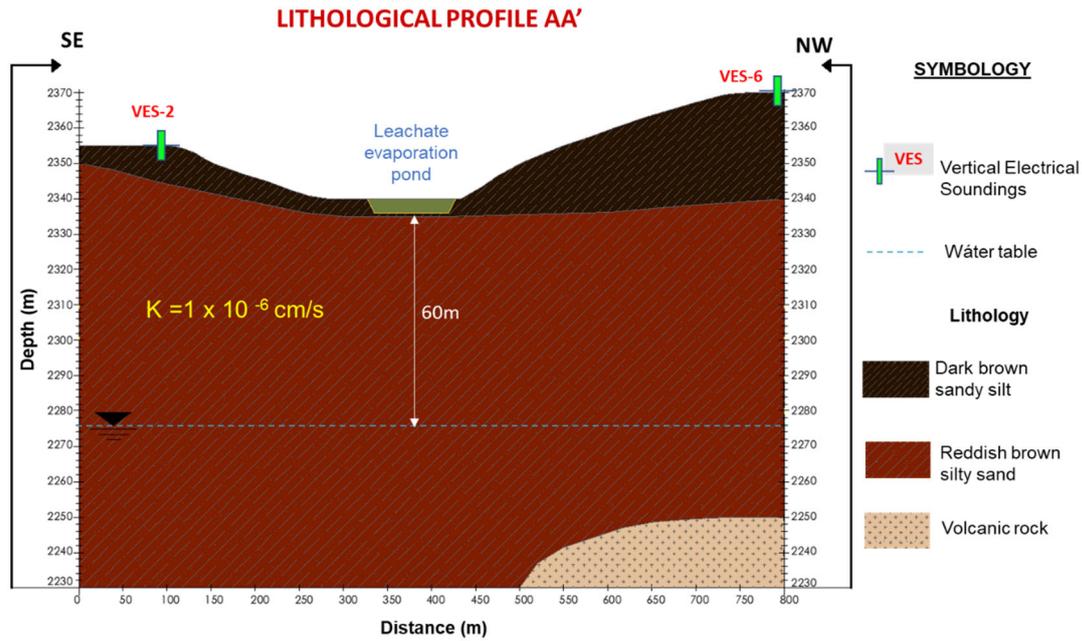


Fig. 5 Conceptual model of the site. Prepared from geoelectrical data from VES (Geophysical survey, 2009) and soil mechanics. (Study of soil mechanics, 2009).

The permeability of the materials in the unsaturated zone is low to medium, with a hydraulic conductivity value of $1 \times 10^{-6} \text{ cm/s}$. Regarding the water table, a water table is considered at a depth of 60 m, because CONAGUA [6] indicates that the depths of the static level to the south of the Cuautitlan-Pachuca aquifer, area of interest, varies between 65 and 120 m deep.

In this way, for the modeling, it was considered that the lead contained in the leachate's infiltrates and transports in a 60 m long column of tuffaceous sandy silt materials and poorly permeable sandy silt, until reaching the water table.

5.2 Field analytical results

The results of the analytical measurements with field equipment are shown in **Table 2**, where it is observed that the pH of the leachates is in a range of 6.5 to 7.89 and the temperature oscillates between 26 °C and 41 °C, being the sample LIX-01-LAG, taken in the direct discharge to the leachate lagoon, the one with the lowest pH and the leachate samples taken from the pits in the inactive cell (LIX-02-CARC1 and LIX-03-CARC2), present a higher pH. On the other hand, the water sample has a more alkaline pH. The rest of the parameters measured in the field for the leachates are above the quantification limit of the field equipment.

Table 2 Result of field measurements in leachate and water from the extraction well

ID	UTM coordinates		Matrix	Field parameters				
	X	Y		pH	EC (mcS)	TSS (PPT)	T (°C)	DO (ppm)
LIX-01-LAG	470804.23	2173216.45	Leachate	6.52	>20	>10	30.10	-
LIX-02-CARC1	470868.44	2172732.77	Leachate	7.89	>20	>10	26.00	-
LIX-03-CARC2	470589.41	2172631.42	Leachate	7.89	>20	>10	41.50	-
MA-01-POZO VIRREYES	468108.82	2172193.06	Ground-water	8.20	0.16	0.08	24.20	0.23

5.3 Laboratory analytical results

The results of the laboratory analysis for the leachate and water samples for the determination of lead are presented in **Table 3**, where it is observed that all the samples present lead concentrations. The highest concentration occurs in the sample LIX-02-CARC1, with a concentration of 0.894 mg/L, and the lowest concentration occurs in the sample LIX-03-CARC2, with a concentration of 0.559 mg/L. Obtaining an average concentration of 0.68 mg/L of lead in leachate.

On the other hand, the water sample taken upstream of the sanitary landfill shows lead concentrations below the equipment quantification limit.

Table 3 Lead concentrations in leachate and extraction well water.

ID	UTM Coordinates		Matrix	Analysis of the sample (mg /L)		
	X	Y		Pb Method NMX- AA-051-SCFI- 2001	*SD	**LQ
LIX-01-LAG	470804.23	2173216.45	Leachate	0.590	0.008	0.2
LIX-02-CARC1	470868.44	2172732.77	Leachate	0.894	0.003.	0.2
LIX-03-CARC2	470589.41	2172631.42	Leachate	0.559	0.002	0.2
MA-01-POZO VIRREYES	468108.82	2172193.06	Groundwater	< 0.2	0.001	0.2

* Standard Deviation
** Limit of Quantification

The field analytical determinations allowed to identify that the pH of the leachate from the study site is in a range of 6.5 to 7.89; so, it can be inferred that, according to Tchobanougus *et. al.* [24], the leachate is in a methanogenic phase. On the other hand, according to Yao [29], the pH values of the leachates classify it as an intermediate to stabilized leachate.

On the other hand, the temperature of the samples oscillates between 26 °C and 41 °C, finding the highest temperature in the sample from the pit of the inactive cell whose leachate level was deep and the lowest temperature in the pit with a shallow leachate level. The rest of the parameters measured in the field, such as: electrical conductivity, dissolved oxygen and total suspended solids, were found to be above the quantification limit of the equipment used.

The results of the leachate samples analyzed in the laboratory indicate that all the samples present lead concentrations. The highest concentration is found in the sample LIX-02-CARC1, with a concentration of 0.894 mg/L, and the lowest concentrations are found in the samples LIX-03-CARC2 and LIX-01-LGAG, with lead concentrations of 0.559 mg/L and 0.590 mg/L, respectively.

The lead found in greater quantities was present in the sample from one of the pits of an inactive cell, whose leachate level was shallow and with a pH of 7.89, slightly alkaline. On the contrary, the lowest concentration occurred in the leachate sample taken from the pit at a deep level, with a pH of 7.89. The sample collected directly from the leachate discharge to the evaporation pond, presented a pH of 6.52 and a lead concentration of 0.59 mg/L.

Lead concentrations found in sanitary landfill leachates can be considered low when compared to the range reported by Christensen *et. al.* [4] for lead in leachate from landfills, ranging from 0.001 to 5 mg/L. **Fig. 6** presents the position of the samples analyzed, with respect to the range of minimum and maximum lead concentrations reported for leachates according to Christensen *et. al.* [4].

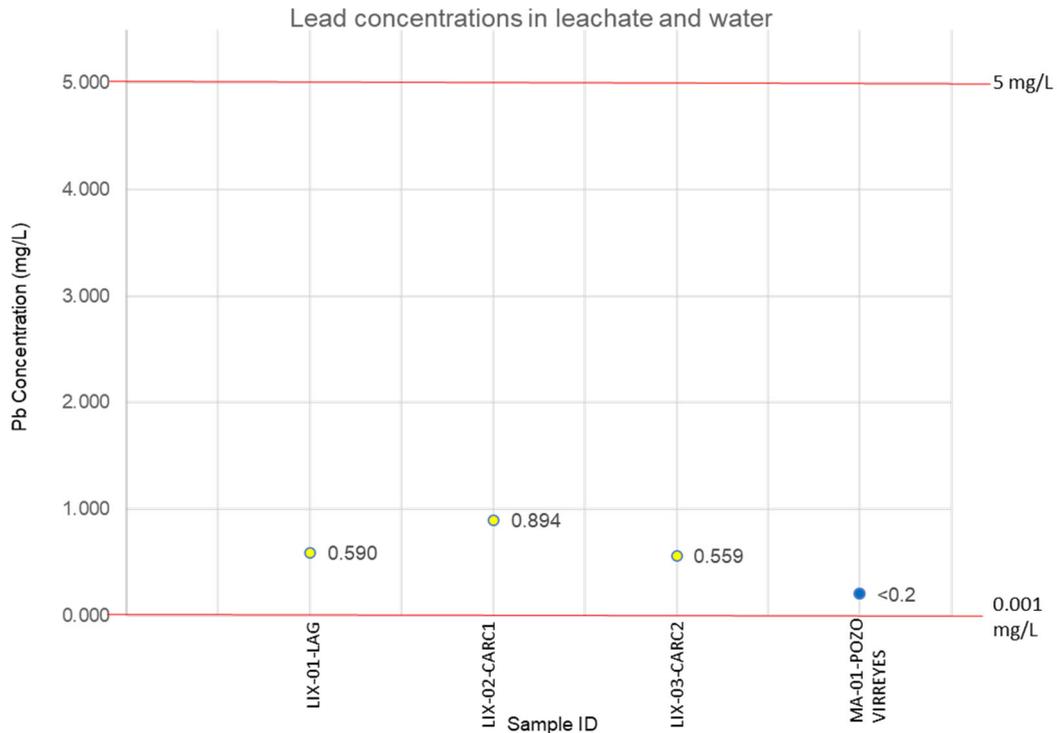


Fig. 6 Lead concentrations in leachate and groundwater. Leachate samples (yellow dots), water sample (blue dot), range of typical lead concentrations in leachate from landfills (red lines)

Regarding the groundwater taken upstream of the sanitary landfill, it has a pH of 8.2, slightly alkaline, an electrical conductivity of 0.16 mS, a dissolved oxygen concentration of 0.23 ppm and 0.08 ppt for total suspended solids. Regarding the presence of lead, the results indicated concentrations below the equipment's quantification limit, that is, less than 0.2 ppm.

The results of the concentrations of lead obtained in groundwater cannot be compared with the maximum permissible limit of the applicable national and international legislation for water for human use and consumption, because the ranges of the legislation are of the order of 0.01 mg/ L, are less than those quantified by the analysis equipment used; however, measurements of lead in groundwater were only considered to determine whether the area's water contains lead and were not included in the model.

5.4 Modeling

For this stage, a fictitious scenario was considered, assuming a leachate leak in the leachate evaporation lagoon and a modeling was carried out using the PHREEQC code, of the advective, dispersive and reactive transport of lead from the leachate generated in the sanitary landfill of the State of Mexico, Mexico, with a lead concentration of the initial leachate solution used for modeling equal to 0.894 mg/L, the maximum amount of lead found in the leachate from the study sanitary landfill. Likewise, the adsorption phenomenon was included as the main attenuation mechanism for lead. **Fig. 7** shows the input sheet coding of the code used for modeling reactive lead transport.

```

1 SOLUTION_MASTER_SPECIES
2 Montmorillonite Montmorillonite 0.0 367.017 367.017
3 SOLUTION_SPECIES
4 Montmorillonite = Montmorillonite
5 -log_k -45.027
6 -delta_h 58.373kcal
7 PHASES
8 Ca-Montmorillonite
9 Ca0.165Al2.33513.67010(OH)2 + 12 H2O = 0.165Ca+2 + 2.33 Al(OH)4- + 3.67 H4SiO4 + 2 H+
10 -log_k -45.027
11 -delta_h 58.373kcal
12 -vm 156.16
13 SOLUTION 0 Pulse solution leachate #solution injected
14 units ppb
15 pH 7.89
16 density 1.00
17 temp 26.0
18 Na 200000
19 Cl 400000
20 Pb 894
21 SOLUTION 1=60 background solution initially filling column
22 units ppb
23 pH 8.4
24 density 1.00
25 temp 24.7
26 Na 121840
27 Cl 31760
28 SELECTED_OUTPUT
29 -file grid4disp.csv
30 -selected_out true
31 -high_precision false
32 # set value for all identifiers to follow (lines 1 - 6)
33 -reset true
34 -simulation true
35 -state true
36 -solution true
37 -distance true
38 -time true
39 -step true
40 -percent_error true
41 -totals Cl Na Pb
42 SURFACE_MASTER_SPECIES
43 Hfo_s Hfo_sOH
44 SurFa_s SurFa_sMontmorillonite

```

Fig. 7 Modeling code entry sheet in PRHEEQC

The results of the Pb transport modeling indicate that, for a 60 m column, the amount of dissolved lead that would be reaching the end of the column would correspond to 3.54×10^{-10} mg/L, contained in different species, as shown. shown in **Table 4**. Regarding the amount of lead that would be retained on the two adsorbent surfaces considered for modeling, the results indicate an amount of 0.89 mg/L.

Table 4 Dissolved and adsorbed lead concentrations at the end of the column

Species	Dissolved		Adsorbed		
	Molality (mol/kg)	mg/L	Adsorbent surface	Molality (mol/kg)	mg/L
PbOH ⁺	8.16E-16	1.69E-10	Hydrated Ferric Oxide	4.315E-06	0.8941
Pb ⁺²	6.65E-16	1.38E-10	Montmorillonite	2.091E-09	0.0004
PbCl ⁺	2.02E-16	4.19E-11	-	-	-
Pb(OH) ₂	2.45E-17	5.08E-12	-	-	-
PbCl ₂	2.88E-18	5.97E-13	-	-	-
Pb3(OH) ₄ ⁺²	2.68E-20	5.56E-15	-	-	-
Pb2OH ⁺³	2.60E-20	5.39E-15	-	-	-
PbCl ₃ ⁻	1.74E-22	3.60E-17	-	-	-
Pb(OH) ₃ ⁻	7.19E-24	1.49E-18	-	-	-
PbCl ₄ ⁻²	1.84E-29	3.81E-24	-	-	-
Pb (OH) ₄ ⁻²	1.07E-38	2.21E-33	-	-	-
Total, dissolved	1.71E-15	3.54E-10	Total, adsorbed	4.32E-06	0.894

The amount of lead that is adsorbed is mostly retained by the hydrated ferric oxide surface, with a total of 0.89 mg/L, and to a much lesser extent by the clay, with an amount of 0.0004 mg/L, as shown. shown in **Table 5**.

Table 5 Lead concentrations adsorbed at the end of the column

Surface	Pb adsorbed	
	mol/kg	mg/L
Hydrated Ferric Oxide	4.315E-06	0.8941
Montmorillonite	2.091E-09	0.0004
Total	4.317E-06	0.894

In this way, all the lead contained in the initial leachate solution that will infiltrate through the 60 m column will be adsorbed by the surfaces considered in the model. The 0.894 mg/L of lead will remain adsorbed, reaching the end of the column an amount of dissolved lead equal to 3.54×10^{-10} mg/L, which can be considered negligible. Adsorption takes place predominantly on the surface made up of hydrated ferric oxide, leaving a small amount of lead adsorbed on the clay surface.

Fig. 8 shows the graph with the transport profile of Pb in the leachate for 32 years, where a reduction in aqueous concentrations is observed due to the adsorption of lead in the medium, which limits its transport only to the 1.5 m. In the graph of **Fig. 9**, the lead transport profile for different years is shown.

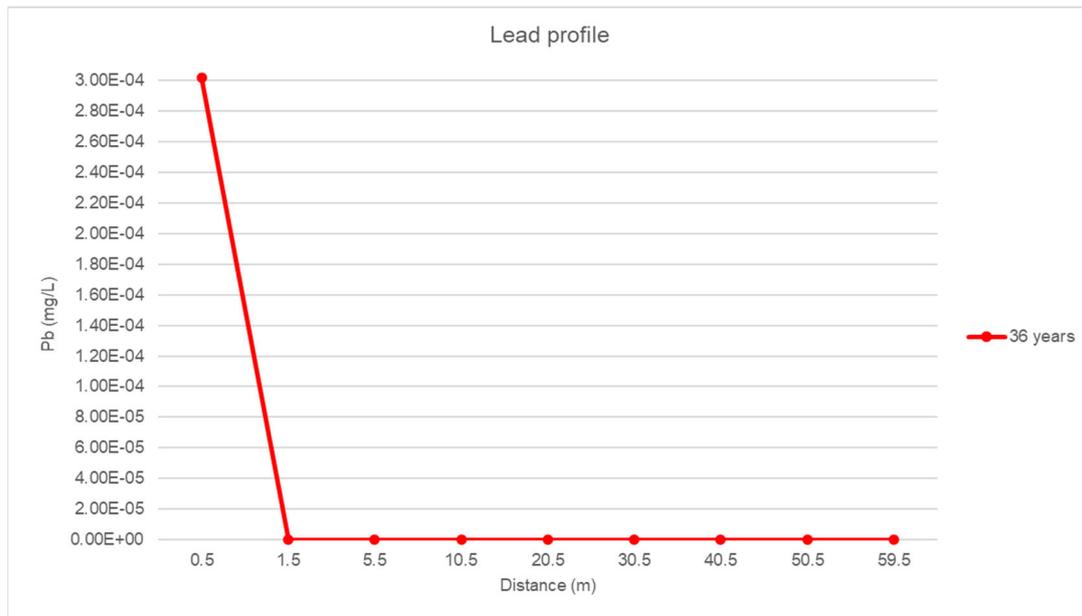


Fig. 8 Transport of Pb in leachate at 36 years

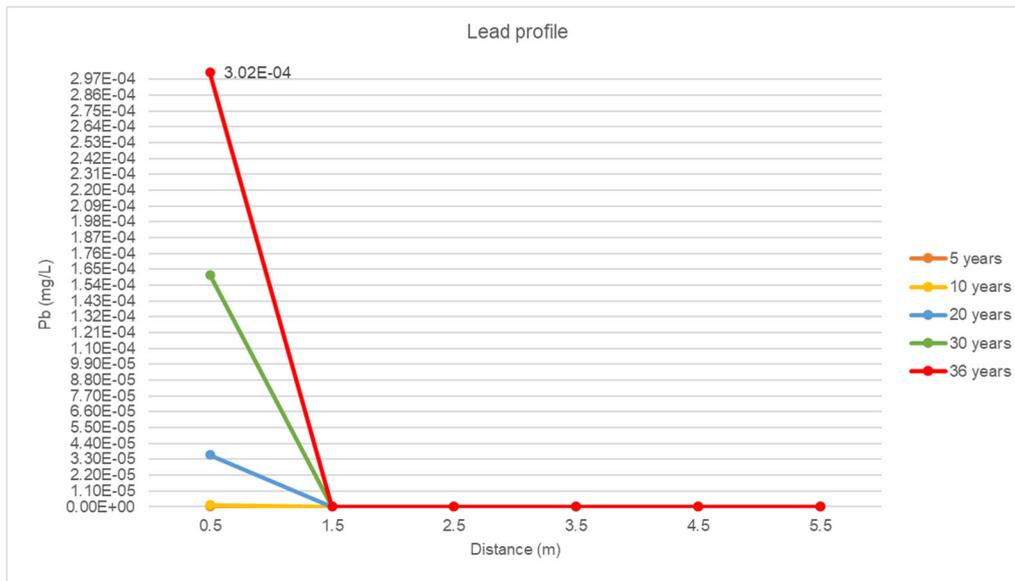


Fig. 9 Transport of Pb in leachate for different years

In the same way, in the graph of **Fig. 10**, the chloride transport profile for different years is shown; particularly for the 36 years all the cells will be washed with the leachate; In the graph it can be seen that the concentrations remain constant up to a distance of 35 m, to later decrease by 241 mg/L up to 60 m; This transport contrasts with that of lead, since chloride is a conservative substance, which does not react with the environment.

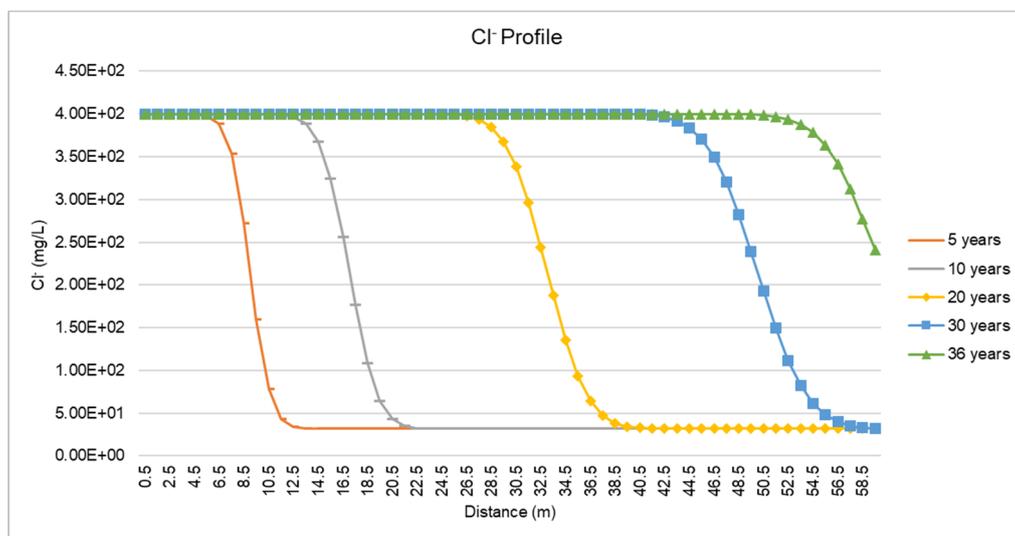


Fig. 10 Transport of Cl⁻ in leachate at 36 years

Based on the previous results, it can be observed that all the lead contained in the leachate that infiltrates the subsoil of the sanitary landfill, caused by a contribution of the contaminant derived from a possible break in the waterproofing, will be adsorbed by the medium and will not reach dissolved to a depth of 60 m, which means that the lead will not reach the water table and will not impact groundwater.

As previously mentioned, the site's own lithology is made up of firm to very hard sandy silts, with some intercalations of compact silty fine sands. The hydraulic conductivity value of the materials that make up the subsoil is of the order of 1×10^{-6} cm/s, which represents low permeability. These environmental conditions help prevent lead from reaching groundwater levels, give the environment an impermeable property, and represent a natural barrier to the migration of contaminants in the landfill. Likewise, the depth at which the water table is located, on the order of 60 m or more, is an important

factor so that lead does not reach groundwater, unlike other studies reviewed for the present investigation, in which evidence of lead was found in groundwater, where the depth of static levels oscillates between 2.5 m and 30 m deep.

6. Conclusions

The site selected for the study was a sanitary landfill located in the State of Mexico in Mexico, which presents conditions of potential risk to the environment, such as the presence of irregular settlements adjacent to the disposal site at a distance less than that established by legislation as a minimum; it is worth mentioning that these inhabitants arrived after the installation of the controlled disposal site.

On the other hand, it is located in an area of high risk of flooding and specifically in a floodable area for a return period of 100 years, which may represent a risk for the control of leachate if the hydraulic works are not efficient for the management of surface water runoff.

The leachate generated in the sanitary landfill has lead concentrations. The analytical results indicate the presence of lead in the three samples taken, with an average concentration of 0.68 mg/L. These concentrations can be considered low, compared to other concentrations reported for lead in leachates from solid urban waste landfills. Regarding the lead concentration of the groundwater sample upstream of the sanitary landfill, this is less than the quantification limit of the equipment used (< 0.2 mg/l); therefore, it is not possible to make a comparison with the permissible limits of the applicable national and international legislation (World Health Organization (WHO) and NOM-127-SSA1-2021), since both indicate a maximum permissible concentration of 0.01 mg/L for lead in water for human use and consumption.

Regarding the results of the modeling of the advective, dispersive and reactive transport of lead in the unsaturated zone and for the hypothetical scenario proposed as a rupture of the membrane in the leachate disposal zone, these indicate that all the lead contained in the leachate that infiltrates the subsoil of the sanitary landfill will be adsorbed in the medium, mostly due to the presence of iron oxides and will not reach a depth of 60 m dissolved, which means that lead will not reach the water table and will not impact groundwater. It should be noted that, particularly for the case of the site studied, the characteristics of the environment inhibit the dispersion of the contaminant.

Finally, the modeling of the transport and destination of lead carried out in the present study, allowed estimating the possible scope of this contaminant in the subsoil of a sanitary landfill that has unfavorable conditions, such as the presence of the surrounding population; assuming a leachate leak in the leachate evaporation pond.

This research represents a support tool for decision makers, since the modeling of the transport of a certain pollutant is an exercise that contributes to the evaluation of the possible risks associated with these sites, it allows estimating the scope they may have when they are released into the environment and, consequently, preventive actions can be taken to protect it.

7. Expressions of gratitude

The first author wishes to express gratitude for the support: to CONACYT for the scholarship provided; to the Postgraduate Studies Support Program (PAEP) of the UNAM, for the financial support provided, to the TERSA del Golfo Company, for its opening and collaboration for access to the sanitary landfill and its technical information and Mr. Eng. Jorge Sánchez Gómez for his cooperation in managing access to the study sanitary landfill.

8. References

1. Abiriga, D., Vestgarden, L., y Klempe, H. (2020). Groundwater contamination from a municipal landfill: Effect of age, landfill closure, and season on groundwater chemistry. *Science of the Total Environment*, 737, 140307.
2. Acebedo, S.O., Ortíz, H.E., Cruz, S.M., Cruz, S.M., Cruz, C.E. (2004). El papel de los óxidos de hierro en suelos. *Tierra Latinoamericana*, 22 (4), 485-497.
3. Cárdenas Moreno, P. R., Robles Martínez, F., Colomer Mendoza, F. J., y Piña Guzmán, A. B. (2016). Herramientas para la evaluación de riesgos sobre el ambiente y la salud, por la disposición final de residuos sólidos urbanos. *Rev. Int. Contam. Ambie.*, 47-62.
4. Christensen, T., Kjeldsen, P., L Bjerg, P., L Jensen, D., B Christensen, J., Baun, A., Heron, G. (2001). Biogeochemistry of landfill leachate plumes. *Applied Geochemistry*, 659-718.
5. CONABIO. (2022). Sistema Nacional de Información sobre Biodiversidad (SNIB). Obtenido de <http://www.conabio.gob.mx/informacion/gis/>
6. CONAGUA. (2015). Actualización de la disponibilidad media anual de agua en el acuífero Cuautitlán-Pachuca. Gerencia de Aguas Subterráneas.
7. De León, G. H., Cruz Vega, C., Dávila Pórcel, R., Velasco Tapia, F., y Chapa-Guerrero, J. (2015). Impacto del lixiviado generado en el relleno sanitario municipal de Linares (Nuevo León) sobre la calidad del agua superficial y subterránea. *Revista Mexicana de Ciencias Geológicas*, 32(3), 514-526.
8. EPA. (1999). Understanding variation in partition coefficient, K_d, values. Volume II: Review of Geochemistry and Available K_d Values for Cadmium, Cesium, Chromium, Lead, Plutonium, Radon, Strontium, Thorium, Tritium, and Uranium. Obtenido de <https://nepis.epa.gov/Exe/ZyNET.exe/00000DGE.TXT?ZyActionD=ZyDocument&Client=EPA&Index=1995+Thru+1999&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&IntQFieldOp=0&ExtQFieldOp=0&XmlQuery=>
9. Fernández, D., Puchulu, M., y Georgieff, S. (2013). Identification and assessment of water pollution as a consequence of a leachate plume migration from a municipal landfill site (Tucumán, Argentina). *Environ Geochem Health*, 36(3), 489-503.
10. Freeze R. A. y Cherry J. A. (1979). Groundwater. Englewood Cliffs, New Jersey: Prentice- Hall.
11. Ghayaza, M., Forestier, L., Muller, F., Tournassat, C., y Beny, J.-M. (2011). Pb (II) and Zn (II) adsorption onto Na- and Ca-montmorillonites in acetic acid/acetate medium: Experimental approach and geochemical modeling. *Journal of Colloid and Interface Science*, 361, 238-246.
12. Giusti, L. (2009). A review of waste management practices and their impact on human health. *Waste Management*, 29(8), 2227-2239.
13. Gobierno de México. (2020). Programa Nacional para la Prevención y Gestión Integral de los Residuos 2017-2018. Obtenido de https://dsiappsdev.semarnat.gob.mx/datos/portal/publicaciones/PNPGIR%20_2017-2018.pdf
14. Gu, X., Evans, L.J., Barabash, S.J. (2010). Modeling the adsorption of Cd (II), Cu (II), Ni (II), Pb (II) and Zn (II) onto montmorillonite. *Geochimica et Cosmochimica Acta*, 74(20), 5718-5728.
15. INEGI (2015). Guía para la interpretación de la cartografía. Edafología. Escala 1:250 000 Serie III. Aguascalientes, Aguascalientes: Instituto Nacional de Estadística y Geografía.
16. Lasaga A.C. (1998). Kinetic Theory in the Earth Sciences. New Jersey: Princeton Legacy Library
17. Nyirenda, J. y Mwamba, P. (2022). Impact of leachate on quality of ground water around Chunga Landfill Lusaka, Zambia and possible health risks. *Heliyon*. 8 (12), e12321.
18. Pérez, G. B. (2012). Riesgo de contaminación por disposición final de residuos. Un estudio de la región centro occidente de México. *Rev. Cont. Amb.*, 97-105.
19. Regadío M., Ruiz A., Rodríguez M., y Cuevas J. (2015). Containment and attenuating layers: An affordable strategy that preserves soil and water from landfill pollution. *Waste Management*, 408-419.

20. Sánchez, F. J. (2022). Hidrología Superficial y Subterránea. 2ª ed. Kindle Direct Publishing., 440 pp.
21. SEMARNAT. (2019). Visión Nacional hacia una gestión sustentable, cero residuos. Ciudad de México: Secretaría de Medio Ambiente y Recursos Naturales. Obtenido de https://www.gob.mx/cms/uploads/attachment/file/435917/Vision_Nacional_Cero_Residuos_6_FEB_2019.pdf
22. SEMARNAT. (2020). Diagnóstico básico para la gestión integral de los residuos. Ciudad de México. Obtenido de <https://www.gob.mx/cms/uploads/attachment/file/554385/DBGIR-15-mayo-2020.pdf>
23. Stefania, G., Zanotti, C., Bonomi, T., Fumagalli, L., y Rotiroti, M. (2018). Determination of trigger levels for groundwater quality in landfills located in historically human-impacted ar. *Waste Management*, (75), 400-406.
24. Tchobanoglous, G., Theisen, H. y Vigil, S. A. (1994). Gestión integral de Residuos Sólidos. Volumen I y II. Madrid, España: Mc Graw Hill.
25. Thyne, G. D. (2007). *PHREEQC 2007 Manual for Short Course*. Obtenido de https://www.researchgate.net/publication/293655512_PHREEQC_2007_Manual_for_Short_Course
26. USGS. (2022). PHREEQC Versión 3. Obtenido de <https://www.usgs.gov/software/phreeqc-version-3>
27. WHO, (2011). Guías para la calidad del agua de consumo humano: Cuarta edición que incorpora la primera adenda. Obtenido de <https://www.who.int/es/publications/i/item/9789241549950>
28. Ya, X., Xiangshan, X., Lu, D., Changxin, N., Yuqiang, L., y Qifei, H. (2018). Long-term dynamics of leachate production, leakage from hazardous waste landfill sites and the impact on groundwater quality and human health. *Waste Management*, 156-166.
29. Yao, P. (2017). Perspectives on technology for landfill leachate treatment. *Arabian Journal of Chemistry*, 10, S2567-S2574.